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NITROGEN LOADING FROM COASTAL WATERSHEDS TO RECEIVING ESTUARIES: NEW METHOD AND APPLICATION

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Abstract. In this paper we develop a model to estimate nitrogen loading to watersheds and receiving waters, and then apply the model to gain insight about sources, losses, and transport of nitrogen in groundwater moving through a coastal watershed. The model is developed from data of the Waquoit Bay Land Margin Ecosystems Research project (WBLMER), and from syntheses of published information.

The WBLMER nitrogen loading model first estimates inputs by atmospheric deposition, fertilizer use, and wastewater to surfaces of the major types of land use (natural vegetation, turf, agricultural land, residential areas, and impervious surfaces) within the landscape. Then, the model estimates losses of nitrogen in the various compartments of the watershed ecosystem. For atmospheric and fertilizer nitrogen, the model allows losses in vegetation and soils, in the vadose zone, and in the aquifer. For wastewater nitrogen, the model allows losses in septic systems and effluent plumes, and it adds further losses that occur during diffuse transport within aquifers. The calculation of losses is done separately for each major type of land cover, because the processes and loss rates involved differ for different tesserae of the land cover mosaic. If groundwater flows into a freshwater body, the model adds a loss of nitrogen for traversing the freshwater body and then subjects the surviving nitrogen to losses in the aquifer. The WBLMER model is developed for Waquoit Bay, but with inputs for local conditions it is applicable to other rural to suburban watersheds underlain by unconsolidated sandy sediments.

Model calculations suggest that the atmosphere contributes 56%, fertilizer 14%, and wastewater 27% of the nitrogen delivered to the surface of the watershed of Waquoit Bay. Losses within the watershed amount to 89% of atmospheric nitrogen, 79% of fertilizer nitrogen, and 65% of wastewater nitrogen. The net result of inputs to the watershed surface and losses within the watershed is that wastewater becomes the largest source (48%) of nitrogen loads to receiving estuaries, followed by atmospheric deposition (30%) and fertilizer use (15%).

The nitrogen load to estuaries of Waquoit Bay is transported primarily through land parcels covered by residential areas (39%, mainly via wastewater), natural vegetation (21%, by atmospheric deposition), and turf (16%, by atmospheric deposition and fertilizers). Other land covers were involved in lesser throughputs of nitrogen.

The model results have implications for management of coastal landscapes and water quality. Most attention should be given to wastewater disposal within the watershed, particularly within 200 m of the shore. Rules regarding setbacks of septic system location relative to shore and nitrogen retention ability of septic systems, will be useful in control of wastewater nitrogen loading. Installation of multiple conventional leaching fields or septic systems in high-flow parcels could be one way to increase nitrogen retention. Control of fertilizer use can help to a modest degree, particularly for optional uses such as lawns situated near shore. Conservation of parcels of accreting natural vegetation should be given high priority, because these environments effectively intercept atmospheric deposition. Areas upgradient from freshwater bodies should be given low priority in plans to control nitrogen loading, because ponds intercept much of the nitrogen transported from upgradient.

Key words: atmospheric deposition; Land Margin Ecosystems Research (LMER); nitrogen loading; nitrogen losses; Waquoit Bay; wastewater; watersheds.

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Introduction

Coastal waters are the most highly fertilized ecosystems on earth (Nixon 1986, Kelly and Levin 1986). Concern with nitrogen loading to coastal watersheds is widespread because loading is increasing, and because rates of primary production in coastal waters are largely limited by nitrogen supply (Valiela 1995, Caraco et al. 1987, Howarth 1988). Eutrophication of coastal waters, prompted by the increasing nitrogen loading from watersheds, is arguably the principal and most pervasive anthropogenic alteration to coastal ecosystems everywhere (GESAMP 1990, LMER Coordinating Committee 1992, National Academy of Sciences 1994).

The major sources of nitrogen to coastal watersheds include wastewater disposal, atmospheric contamination, and fertilizer use (Nixon and Pilson 1983, Lee and Olsen 1985, Nixon 1986, Valiela and Costa 1988, Culliton et al. 1989, Cole et al. 1993). Nitrogen from these sources is in turn carried to coastal waters via atmospheric deposition, sewage outfalls, river discharges, and groundwater flow.

Nutrient transport by surface runoff and streams has been well documented (Stanley 1988, Hinga et al. 1991, Correll et al. 1992, among others). Groundwater transport has received less attention, although it is important in coastal areas underlain by unconsolidated coarse-grained sediments, such as Cape Cod, Rhode Island, Long Island, and elsewhere, (Johannes 1980, Capone and Bautista 1985, Lee and Olsen 1985, Valiela and Costa 1988, MacIntyre et al. 1989, Valiela et al. 1990, Valiela et al. 1992). Even in Chesapeake Bay, where river transport has received much study, groundwater flow by seepage through the margins of Chesapeake Bay is equivalent to the flow through one of the major tributaries (Simmons 1992). In the Waquoit Bay estuarine system, nearly all the nutrient transport takes place by groundwater flow to the receiving estuaries.

We need to understand nitrogen delivery to coastal receiving waters for basic and applied reasons. Basic knowledge of nutrient dynamics of coastal landscapes needs to go beyond mass balance estimates of nitrogen transport (Valiela et al. 1978, Valiela and Teal 1979, Correll et al. 1992) to examine processes that control transport and fates of nutrients as they traverse the soil, the vadose zone, the aquifer, and seep out to the estuaries. Coastal zone managers and planners have an immediate need for nitrogen loading estimates from land to inform decisions on water quality management (Costa et al. 1992). Although methods for the estimation of nitrogen loading are still rudimentary, the use of such methods is already underway (Buzzards Bay Project 1991, for example). Some coastal municipalities have enacted bylaws that specify nitrogen loading thresholds that are not to be exceeded by new construction projects (Costa et al. 1992).

Existing methods to estimate nutrient loading to freshwaters (Dillon and Kirchner 1975, Schaffner and Oglesby 1978, Gibbs 1979, Norvell et al. 1979, Molot and Dillon 1993) and to coastal waters (Koppelman 1982, Lee and Olsen 1985, Nelson et al. 1988, Valiela and Costa 1988, Hayes et al. 1990, Hinga et al. 1991, Weiskel and Howes 1991) differ in assumptions about sources and losses, processes included, data required, and the types of land use categories used in the calculations. Below we examine and update these methods.

All estimates of nitrogen loading have uncertainties that derive from variations inherent in each step of the calculations, but the propagated uncertainty has seldom been estimated. Some assessment of uncertainty is essential to compare different watersheds, evaluate processes leading to loading, understand reliability and precision of loading estimates, and serve as a frame of reference in making management decisions (Hinga et al. 1991, Weiskel and Howes 1991). Replicate watersheds are seldom available, so it is not possible to obtain uncertainty estimates in the usual way. Propagation of error calculations (Meyer 1975), or newer resampling methods (Manly 1991, Potvin and Roff 1993) can provide estimates of uncertainties (G. W. Collins et al. *unpublished manuscript*).

The Waquoit Bay Land Margin Ecosystems Research project (WBLMER) has been evaluating mechanisms that transport nitrogen from the watershed to the Bay. To synthesize the results of WBLMER research (LMER Coordinating Committee 1992, Valiela et al. 1992, Lajtha et al. 1995, Sham et al. 1995, and unpublished WBLMER data), we needed a model protocol that incorporated the variety of mechanisms we have identified and measured, as well as included supplementary information from other sources. We therefore set out to build the WBLMER nitrogen loading model.

In this paper we first develop the WBLMER nitrogen loading model by combining results from Waquoit Bay with additional information synthesized from published sources. The intent of the model is to (1) estimate nutrient loading to watersheds by atmospheric, fertilizer, and wastewater nitrogen to watershed surfaces; (2) calculate losses of nitrogen from each source during travel through different land cover types and underlying soils, the vadose zone, the aquifer, and in freshwater bodies; and (3) estimate nitrogen loads to receiving waters by adding the amounts of nitrogen derived from each source that traverse the vadose zone and the aquifer downgradient from the different tesserae of the watershed mosaic. We also calculated associated uncertainties to assess the estimates produced by the model.

The model is applicable to coastal watersheds where dominant land uses range from forested to rural to suburban, where the major nutrient sources are atmospheric deposition, fertilizer use, and in situ septic systems. We also restrict our loading calculations to watersheds underlain by unconsolidated coarse-grained sediments, where delivery of nutrients to receiving waters is primarily via flow of groundwater.

In the Methods section we discuss the development of the model, assess background information used to determine each of the components, and then summarize the WBLMER model. Then, in the Results section we apply the model to calculate nitrogen loadings to Waquoit Bay, and use the data to estimate the (1) relative magnitude of different nitrogen sources; (2) effects of different types of land cover on losses of nitrogen; and (3) losses of nitrogen during transport through the watershed surface, the vadose zone, unsaturated sediments, and the aquifer.

Methods: Development of the WBLMER Nitrogen Loading Model

We first consider how nitrogen is delivered to watersheds by atmospheric deposition, fertilizer applications, and by wastewater disposal; and comment on its fate in vegetation and soils. Then we develop the details regarding the fate of this nitrogen in the vadose zone and the aquifer.

Nitrogen inputs and losses in watershed surface and soils

Atmospheric deposition.—To calculate nitrogen delivery to watersheds, our model uses local data on wet deposition of organic and inorganic nitrogen, and calculates dry deposition using a relationship derived from literature data.

Calculations of atmospheric deposition of nitrogen require updated and locally applicable deposition data because there is considerable geographic and secular variation in the chemistry of precipitation (Pack 1980, U.S. Environmental Protection Agency 1982, Stensland et al. 1986, Davies et al. 1992, Fricke and Beilke 1992, Shannon and Sisterson 1992, Ollinger et al. 1993). For example, concentrations of ammonium and nitrate in acid precipitation increased from the 1940s to the 1960s, but more recently have remained at high but fairly uniform levels in most of North America (Stensland et al. 1986, Dillon et al. 1988, Likens et al. 1990, Fricke and Beilke 1992). There are indications of decreased concentrations of nitrogen in precipitation after the mid-1980s in certain areas of North America and Europe (Dillon et al. 1988, Fricke and Beike 1992).

Most loading calculations ignore atmospheric sources, or only include wet deposition of inorganic atmospheric nitrogen. Recent work, however, has shown that there may be significant additional deliveries of atmospheric nitrogen through dry deposition and droplet capture (Lovett and Kinsman 1990, Hanson and Lindberg 1991, Lovett 1991, Meyers et al. 1991). Dry deposition occurs by accumulation of particles, and by adsorption of NO_x gases and ammonia through leaves. The importance of the different components of dry deposition varies from place to place. NO_3 in particles and HNO_3 vapor are the main sources in forests of the eastern U.S., while adsorption of NH_3 is relatively more important in agricultural areas (Lovett 1991).

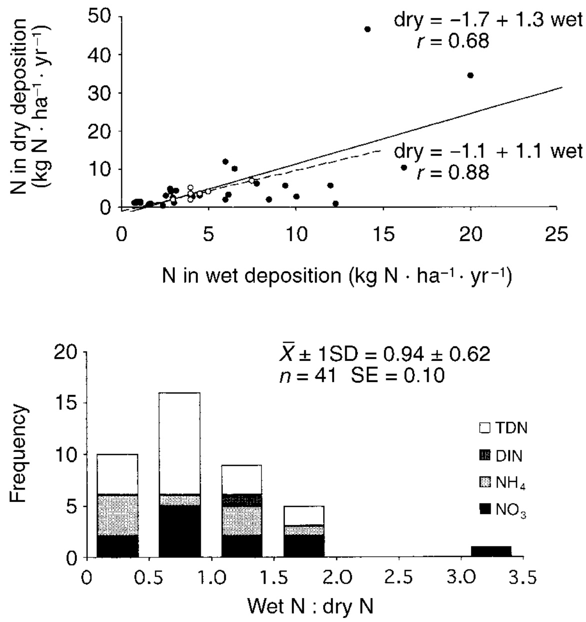

FIG. 1. Relationship between nitrogen content in wet and dry atmospheric deposition. Top: nitrogen delivery rates by dry vs. wet deposition, with linear regression for all points (data from Chan et al. 1984, Mozello et al. 1988, Lovett and Kinsman 1990, Lovett and Lindberg 1993, Prakasa et al. 1991, Sutton et al. 1991, Smiriniondi and Siskos 1992, and Munger et al. 1993; solid line) and for only International Forest Study sites (data from Lovett and Lindberg 1993; dashed line). Bottom: frequency distribution of ratios of wet to dry deposition for different species of nitrogen (data from Chan et al. 1984, Mozello et al. 1988, Lovett and Kinsman 1990, Prakasa et al.1991, Sutton et al. 1991, Lindberg and Lovett 1992, Smiriniondi and Siskos 1992, Lovett and Lindberg 1993, and Munger et al. 1983).

Unfortunately, there are relatively few measurements of dry deposition. We estimated dry deposition from the more readily available data on wet deposition by plotting dry vs. wet nitrogen deposition (Fig. 1, top). The slopes of the two regression lines were similar (Fig. 1, top). We also calculated ratios of wet to dry deposition from published data, and obtained a mean ratio of 0.9 (Fig. 1, bottom). Given the scatter of the data, these three estimates of the ratio of dry to wet deposition were indistinguishable from 1, agreeing with Hinga et al. (1991), who concluded that wet and dry deposition were of the same magnitude. Thus, we estimate atmospheric deposition to be twice as much as wet deposition (Table 1).

Nitrogen delivery by fog and cloud droplets captured by vegetation may be a significant additional source of nitrogen near the sea, because marine fogs may bear high nitrogen contents. Waldman et al. (1983) report marine fogs in southern California with concentrations of 821–4500 μmol/L NO_3 and 1123–4059 μmol/L NH_4. There is so little known about coastal fogs as a

TABLE 1. Fate of atmospheric nitrogen deposited on forested areas of the Waquoit Bay watershed. Data are concentrations of dissolved inorganic (DIN), dissolved organic (DON), and total dissolved nitrogen (TDN) in water falling onto surface of watershed, leaving soil, leaving the aquifer, and in receiving river water. Estimates based on tens to hundreds of measurements for each number, from references cited below.

Water source	DIN (μmol/L)	DON (μmol/L)	TDN (μmol/L)	DON (%)
Atmospheric deposition	120†	89	209	43
Leaving soil‡	18	56	74	76
Leaving aquifer§	8	11	19	59
River water‖	8	12	20	60

† Data from Lajtha et al. (1995). Value calculated to allow for dry deposition, as well as account for evapotranspiration of water, as (14 μmol/L NO_3 + 13 μmol/L NH_4) × 2/0.45 = 120 μmol/L DIN. Water deposited on watersheds evaporates and is transpired, so that the flow of precipitation through soil, and hence recharge to groundwater, is only a fraction of precipitation (Thornthwaite and Mather 1957, Running et al. 1988). At the latitude of Cape Cod, where Waquoit Bay is located, for example, evapotranspiration accounts for 55% of the annual precipitation, so that ≈45% of the annual water input percolates through the unsaturated subsoil and recharges the aquifer (Eichner and Cambareri 1992).

‡ Data from Lajtha et al. (1995).

§ Data from I. Valiela et al. (*unpublished manuscript*), for groundwater about to emerge from forest-covered catchment areas.

‖ Data from K. Foreman et al. (*unpublished data*), for water that has just seeped into upper reaches of Quashnet River. Data include only upper reaches, where river water derives from forested catchment areas.

nitrogen source, however, that we omitted this term in our model.

Most loading studies concentrate on dissolved inorganic nitrogen (DIN) and ignore dissolved organic nitrogen (DON) because of the difficulty of analysis, and because DON may be refractory to biological degradation. DON in wet precipitation is quantitatively important in atmospheric deposition (Valiela et al. 1978, Morris 1991), averaging 31% of total dissolved nitrogen (TDN) falling on a wide variety of geographical sites (Cornell et al. 1995). DON is nearly half of the TDN falling onto the Waquoit Bay coastal zone (Table 1, first row).

Little is known about lability of DON in precipitation or in subsurface waters. In the Waquoit Bay watershed, the percent of DON relative to TDN changes as water moves through different components of the subsurface ecosystem, first increasing under soils, then decreasing (Table 1, last three rows). DON is not, therefore, just a passive refractory pool that passes unchanged through coastal landscapes. Changes such as those of Table 1 suggest at least some transformations, so DON may be at least partially labile.

The DIN and DON delivered by wet plus dry atmospheric deposition falls onto different land covers. The fate of the atmospheric nitrogen may vary depending on whether a land parcel is covered by natural vegetation, turf or agricultural land, or impervious surfaces. We therefore assessed the fate of atmospheric nitrogen in each major type of land cover for inclusion in our model, as follows.

1. *Deposition to naturally vegetated parcels.*—For application of the model to the Waquoit Bay watershed we used a 2 yr record of WBLMER data on wet deposition. Total atmospheric deposition to forests, as already discussed, was taken to be twice that of wet deposition.

Atmospheric nitrogen deposited onto vegetated parcels of the Waquoit watershed is partially intercepted by vegetation and soils (Tables 1, 2). Aggrading forests such as those of Cape Cod (Valiela and Costa 1988) take up and store precipitation-borne nitrogen (Vitousek and Reiners 1975, Boring et al. 1988, Aber et al. 1989). We found that 65% of atmospheric nitrogen delivered to coastal forests is retained in trees and soil of the watershed of Waquoit Bay. Retention was calculated based on differences in concentrations of TDN in atmospheric deposition relative to concentrations in water collected by lysimeters installed below the root zone (Table 2). Our estimate of 65% retention of atmospheric nitrogen by coastal forests is lower than the 80–90% retention reported for upland forests of New England (Aber et al. 1993), but similar to the 68% retained in a riparian forest in Georgia (Lowrance et al. 1984). It is difficult, however, to compare different studies because of differences in the kinds of nitrogen included.

2. *Deposition to turf.*—Lack of data forced us to assume that total deposition from the atmosphere onto turf occurred at the same rates as on forest parcels. This assumption overestimates nitrogen delivery, because turf vegetation has lower leaf surface area per unit land surface, and hence is likely to collect less dry deposition than forest vegetation.

We calculated losses of 62% of atmospheric-derived nitrogen in turf plants and soils (Table 2) of the Waquoit Bay watershed by comparing nitrogen concentrations in lysimeters placed under lawns and a grass field (79.6 μmol/L TDN, from Lajtha et al. 1995) with calculated concentrations in atmospheric deposition (209 μmol/L TDN, from Table 2) onto the Waquoit Bay watershed.

3. *Deposition to agricultural land*.—We assumed that atmospheric nitrogen deposited onto agricultural land suffers fates similar to nitrogen deposited on turf, and calculated atmospheric nitrogen loading to agricultural parcels accordingly (Table 2). If crops are consumed outside the watershed, harvest and removal of the crop may be an additional loss of nitrogen. This would be important in areas with well-developed industrialized agriculture, but in suburban landscapes such as the Waquoit Bay watershed most crops are consumed locally.

4. *Deposition to impervious surfaces*.—Deposition of atmospheric nitrogen on roads, parking lots, roofs, etc., is a prominent component of some loading cal-

TABLE 2. Fate of atmospherically derived nitrogen deposited on different land cover types. Data show losses and concentrations of nitrogen as it travels through vegetation and soils, vadose zones, aquifers, and seepage faces on the way to the receiving rivers.

Type of land cover	Surfaces and components of ecosystem	Concentration of TDN (μmol/L)	Percentage of TDN entering component lost within component	Percentage of total TDN loss in each component
Naturally vegetated land	Watershed surface	209†		
	Forests and soils		65	61
	Top of vadose zone	74		
	Vadose zone		61	24
	Water table	29		
	Aquifer		35	5
	Seepage face	19		
	Total		91	100
Cultivated land‡	Watershed surface	209		
	Plants and soils		62	58
	Top of vadose zone	80		
	Vadose zone		61	26
	Water table	31		
	Aquifer		35	6
	Seepage face	20		
	Total		90	100
Roofs and driveways	Watershed surface	188§		
	Plants and soils		62	58
	Top of vadose zone	71		
	Vadose zone		61	25
	Water table	28		
	Aquifer		37	6
	Seepage face	17		
	Total		91	100
Roads, runways, and commercial areas‖	Watershed surface	188		
	Top of vadose zone	188		
	Vadose zone		61	82
	Water table	73		
	Aquifer		37	17
	Seepage face	49		
	Total		74	100

† Concentration that should appear in recharge water, once evapotranspiration has removed 55% of precipitation.
‡ Includes turf and agricultural land. Losses by crop harvest are ignored in this table.
§ Assuming 50% evaporation of precipitation that falls on impervious surfaces.
‖ Atmospheric nitrogen delivered to roads, runways, and commercial areas is not subject to losses in vegetation and soil, because these impervious surfaces are often provided with catch basins that deliver water below the soil.

culations (IEP 1986, U.S. Environmental Protection Agency 1982, Frimpter et al. 1990). The prominent role given to impervious surfaces stems from reports that nitrogen concentrations in runoff from them are roughly twice as large as concentrations in wet deposition (Novotny et al. 1985, Schmidt and Spencer 1986). Increased concentrations do not necessarily represent new sources of nitrogen delivered to impervious surfaces, but likely result from the concentration of dissolved nitrogen by evaporation, and from dry particulate deposition of atmospheric nitrogen.

Nitrogen loading from roofs and roads has been considered separately in the literature, because concentrations of nitrogen in runoff from different types of surfaces appear to differ (average 0.75 mg N/L from roofs and 1.5 mg N/L from roads [IEP 1988]). Such data are difficult to evaluate, since concentrations depend on the amount of rainfall in specific days and should be collected over suitably long times to capture precipitation events of different magnitude. Dry deposition may accumulate during dry weather, so that concentrations are affected by the time lapsed since the last rain.

For lack of better data we assumed that dry deposition was the same for forests and impervious surfaces. We therefore overestimate nitrogen delivery to impervious surfaces, because impervious surfaces have smaller areas of total surface per hectare than forests. Atmospheric nitrogen that falls onto roofs and driveways flows onto adjoining turf, where there are losses of nitrogen (Table 2).

Water falling on impervious surfaces evaporates, and further loss of water occurs by evapotranspiration in turf adjoining impervious surfaces. We estimated from

TABLE 3. Estimates of rates of fertilizer addition to turf (lawns and golf courses).

	Sites and methods	Fertilizer addition rates (kg N·ha^{-1}·yr^{-1})	References
Lawns	Long Island	49	Koppelman (1978)
	Cape Cod	147	Horsley and Witten (1991)
	Cape Cod		
	by homeowners	137	Nelson et al. (1990)
	by professionals	227	
	weighted average	171	
	Mean ± SD	104 ± 64	
Golf courses	Cape Cod	152–181	Frimpter et al. (1988)
	Cape Cod	83–151	Eichner and Cambareri (1988)
	Various sites	49–73	Petrovic (1990)
	Mean	115	

personal observations that water losses there must be at least as great as those in forests, so we assumed a 50% recharge of water. With these assumptions we calculated concentrations of TDN in soil and losses of atmospheric nitrogen falling on impervious surfaces (Table 2).

Atmospheric nitrogen deposited on larger roads, runways, and commercial areas largely flows into gutters and drains, and accumulates in catch basins in the unsaturated vadose subsoil. Thus, we presume that nitrogen falling onto large roads, runways, and commercial areas is not exposed to losses in adjoining turf.

The model calculations show the reduced concentrations of TDN derived from the atmosphere during passage through vegetation and soils, the vadose zone, and the aquifer (Table 2, first column of numbers). About 90% of atmospherically derived nitrogen is intercepted, so that only 10% reaches receiving waters. Soils intercept about two-thirds of the total incoming atmospheric nitrogen; the vadose zone removes about one-fourth, and the aquifer removes a more modest amount (Table 2, third column of numbers).

Fertilizer applications.—Our model treats turf and commercial horticulture application of fertilizers separately, because the processes involved differ. Use of fertilizer for household horticultural purposes is generally minor compared to lawn use, so we ignored fertilization of vegetable and flower gardens.

TABLE 4. Fate of nitrogen added as fertilizer to turf: standing stocks and losses (mean ± 1 SD). Restructured from data from many places, compiled by Petrovic (1990).

	N added as fertilizer (%)	Number of observations
Standing stocks:		
Vegetation, clippings	46.5 ± 16.3	52
Vegetation, roots, crowns	11.5 ± 13.2	13
Vegetation, thatch	23.5 ± 3.5	2
Soil	33.3 ± 30.0	10
Total stocks	104.8	
Losses:		
Leaching	10.3 ± 16.3	67
Gaseous losses	38.5	
Volatilization	15.3 ± 18.0	31
Denitrification	23.2 ± 31.7	9

1. *Turf use*.—Fertilizer nitrogen is used in suburban or semirural watersheds for lawn and golf course maintenance. Our model calculates fertilizer inputs by combining application rates (Table 3), the percentage of households that use fertilizers (34% of households in Cape Cod use fertilizers [Giblin and Gaines 1992; A. Giblin, *personal communication*]), and areas of turf, including lawns and golf courses. Lawns in suburban to semirural coastal areas such as Long Island and Cape Cod average 0.05 ha in area (Koppelman 1978, IEP 1986, Frimpter et al. 1990, Eichner and Cambareri 1992).

Fertilizer nitrogen delivered to turf is stored, transported to groundwater below, or lost in gases. Nitrogen is taken up by plants and soil (Table 4), but net nitrogen storage in plants and soil does not increase after the first decade following establishment of turf parcels (Porter et al. 1980). Our model assumes that, on average, turf parcels in entire watersheds are older than 10 yr. This allows us to assume that there is no net increase in the storage of nitrogen in turf if we consider entire watersheds.

Most calculations of nitrogen loading rates by fertilizers have focused on leaching rates, usually measured within days to months after application of a fertilizer. Such short-term measurements underestimate throughput of fertilizer nitrogen (Table 4), because nitrogen taken up in older turf by plants and soil will eventually be leached to the subsoil. It seems better to consider actual losses of nitrogen as gases. The WBLMER model therefore treats losses of fertilizer nitrogen as taking place via denitrification and volatilization from turf. These gaseous losses account for ≈39% of fertilizer nitrogen (Table 4). We therefore consider that 61% of nitrogen applied as fertilizer reaches the subsoil below turf parcels.

2. *Horticultural use*.—The WBLMER model cal-

culates inputs of horticultural fertilizer as the product of dosage rates and areas of agricultural land use.

Our model uses an average of 136 kg N/ha, an average obtained from data on a variety of crops (Loehr 1974, Stanley 1988, Hayes et al. 1990, Correll et al. 1992). We opted to use an average dosage that represents a variety of crops, because in the suburban to rural areas which comprise most of the Waquoit Bay watershed, farms are small, and crops are mixed. Crops and application rates vary so much geographically that for modeling purposes it is imperative to apply fertilizer use rates appropriate to the region being considered (Loehr 1974, Stanley 1988, Frimpter et al. 1990, Hayes et al. 1990, Correll et al. 1992, Jaworski et al. 1992).

Fertilizer nitrogen taken up by crops may be harvested and removed from the watershed where the crops grew. If crops are grown and consumed locally by people, nitrogen taken up by crops should be ignored, because it is already included in the calculation of wastewater contributions. In the case of the Waquoit Bay watershed, for example, virtually all crops, except cranberries, are consumed locally. Cranberry bogs are limited in area, and are relatively small net sources of nitrogen to the rest of the watershed (13 kg $N \cdot ha^{-1} \cdot yr^{-1}$, Teal and Howes 1995). We therefore ignored the harvest of crops in our calculations for Waquoit Bay.

Animal sources.—The WBLMER model does not include animals in the watershed as sources of nitrogen. With few exceptions, wastes from livestock, domestic animals, or waterfowl do not add significant amounts of nitrogen to receiving waters. Arguments justifying the omission of animals as nitrogen sources are outlined in the paragraphs below.

1. *Livestock*.—Under very intensive husbandry conditions, livestock can be important sources of nutrients to coastal watersheds and receiving waters (Loehr 1975, Stanley 1988, Postma et al. 1991). On Long Island, New York, duck farms whose runoff ran directly into Moriches Bay eutrophied the waters of this coastal lagoon (Ryther 1954). Loading calculations suggested that livestock contribute 61% of the nitrogen entering the Upper Potomac River watershed (Jaworski et al. 1992).

The concept of livestock as a nutrient source is ambiguous, however. Animals on pastures or fed on locally grown silage or hay are not new sources of nitrogen to the watershed, because the nitrogen in their feces derives from the atmosphere or from fertilizers. In our model, nitrogen passing through livestock has already been accounted for as atmospheric or as agricultural fertilizer nitrogen.

Contributions from livestock should be included only if the animals were fed on nitrogen brought in from outside the watershed, as would be the case in feedlots or intensive poultry farms. In these circumstances, livestock are fed midwestern corn or meal from fish caught in the sea, and their manure constitutes additional nitrogen. In these cases, livestock contribution to nitrogen loading should be calculated (Stanley 1988, Jaworski et al. 1992). Different species of livestock may be more or less abundant in one area or another, and so may contribute different amounts of nitrogen (Table 5). The calculation requires a census of total livestock on the watershed, and these numbers are then multiplied by species-specific rates of nitrogen release. The Waquoit Bay region lacks such industralized husbandry.

TABLE 5. Estimates of annual nitrogen release per individual for people and livestock (data from Watson et al. 1967, Brandes 1978, Siegrist et al. 1976, and North Carolina Department of Natural Resources and Community Development 1986).

	Annual release of nitrogen ($kg \cdot individual^{-1} \cdot yr^{-1}$)
People	4.8†
Dairy cattle	108
Beef cattle	51
Horses	60
Swine	10
Sheep	6.6
Turkeys	3.3
Layer poultry	0.7
Broiler poultry	0.3

† Mean value; SD = 2.3; SE = 0.9.

2. *Domestic animals*.—Nitrogen loading estimates in suburban watersheds of Long Island, New York include nitrogen from domestic animals (Koppelman 1978). Nitrogen loadings of 2 $kg \cdot dog^{-1} \cdot yr^{-1}$ and 1.5 $kg \cdot cat^{-1} \cdot yr^{-1}$, and ratios of eight people per dog and 12 people per cat, were multiplied by the number of people on the watershed. Since ≈50% of fecal nitrogen is lost by volatilization (Porter et al. 1980), contributions by dogs and cats amounted, at most, to 2.6 and 1.6% of the nitrogen loadings attributed to people. Unless a watershed contains an inordinately dense pet population, input from domestic animals is unimportant relative to human input, and should be ignored in nutrient loading calculations.

3. *Waterfowl*.—There is much popular discussion on nitrogen contributions by waterfowl to coastal water bodies, inspired perhaps by accounts of large release of nitrogenous wastes in rookeries (Mizutani and Wada 1988, for example). These accounts refer to breeding colonies of birds such as gulls, terns, and penguins, in which the adults forage widely to collect food for young nestlings and defecate near the colony. Such situations are rarely encountered in the kinds of watersheds we are discussing, where nitrogen loading from waterfowl can be ignored for three reasons. First, for most watersheds, the amount of nitrogen that passes through waterfowl is minute compared to new nitrogen contributed by other sources. In Buttermilk Bay, Massachusetts, overwintering waterfowl were abundant, yet the total amount of nitrogen passing through the birds amounted to only 0.2% of total inputs (Valiela and Costa 1988). Second, many species of waterfowl

(ducks in particular) feed on material produced within the aquatic system, so they in effect recycle nitrogen within the water body rather than constitute a new, external source. Third, species of waterfowl that forage on land, such as geese (Buchsbaum et al. 1986), defecate so frequently (on average every 30 min in geese [Owen 1975]) that when they return to the water body to roost they bring little new nitrogen with them to the water. For these reasons it seems unnecessary to include waterfowl as a source of nitrogen in loading calculations to water bodies that host any but the most extraordinarily high waterfowl densities.

Fate of nitrogen in vadose zones and aquifers

In the Waquoit Bay and similar watersheds, nitrogen is inserted below the watershed surface in two major ways. Nitrogen originally delivered to the watershed surface by atmospheric deposition and by fertilizer use percolates to the unsaturated vadose zone in a nonpoint, diffuse fashion. Nitrogen from septic systems, the other major nitrogen input, is delivered directly to the unsaturated zone in distinct plumes emanating from septic system leaching fields. Both diffuse- and plume-conveyed nitrogen then cross the water table and travel through the aquifer, where further losses may take place. Our model incorporates these entries and losses in the vadose zone and aquifer.

Transport and losses of surface-delivered nitrogen.—The fate of nitrogen during travel through the vadose zone is not well known (Keeney 1986, Korom 1992), although losses of nitrogen in unsaturated vadose sands under agricultural fields have been reported (Cameron and Wild 1982, Starr and Gillham 1993). About 61% of the nitrogen that reached the vadose zone of forests in the Waquoit Bay watershed was lost in the unsaturated sediments (Table 2). Our model therefore allows 39% of atmospheric nitrogen that percolated from forests to go through the vadose zone (Table 2). We assumed that these values are applicable to other land cover types and repeated the calculation for the remaining land uses.

We have no data about losses of nitrogen in the Waquoit aquifer. Although denitrification may be active in wastewater plumes travelling through aquifers, there is no convincing information in the literature on losses of nitrogen travelling in diffuse form in aquifers. This is one of the largest unknowns in the whole topic of nitrogen loading. Different opinions can be found in the literature.

It has been argued that nitrogen losses within aquifers should be negligible because of the lack of anoxic conditions and of insufficient organic matter to provide energy for denitrifier bacteria (Keeney 1986, Postma et al. 1991, Starr and Gillham 1993). Sulfide- or pyrite-based denitrification may be significant in sediments of marine or lacustrine origin (Trudell et al. 1986, Böttcher et al. 1990, Postma et al. 1991), but this pathway is unlikely to be important in sandy sediments of glacial origin, such as those of Cape Cod.

On the other hand, several lines of evidence suggest that denitrification or other losses must occur within aquifers. First, nitrate and dissolved organic matter decrease downgradient in a coastal aquifer in Maryland (McFarland 1989), in sands in Ontario (Trudell et al. 1986, Gillham 1991), and in a dolomite aquifer in Wisconsin (Cherkauer et al. 1992). Second, mass balance calculations suggest a loss of 20–35% of the nitrogen during travel in the aquifer leading to Buttermilk Bay (Valiela and Costa 1988, Weiskel and Howes 1991), and 62% to Lake Michigan (Cherkauer et al. 1992). Third, denitrification was measurable in an Ontario sandy aquifer in which the water table was sufficiently shallow for organic matter from soil to reach the aquifer (Starr and Gillham 1993).

Although we suspect that losses of diffusely transported nitrogen in aquifers occur, there is insufficient published information with which to estimate the magnitude of losses (Korom 1992). To fill the gap, we estimated nitrogen loss in the Waquoit Bay aquifer as the difference between nitrogen concentrations in groundwater near the water table, and in groundwater about to enter receiving estuaries (I. Valiela et al., *unpublished data*). This estimate is based on data from samples of groundwater that was under forested land parcels. The rough estimate is that 35% of diffuse nitrogen entering the aquifer under the Waquoit Bay watershed may be lost within the aquifer (Table 2). We emphasize that our rough guess needs to be replaced by direct measurements of nitrogen losses in aquifers.

Our estimate of the total loss of diffuse nitrogen in the aquifer is a steady-state simplification: nitrogen losses during travel in aquifers are likely to be distance- or travel time-dependent. G. N. Collins et al., (*unpublished manuscript*) compared steady-state loading estimates to Waquoit Bay with similar model calculations in which allowances for loss during travel time were made. Difference between the two were ≈10%, too small to alter our interpretation here. In watersheds with much larger areas and longer travel times than those in Waquoit Bay, the differences may be more marked (G. N. Collins et al., *unpublished manuscript*). Until in situ rates of denitrification in groundwater are available, we have to use our steady-state approach, and our model therefore allows 65% of the entering nitrogen to traverse aquifers, regardless of travel distances or time in transit (Table 2).

Our model provides the best, albeit rough, estimates based on the available literature and our own data. It should be clear, however, that rates of denitrification and other losses in vadose zones and within aquifers, are probably some of the most important, yet poorly defined, terms in loading calculations. High priority needs to be given to actual measurements of rates of loss of nitrogen in aquifers and unsaturated sediments.

Delivery and losses of wastewater.—Unlike atmo-

TABLE 6. Average concentrations (± SE) and percentage losses of wastewater nitrogen during passage through components of septic systems, plumes, and aquifers.

Steps along path of wastewater flow	Components	Total dissolved nitrogen concentration (mg/L)	Percentage loss of TDN that enters each component	Percentage of total loss of TDN that occurs in each component
Raw wastewater entering septic tanks		72 ± 12		
	Septic tanks		6	6.4
Effluent leaving septic tanks		68 ± 7		
	Leaching fields		35	38.6
Effluent leaving leaching fields		44		
	Plumes		34	46.6
Effluent leaving plumes		15		
	Aquifer		35	8.4
Wastewater nitrogen leaving aquifer		9.8		
	Total loss relative to raw wastewater nitrogen inputs		86	100

Sources: Values are compiled from Watson et al. (1967), Polkowski and Boyle (1970), Walker et al. (1973*a, b*), Hall (1975), Viraraghavan and Warnoch (1976), Brandes (1977, 1978), Gibbs (1977), Andreoli et al. (1979, 1989), Starr and Sawhney (1980), Whelan and Titamnis (1982), Brown et al. (1984), Whelan and Barrow (1984), Cogger et al. (1988), Close (1989), Robertson et al. (1991), Robertson and Cherry (1992), and references cited in Fig. 4; other values are calculated.

spheric deposition and fertilizer use, disposal of residential wastewater by septic systems inserts nitrogen directly below the subsoil and into the vadose zone. Moreover, nitrogen from wastewater moves into the aquifer in relatively well-defined tubular plumes. In contrast to the spatially diffuse delivery of nitrogen from the atmosphere and fertilizers, septic systems in a watershed are more like a network of point sources. Such differences require that our model calculations of nitrogen loading from septic systems be treated in a different manner (Table 6).

The septic systems we consider in this review are mainly on-site wastewater disposal systems of conventional design (Kaplan 1991), where raw wastewater first enters a septic holding tank in which sedimentation and microbial degradation take place. Wastewater effluent then overflows from the septic tank to a leaching field or to a leaching tank, from which the effluent disperses into surrounding nonsaturated soils and sediments.

The solutes from septic leaching fields then flow downgradient in plumes—pencil- or snake-shaped features—revealed by high concentrations of nutrients and organic matter (Robertson et al. 1991, Robertson and Cherry 1992). Septic system plumes are much longer than wide, and maintain their shape for some distance. Robertson et al. (1991) and Robertson and Cherry (1992) show the best described sewage effluent plumes in sandy aquifers. From data on solute concentrations, Robertson et al. (1991) show that septic system plumes may extend to 130 m, and probably farther. At some unknown distance from the source, dispersion must broaden the shape of plumes so that eventually the plume loses its pencil shape. The solutes in the plume should continue to travel downgradient, dispersed in the surrounding groundwater.

To capture the fate of wastewater nitrogen in our model, we incorporated a description of nitrogen inputs to septic systems, losses in septic tanks and leaching fields, and losses of nitrogen in plumes and beyond in aquifers. The inputs minus the cumulative losses in these three compartments are our best estimate of loading from wastewater. All relationships used in the wastewater part of the model came from surveys of literature data.

1. *Nitrogen inputs and losses in septic systems*.—Calculations of nitrogen inputs to and from septic systems have been done by diverse methods which fall into two basic approaches, the ''per capita'' and ''water use'' methods. These two alternatives are compared in I. Valiela et al. (*unpublished manuscript*). Here we use the per capita method, which calculates nitrogen delivery via septic tanks as per capita release of N per year × average number of people per household × number of houses in the watershed (Koppelman 1982, Valiela and Costa 1988).

On average, 4.8 kg nitrogen is released per person per year, with a range of 1.8–5.4 kg (Table 3). These values are similar to those found by others (1.8–7.4 kg $N \cdot person^{-1} \cdot yr^{-1}$ [Vollenweider 1987; Koppelman 1978, U.S. Environmental Protection Agency 1980; J. E. Costa et al., *unpublished manuscript*]).

The total number of residences or buildings within a watershed can be estimated from aerial photos, from Geographic Information System (GIS) data (Lindhult and Godfrey 1988, Sham et al. 1995) or from municipal records. The number of people per house can be obtained from municipal records, from census data, or by using data from other areas of similar ecological setting and socioeconomic background (Valiela and Costa 1988, Hayes et al. 1990). House occupancy ranges from 1.8 to 3.0 people per house in rural to suburban coastal areas of the U.S. Data obtained from several areas of the U.S. before the mid-1960s showed an average of 3.8 people per house, with a range of 3.0–4.7 (Cotteral and Norris 1969). Nitrogen loading calculations for Long Island, New York used 3.0 people per house (Koppelman 1978), while 2.8 was used in Texas (Hayes et al. 1990), and 2.7 in Cape Cod (Nelson et al. 1988, Valiela and Costa 1988, Frimpter et al. 1990, Costa et al. 1992, Eichner and Cambareri 1992).

For our Waquoit Bay calculations, we calculated occupancy for our area using land parcel data put into a GIS, and aerial photos. In Cape Cod, many houses are occupied only during summer, vacations, or on weekends. In addition, Cape Cod is home to many retired people who spend the winter in warmer climates. To compensate for the seasonal occupancy, we obtained 1990 census data on the duration of occupancy of individual houses within the Waquoit Bay watershed. From these data we calculated a weighted mean (± 1 SD) of 1.8 ± 0.6 people per house. This value is similar to the seasonally adjusted value of 1.9 people per house suggested for an area where 23% of the houses were used only in summer (Weiskel and Howes 1991).

Some estimates of nitrogen loading from septic systems have included commercial, industrial, and other buildings (U.S. Environmental Protection Agency 1980, Frimpter et al. 1990, Eichner and Cambareri 1992). In rural or small town communities where most people work and live within the same watershed, the addition of inputs from nonresidential buildings may not be appropriate. Adding nitrogen contributions from people that occupy those buildings during the work day is double accounting, since they have already been included as residents of houses in the per capita method.

The next step in the per capita procedure is to multiply the amount of nitrogen delivered to septic tanks by a value that describes losses of nitrogen (by denitrification, volatilization of ammonia, or by adsorption of ammonium) within septic systems. Since such losses of nitrogen have not been well established, the default assumption has been to say that 50% of the per capita nitrogen load was lost in septic systems (Koppelman 1978, Frimpter et al. 1990, Costa et al. 1992). To see if the 50% loss term was appropriate, we used published data to calculate losses of the nitrogen in septic tanks and in leaching fields.

To ascertain if there were losses of nitrogen within septic tanks, we compared published data measuring nitrogen concentrations in wastewater entering septic tanks, to data on nitrogen contents in effluent leaving septic tanks (Table 6). Concentrations of TDN in water entering septic tanks were ≈5% higher than those in effluent leaving septic tanks (Table 6), somewhat smaller than the 10–20% suggested by others (Andreoli et al. 1979; J. E. Costa et al., *unpublished manuscript*). Some septic holding tanks are pumped out from time to time, but this removal only amounts on average to 4% of nitrogen entering septic systems (Kaplan 1991), and the septage is often released in septage lagoons within the watersheds.

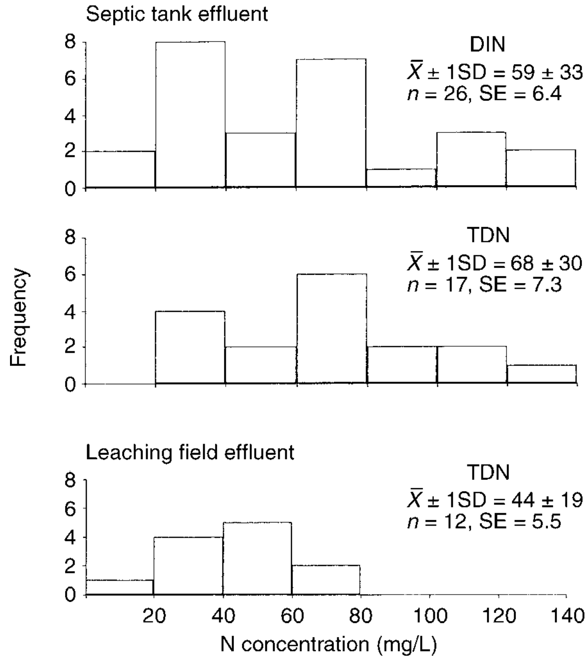

FIG. 2. Frequency distributions of concentrations of DIN (top), of TDN in effluent leaving septic holding tanks (middle), and of TDN in effluent leaving septic leaching fields (bottom). Means ± SE are shown on top right of each histogram. Data for the top graph are from Polkowski and Boyle (1970), Walker et al. (1973*a*, *b*), Viraraghavan and Warnock (1976), Brandes (1977), Gibbs (1977), Whelan and Titamnis (1982), Brown et al. (1984), Whelan and Barrow (1984), Cogger et al. (1988), Robertson et al. (1991), and Robertson and Cherry (1992). Data for the middle graph are from Walker et al. (1973*a*, *b*), Gibbs (1977), Starr and Sawhney (1980), Whelan and Titamnis (1982), Brown et al. (1984), and Alhajjar et al. (1989). Data for the bottom graph are from Walker et al. (1973*a*, *b*), Reneau (1979), Starr and Sawhney (1980), Alhajjar et al. (1989), and Robertson et al. (1991).

To assess losses of nitrogen within leaching fields, we compared reported total dissolved nitrogen concentrations in effluent from septic holding tanks (Fig. 2, middle) vs. concentrations measured in effluent leaving leaching fields (Fig. 2, bottom). Data on concentrations of solutes in effluent came from various studies, and were collected at a variety of distances away from leaching fields. We extrapolated measured concentrations at given distances to concentrations that should have been found near leaching fields. Linear regressions of log-transformed data were used to calculate expected concentrations at 0.1 m away from the leaching field, a distance where chloride concentrations showed minimal dilution.

We used two different ways to estimate retention of

TABLE 7. Concentrations (mean ± SE) of chloride in wastewater leaving septic holding tanks and leaching fields. Data are from Polkowski and Boyle (1970), Walker et al. (1973*a*, *b*), Brandes (1977, 1978), Cogger et al. (1988), Alhajjar et al. (1989), Robertson et al. (1991), and Robertson and Cherry (1992).

Source	Chloride (mg/L)
Effluent leaving septic tanks	75 ± 33
Effluent leaving leaching fields	90 ± 68

nitrogen in leaching fields. First, we calculated retention of nitrogen in septic system leaching fields as the difference in the concentration of nitrogen in septic tank effluent relative to the concentration in effluent from leaching fields. These figures came from all available data on the concentrations in effluent from septic tanks and leaching fields. Mean concentrations of nitrogen in wastewater leaving septic tanks (Fig. 2, middle) were 35% greater than mean concentrations in effluent leaving leaching fields (Fig. 2, bottom). The 35% loss could be a true loss caused by denitrification, or merely a dilution effect. There was no obvious dilution in leaching fields, because chloride concentrations in leaching effluent reported in the studies did not significantly decrease relative to chloride or sodium in septic holding tanks (Table 7). Actual losses of ≈35% of the nitrogen must therefore take place within leaching fields (Table 6).

We obtained the second estimate of nitrogen retention using data from papers (see legend for Fig. 3) that provided concentrations of nitrogen in both inputs to holding tanks and outputs from leaching fields of the same septic systems. For each system we calculated ''retention efficiency'' as nitrogen inputs (given by concentrations in holding tanks) divided by outputs (concentrations at 0.1 m from leaching field). System-specific nitrogen retention efficiencies were remarkably variable, ranging from 10 to 90%, with an average of 46% retention (Fig. 3).

We averaged the 35% loss in leaching fields obtained from the first estimate from all available data, and the 46% from the system-specific nitrogen retention, to obtain 40% as our best estimate of nitrogen losses in septic systems of conventional design. Hence, ≈ 60% of wastewater nitrogen is likely to travel beyond leaching fields (Table 6). Given the variation in the data, our estimate of a 40% nitrogen loss within septic systems is probably not different from the 50% loss assumed by practitioners of the per capita method.

There is substantial variation associated with retention estimates for septic systems of conventional design (Fig. 3). One source of variability may be differences in the rate of wastewater flow (Kaplan 1991). We examined the effect of water use on nitrogen retention using data from some of the septic system papers from which we calculated nitrogen retention (Fig. 4, top). Nitrogen retention was lower in septic systems with faster flow, so differences in water use do contribute

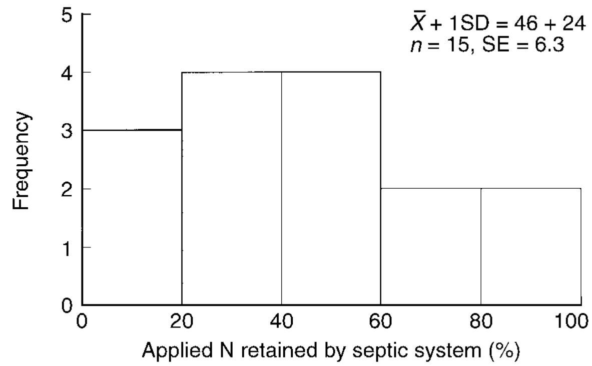

FIG. 3. Nitrogen retention efficiencies (concentration of TDN at 0.1 m from leaching field/concentration of septic tank effluent × 100) in septic systems of conventional design. Data are from Walker et al. (1973*a*), Reneau (1979), Starr and Sawhney (1980), Lamb et al. (1987), Cogger et al. (1988), Alhajjar et al. (1989), Robertson et al. (1991), and Robertson and Cherry (1992).

to the large variation in nitrogen retention found among conventional systems.

Another source of variation in the retention of nitrogen may be the age of the systems—septic systems do fail over time. Some extensive surveys show that 97% of the systems survived after 10 yr (Clayton 1975). On the other hand, Cotteral and Norris (1969) show that the best survival rate was 70% after 12 yr, but many septic systems failed after 12 yr. Satisfactory

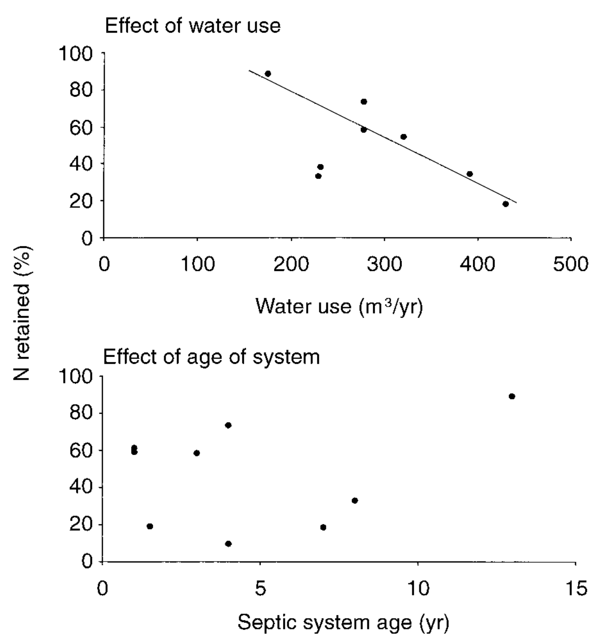

FIG. 4. Percent retention of nitrogen (concentration of TDN at 0.1 m from leaching field/concentration of septic holding tank effluent × 100) in relation to household water use (top graph; data from Walker et al. 1973*a*, Reneau 1979, and Starr and Sawhney 1980), and to septic system age (bottom graph; data from Walker et al. 1973*a*, Starr and Sawhney 1980, Alhajjar et al. 1989, and Robertson et al. 1991).

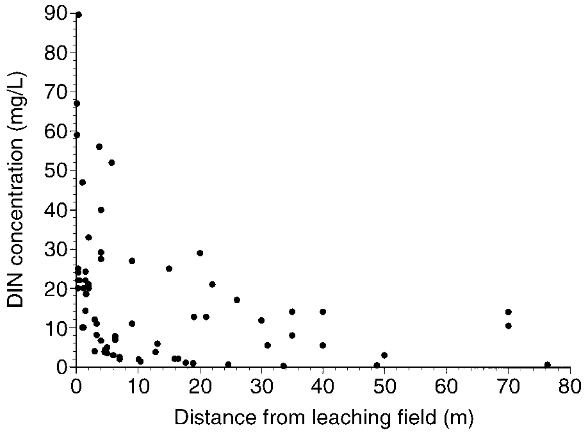

FIG. 5. Dissolved inorganic nitrogen concentrations in water collected at various distances from leaching fields. Data are from Walker et al. (1973*b*), Reneau (1979), Starr and Sawhney (1980), Alhajjar et al. (1989), Gold et al. (1990), and Robertson et al. (1991).

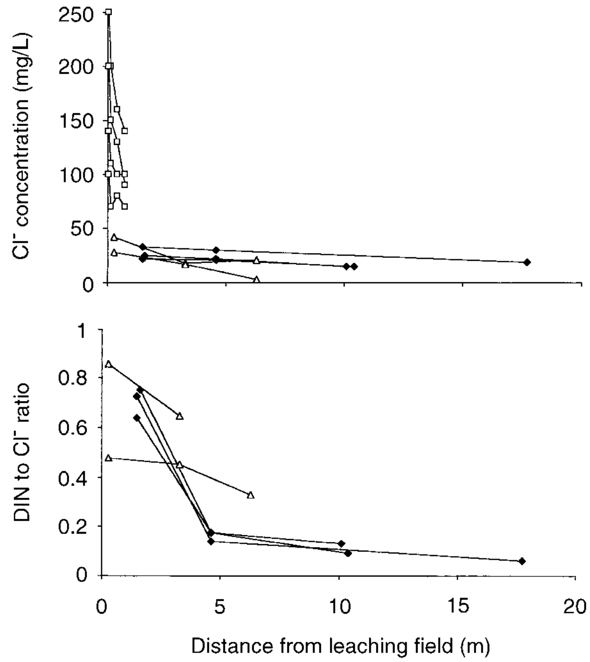

FIG. 6. Chloride concentrations (top) and nitrogen to chloride ratios (bottom) in water sampled at different distances from leaching fields. Data are from Walker et al. (1973*b*) (□), Reneau (1979) (◇), and Alhajjar et al. (1989) (△).

use of septic systems was less than 10 yr in California (Cotteral and Norris 1969). We, however, found no significant relation of retention of nitrogen and age in data from septic systems of 1–15 yr old (Fig. 4, bottom). Within this span of time, factors other than system age are apparently more important in determining nitrogen retention.

In rural to suburban watersheds, some buildings may have cesspools. In the Waquoit Bay watershed, for example, 9% of houses still have cesspools. Cesspools are a traditional, but now largely illegal way to dispose of domestic wastewater. Cesspools consist of a simple holding receptacle from which effluent can flow directly into the subsoil. Cesspools lack a leaching field, the component of a septic system that is most effective at reducing the nitrogen content of effluent. We calculated nitrogen loading from cesspools as in the case of septic systems, except that we omitted losses in leaching fields.

2. *Nitrogen losses away from leaching fields and in plumes*.—Additional losses of nitrogen occur as effluent moves away from leaching fields in the vadose zone and aquifer (Gibbs 1977, Andreoli et al. 1979, Nelson et al. 1988, Robertson et al. 1991, Robertson and Cherry 1992). Such losses are likely to be caused by denitrification. Adsorption of ammonium to sediment particles could take place, but is more likely to occur near septic tanks, before nitrification lowers ammonium concentrations (Walker et al. 1973*a*, 1973*b*, Andreoli et al. 1979, Robertson and Cherry 1992).

There are few quantitative descriptions of nitrogen losses in septic system plumes. To assess nitrogen loss in plumes, we compiled published analyses of nitrogen and chloride or sodium in samples of water taken near and away from leaching fields (Figs. 5 and 6, top). When possible, we used data that followed the core of the plume emanating from a septic system. In unsaturated zones there was most often a vertical downward path, which turned in the direction of regional groundwater flow when the plume reached the saturated zone. Values for distances were obtained by measuring distances along the plume center in published figures, or from distances provided by the authors.

Concentrations of dissolved inorganic nitrogen decreased with distance away from leaching fields (Fig. 5). Such decreases could be a true loss caused by denitrification, or an apparent loss caused by dispersion. We examined these two possibilities by comparing decreases in concentrations of DIN with decreases of chloride or sodium in samples where both were reported. Chloride concentrations decreased over distance (Fig. 6, top), so dispersive mixing did take place. Actual losses of nitrogen also took place, since a plot of DIN/chloride vs. distance also showed a decrease over distance (Fig. 6, bottom).

A plot of DIN vs. chloride concentrations showed that losses of DIN exceeded those of chloride, since the majority of points were below the 1:1 line (Fig. 7, top). To better depict actual loss of DIN, we plotted percent loss of DIN (corrected for dispersion by subtracting the percentage of loss of chloride or sodium) vs. distance away from the leaching field (Fig. 7, bottom). Dispersion-corrected loss of DIN increased with distance in most septic systems (Fig. 7, bottom), and reached some asymptote. Loss of nitrogen by denitrification may take place as long as there is sufficient

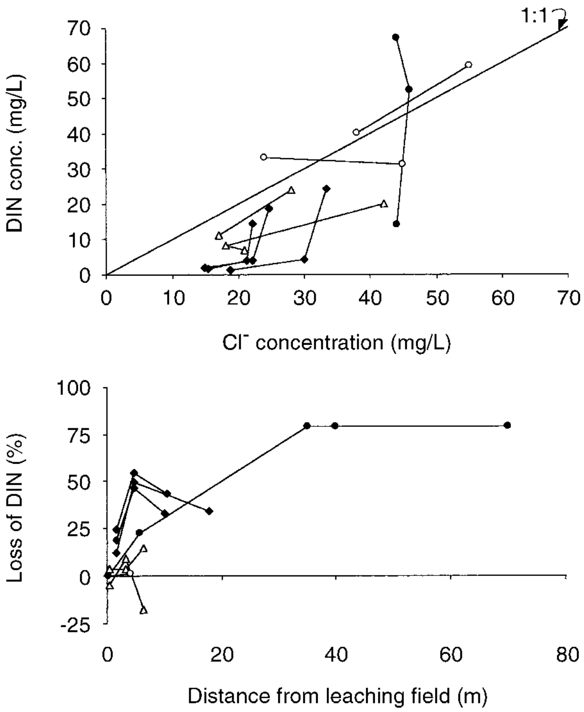

FIG. 7. Top: Nitrogen vs. chloride concentrations in leaching field effluent moving in plumes. The line shows expected relationship if the changes were 1:1. Bottom: percentage of nitrogen losses in leaching field effluent in excess of dilution. Losses were calculated as percentage reduction in initial DIN (estimated at 0.1 m away from leaching field) minus percentage reduction in initial Cl (also estimated at 0.1 m). Data are from Reneau (1979) (◇), and Alhajjar et al. (1989) (△); and from Robertson et al. (1991) and Robertson and Cherry (1992) (○).

organic matter within the plume to sustain denitrifier activity; beyond that distance, nitrogen concentrations may continue unchanged downgradient. The relative amounts of dissolved organic carbon (DOC) and DIN supplied by each septic system vary. The distance at which the asymptotic concentration of DIN occurs may be determined by the supply of DOC in the plume. We accepted the available data (Fig. 6, bottom) as representative, and calculated a mean value from the whole data set. The resulting mean value of septic nitrogen loss during travel in plumes was 34% (Table 6).

Sample collection for the above set of septic systems extended only to about 70 m. In fact, the distances to asymptotic concentrations of nitrogen were less than 50 m (Fig. 6, bottom). As mentioned earlier, it is likely that plumes are longer than 130 m. We guess that 200 m might be a fair estimate of the length of plumes from conventional septic systems. Based on the above analysis of the scant available data, our model assumes that 34% of the nitrogen entering a plume will be lost during the first 200 m. If plumes disperse and lose their integrity beyond 200 m, wastewater nitrogen that is still present in groundwater beyond a distance of 200 m from the leaching field travels in a dispersed, diffuse

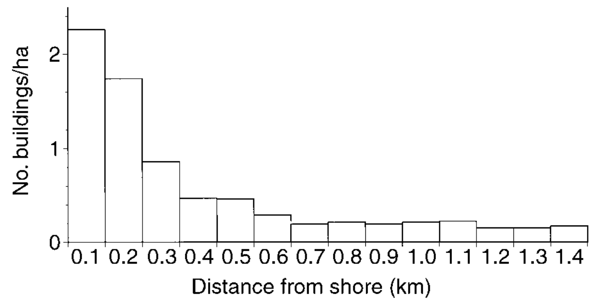

FIG. 8. Number of houses at different distances from the shore of Waquoit Bay estuaries. The distances are in intervals of 100 m, which is approximately the distance that groundwater travels in one year in the Waquoit Bay area.

fashion, analogous to nitrogen derived from nonpoint sources. Our model therefore calculates that beyond 200 m, the remaining wastewater nitrogen in the aquifer is subject to an additional 35% loss (Table 2), the loss we earlier calculated for atmospheric nitrogen in aquifers (Table 2).

One important implication, if our reasoning is correct, is that septic systems closer to shore than 200 m are likely to make significantly greater contributions to nitrogen loading of estuaries than septic systems located farther away. This suggests that a distance-tiered system of requirements for treatment of domestic sewage could be devised, such that septic systems farther than 200 m (or some such limit) could be subject to less strict regulatory restrictions, with no decrease in protection of water quality. In the Waquoit Bay watershed, for example, even though more buildings are built near the shore, only 29% of the septic systems are located within the 200 m band (Fig. 8).

The assumptions in our model about the fates of wastewater nitrogen in plumes and aquifers certainly need verification. If we had data to measure the attenuation of nitrogen within plumes and in diffuse travel in the aquifer we would also be able to define how in situ wastewater disposal at different distances from shore is likely to affect loading to estuaries. Such knowledge would help define how to best, and most fairly, implement management for control of nitrogen loading.

The location of buildings may affect loading calculations in other ways than plume length. First, because of the time required for travel within the aquifer, certain septic systems may not yet have contributed to loading. Nitrogen from houses at long distances away from shore (Fig. 8) may not have had time to reach receiving waters, if the time since a building was constructed is shorter than the travel time for groundwater to reach the shore. On Cape Cod a horizontal distance of 50–200 m downgradient along the aquifer is approximately equivalent to one year of travel (LeBlanc 1984, Sham et al. 1995; G. N. Collins et al., *unpublished manuscript*). Plumes from houses less than 1 yr old and farther than 200 m from shore are unlikely to

TABLE 8. Nitrogen retention in several versions of two general types of nonconventional septic system design.

Percentage of nitrogen retained	Reference
Septic tank effluent passes through sand and gravel filters of various design	
32	Magdoff et al. 1974
44	Harkin et al. 1979
35	Andredakis 1987
70	Piluk and Hao 1989
36	Andreoli et al. 1979
Addition of organic supplements to septic effluent	
80	Laak et al. 1981
37–47	Yahata 1981
27–47	Dillaha et al. 1985
89.3	Rock et al. 1991

have reached the shore. Eventually, these distant buildings will add nitrogen to the estuaries. The issue of travel time and its broader application to projections of past and future loading (and the simulation of differing rates of urbanization) are dealt with in companion papers (Sham et al. 1995; G. N. Collins et al., *unpublished manuscript*).

In general, our data and model calculations suggest that concentrations of wastewater nitrogen decrease during transport toward the shore (Table 6, first column of numbers). Losses of nitrogen are relatively minor in septic holding tanks, but roughly a third of nitrogen entering leaching fields, plumes, and the aquifer is lost in each compartment (Table 6, second column of numbers). In terms of total contribution to losses of wastewater nitrogen (Table 6, third column of numbers), losses in leaching fields and plumes are by far the most important. These results highlight that management of leaching fields and their effluent may be the most practical way to increase interception of wastewater nitrogen.

Knowing the amount of nitrogen lost within the different components of septic systems has important management implications. Since most of the loss occurs in and beyond the leaching field, specific designs of septic tanks may be less important than the locations of leaching fields. The fact that some conventional septic systems achieved a 90% nitrogen retention (Fig. 3) means that high retention is possible, even in systems of standard design. Study of such high performance systems might reveal how retention efficiency may be increased. If indeed the ratio of DOC to DIN determines rates of denitrification, as we suggest above, the addition of labile carbon sources to the leaching field, or beyond, offers a key to the improvement of nitrogen interception.

Our discussion has focused on septic systems of conventional design. The large variation in retention efficiency of conventional designs (Fig. 3) raises the issue of how to evaluate other, perhaps better, septic system designs (Table 8). Proposed improved septic system designs also show remarkably variable retention. As in

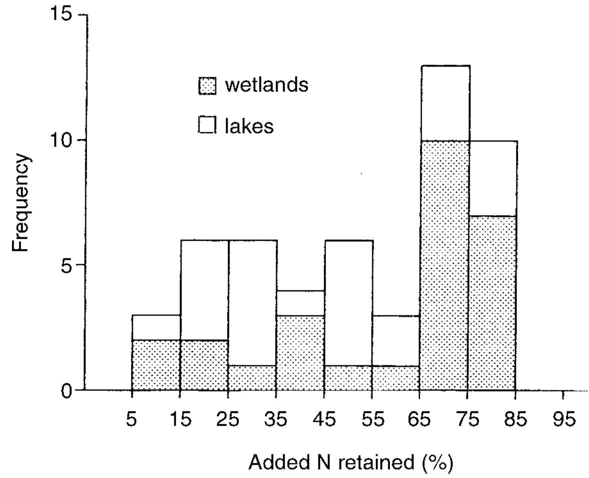

FIG. 9. Frequency distribution of values of percentage retention of nitrogen entering freshwater ponds and lakes, and freshwater wetlands. Data were obtained from Billen et al. (1985, 1991), Johnston et al. (1990), Johnston (1991), and Molot and Dillon (1993).

the case of conventional septic design, waterflow affects performance of the so-called improved designs (Rock et al. 1991). Given the magnitude of variation in performance within designs, in some circumstances installation of a second conventional leaching field or septic system may be a more effective way to assure nitrogen retention. If, for example, it was desired to maintain a 60% nitrogen retention standard in a watershed, any building with projected water use larger than 300 m^3/yr (Fig. 4, top) could be required to install a second separate leaching field or septic system.

Nitrogen inputs and throughputs in freshwater bodies.—Freshwater bodies within coastal watersheds capture groundwater flow, and the amount of nitrogen that traverses ponds, lakes, and wetlands and moves downgradient is less than the nitrogen that enters them. Mass balance studies show retention of 14–100% of nitrogen entering lakes, ponds, and freshwater wetlands (Fig. 9). Wetlands retain a median of 77% of the nitrogen that they receive, while median retention of nitrogen in ponds and lakes is 56% (Fig. 9). These studies underestimated retention because they did not include inputs by dry precipitation.

There are four major ponds within the Waquoit Bay watershed (Fig. 10). The ponds are shallower (McCann 1969) than the aquifer (Cambareri et al. 1992, Eichner and Cambareri 1992), so it is possible for groundwater to flow under the ponds (Cambareri et al. 1992, Dearborn et al. 1994). To assess the possible underflow, we did three-dimensional simulations with a hydrological particle flow model (MODPATH) in which we followed the paths of ''particles'' of water from the shore of Waquoit Bay (J. W. Brawley et al., *unpublished manuscript*). We plotted these paths on a map so we had a plan view of them, and could see which particles crossed under the ponds and which were intercepted

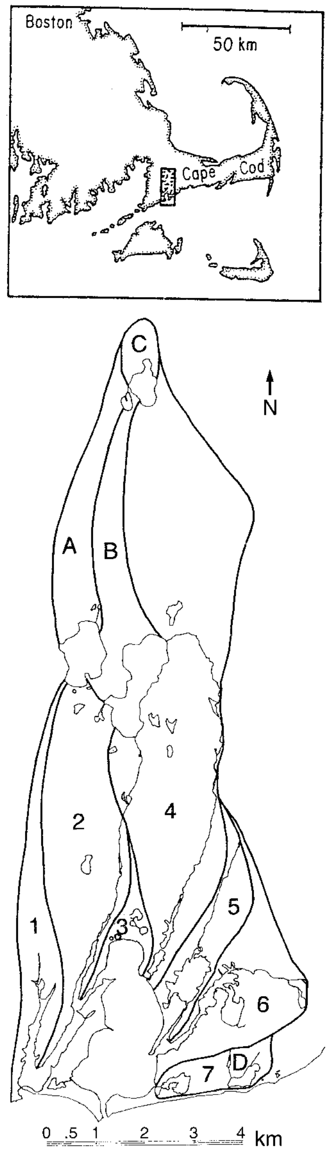

FIG. 10. The watershed of Waquoit Bay, with delineated subwatersheds indicated by numbers in cases where groundwater discharges directly to estuaries, and by letters where discharge is to freshwater ponds. Ponds in turn discharge into aquifers downgradient. Subwatersheds: (1) Eel Pond, (2) Childs River, (3) Head of the Bay, (4) Quashnet River, (5) Hamblin Pond, (6) Jehu Pond, (7) Sage Lot Pond, (A) Ashumet Pond, (B) John's Pond, (C) Snake Pond, (D) Flat Pond.

by the ponds. Only 11.3, 9.5, and 5.6% of the water particles flowed under John's, Ashumet, and Snake Ponds, respectively. This simulation suggests that the major ponds in the Waquoit Bay watershed might intercept most of the groundwater flow. We explored the possible effect of underflow on nitrogen loading downgradient by calculating throughput of nitrogen with the assumption that 25% of the recharge flows beneath the ponds. The 25% underflow was chosen as an overestimate. The simulation showed that with this underflow, an additional 280 kg N/yr moved downgradient, and total nitrogen passing downgradient was 9% of the nitrogen delivered to the watersheds upgradient from the ponds. Effects of pond underflow on nitrogen loading thus seem likely to be modest, even when we overestimated underflow. From these considerations we drew the subwatersheds as shown in Fig. 10, and assumed in the model that the ponds capture all the groundwater flow.

Our model therefore first calculates nitrogen loading for the subwatershed draining into a freshwater body, then adds atmospheric deposition onto the waterbody surface, and subsequently allow 23 or 44% (depending on whether the body is a wetland or a pond) of entering nitrogen to pass through to groundwater downgradient. Lastly, the surviving nitrogen is subject to a further 35% loss in the downgradient aquifer.

Inputs of nitrogen delivered upgradient from larger ponds may be of minor importance to the loading of estuarine waters. The delivery of nitrogen to subwatersheds of the four ponds amounts to 27 904 kg N/yr. Only 2804 kg N/yr is transported from the ponds to the downgradient aquifers. After the further loss of 35% of the nitrogen within the aquifer, only 1767 kg N/yr arrives at the edge of the estuaries. This means that only ≈8% of the nitrogen delivered to the watersheds of the ponds reaches estuaries of Waquoit Bay. The finding that passage through ponds leads to marked interception of nitrogen suggests that efforts to control nitrogen loads to estuaries should focus on nearshore areas, where groundwater flows directly to the estuary. Catchment areas upgradient from large bodies of freshwater are of lesser importance to loading, because their contribution of nitrogen to coastal waters will be modest.

Summary of the WBLMER nitrogen loading model

Thus far we have provided the background information and reasoning that was used to structure the different components of our model. The various parts and links of the model are numerous enough that it seems appropriate to summarize the components and calculations (Table 9) here before applying the model to the Waquoit Bay watershed. In our development of the WBLMER model we have been using concentrations as units, because these allowed us to readily do various calculations to show losses. In our summary of the model we shift to kilograms of nitrogen per year or kilograms of nitrogen per hectare per year, units more familiar to those working with nutrient loading calculations.

The values of Tables 2 and 6, which are used in our model (Table 9) are for the Waquoit Bay example. If the model were to be used for other sites, local values should be inserted if available. We intended, however, that the values of Tables 2 and 6 be used as default values for calculation of loading to other estuaries if comparable site-specific data were unavailable. In

TABLE 9. Summary of the WBLMER nitrogen loading model. The model estimates nitrogen delivery to and throughput on watershed surfaces, and to vadose zones and aquifers, and calculates resulting nitrogen loads from watersheds to receiving coastal waters. Units are shown in parentheses.

Nitrogen to and through watershed surfaces (kg N/yr)

Via atmospheric deposition to:

- *Natural vegetation* [1]: atmospheric deposition (kg N·ha^{-1}·yr^{-1}) × area (ha) of naturally vegetated land × 35% not retained in plants and soil
- *Turf* [2]: atmospheric deposition (kg N·ha^{-1}·yr^{-1}) × area (ha) of turf × 38% not retained in plants and soil
- *Horticultural land* [3]: atmospheric deposition (kg N·ha^{-1}·yr^{-1}) × area (ha) of horticultural land × 38% not retained in plants and soil
- *Impervious surfaces* [4]: {[atmospheric deposition (kg N·ha^{-1}·yr^{-1}) × area of roofs + driveways (ha)] × 38% not retained in plants and soil]} + [atmospheric deposition (kg N·ha^{-1}·yr^{-1}) × area of roads (ha) + runways + commercial areas]

Via fertilizer application to:

- *Turf* [5]: fertilizer application rate (kg N·ha^{-1}·yr^{-1}) × area (ha) of lawns × 34% of houses fertilizing lawns × 61% not lost as gases
- *Agricultural land* [6]: [crop fertilization rate × area (ha) under cultivation × 61% not lost as gases] − nitrogen removed as crop†

Nitrogen to and through vadose zone and aquifer (kg N/yr)

- Via nitrogen percolating diffusely from watershed surface [7]: [sum of items 1–6] × 39% not lost in vadose zone × 65% not lost in aquifer
- Via wastewater from septic systems‡ [8]: N released per person per year × people/house × number of houses × 60% not lost in septic tanks and leaching fields × 66% not lost in plumes × 65% not lost in aquifer§

Nitrogen loading to estuary (kg N/yr): sum of items 7 + 8.

† This term is appropriate where crops are not consumed within the watersheds. In watersheds with intensive export agriculture, harvest loss should be subtracted.

‡ Assuming conventional design. Where wastewater disposal is via cesspools (9% of the houses in the Waquoit Bay watershed, for example) the calculation for loading by cesspools is the same as for septic systems, except that losses in leaching fields are omitted.

§ Houses closer than 200 m from shore are not allotted to loss in aquifer.

many parts of the model, the data reviewed came from a wide variety of sites, so that the values synthesized from our literature survey are representative values.

NLOAD, an easy-to-use computer program that carries out the calculations summarized in this table and that allows simulations involving any land use scenario of interest, is available from the authors or from the Waquoit Bay National Estuarine Research Reserve, Waquoit, Massachusetts 02536, USA.

Our model estimate of nitrogen loading to estuarine receiving water is the sum of atmospheric, fertilizer, and wastewater nitrogen that enters the mosaic of land covers, and survives the complex gauntlet of loss that occurs on the watershed surface, the vadose zone, the aquifer, and in freshwater bodies, to emerge at the seepage face on the estuary shore.

The WBLMER model (Table 9) first estimates atmospheric deposition to surfaces of each of the major types of land use (natural vegetation, turf, agricultural land, and impervious surfaces) within the landscape. Deposition concentrations are corrected for evapotranspiration and evaporation. Then, within each major land cover type, based on changes in the concentrations of nitrogen and a series of loss terms, it calculates amounts of atmospheric nitrogen that is not retained or intercepted in plants and soil. The model then repeats the calculations considering nitrogen applied as fertilizer to turf and agricultural land. That fraction of atmospheric and fertilizer nitrogen that manages to go through the watershed surface is then subjected to further losses in the vadose zone and in the aquifer.

The WBLMER model also estimates nitrogen input from wastewater, calculates losses in septic systems, and by the difference estimates the amount of nitrogen inserted into the vadose zone. Then, the model allows losses in effluent plumes, and subtracts further losses during diffuse transport within aquifers.

The model assumes that if there are freshwater bodies present in the watershed, these will capture groundwater flow. The calculation also subtracts a loss of nitrogen incurred while traversing the freshwater body, and then subjects the surviving nitrogen to a further loss as it travels downgradient in the aquifer.

RESULTS: APPLICATION OF THE NITROGEN LOADING MODEL TO WAQUOIT BAY

Total nitrogen load and its uncertainty

The total nitrogen input to the lower eight subwatersheds of Waquoit Bay (Fig. 10) is calculated by our model as 115 432 kg N/yr (Table 10). Of this input, only 20% makes it to the edge of the estuaries. The loading of nitrogen from the watershed to the Bay is therefore 23 130 kg/N yr.

To compare and interpret estimates of nitrogen loading, we need to know how variable they are likely to be. Since it is difficult to find replicate watersheds, we could not estimate variation by the standard notion of replication. Instead, we (G. N. Collins et al., *unpublished manuscript*) assessed uncertainty by applying a bootstrap (Efron and Tibshirani 1991, Manly 1991, Diaconis and Efron 1983) procedure to estimate means and associated uncertainty. This procedure does not require assumptions about the statistical distributions underlying the data. We obtained a random sample of observations, sampled with replacement, from the pool of available observations for each variable. We then carried out the loading calculation using the means thus obtained for each variable. We repeated the calculation to obtain 2000 loading estimates, each calculated from a different set of means. The 2000 bootstrapped means were then used to calculate measures of variation (Table 11). The estimated standard deviation shows that individual measurements of nitrogen loading rates to Waquoit Bay will fall within 37% of the mean loading rate calculated using the WBLMER model. The vari-

TABLE 10. Estimates of nitrogen loading from atmosphere, fertilizer, and wastewater to the lower subwatersheds of the Waquoit Bay watershed, and losses during passage through different kinds of land cover on the watershed and to open water of Waquoit Bay, Massachusetts (in kg N/yr). Boldface numbers refer to totals for each source; lightface numbers show fate in each type of land cover.

Source of nitrogen	Nitrogen input to watershed	Percentage of nitrogen load to watershed	Percentage of nitrogen input lost within watershed	Total nitrogen input to Waquoit Bay	Percentage of load to estuary
Atmospheric deposition to:					
Natural vegetation	47 308	41	91	4 474	19
Turf	9 974	9	90	960	4
Cranberry bogs	1 488	1	90	143	1
Other agricultural land	90	0	90	9	0
Roofs and driveways	1 281	1	90	123	1
Roads, runways, commercial areas	3 407	3	75	863	4
Ponds†	801	1	56	350	2
Total atmospheric deposition	**64 350**	**56**	**89**	**6 923**	**30**
Fertilizer used on:					
Lawns	7 102	6	84	1 107	5
Golf courses	5 889	5	84	918	4
Cranberry bogs	2 778	2	54	1 290	6
Other agricultural land	816	1	84	127	1
Total fertilizer	**16 585**	**14**	**79**	**3 442**	**15**
Wastewater	**31 693**	**27**	**65**	**10 998**	**48**
Ponds upgradient‡	**2 804**	**2**	**37**	**1 767**	**8**
Grand total	**115 432**	**100**	**80**	**23 130**	**100**

† This refers to direct atmospheric deposition on ponds.
‡ This is an import from larger ponds or lakes that are deep enough to intercept the flow through the aquifer. Nitrogen additions to the watersheds upgradient of the ponds total 25 261 kg/yr (atmospheric deposition: 15 264 kg/yr; wastewater: 3621 kg/yr; fertilizers (6376 kg/yr). Ninety percent of this nitrogen is lost during travel to and within the ponds. The nitrogen that passes through ponds is then subject to 35% interception in the downgradient aquifer.

ability of the mean itself was, surprisingly, given the variability of components of the model, only 12%.

Repeated sampling methods such as the bootstrap may underestimate measures of uncertainty (Mueller and Altenberg 1985, Dixon et al. 1987), so we checked the error estimates by using a standard method for propagating errors (Meyer 1975). The error propagation method is much less computationally demanding, but assumes normality of the components. The standard deviation and standard error were remarkably similar to those obtained by the repeated sampling method (Table 11).

TABLE 11. Estimates of the variability of nitrogen loading calculated by the WBLMER model for Waquoit Bay, obtained by an error propagation method and by a bootstrap method. Values for standard deviation and standard error of the mean are shown as percentages of the mean nitrogen loading rate. Adapted from G. N. Collins (*unpublished manuscript*).

Method	SD	SE
Propagation of error	38	14
Bootstrap	37	12

Nitrogen loading rates obtained from our model should be used with the knowledge that individual estimates can fall within 37–38% of the mean value estimated by the model. This has ramifications. First, loading rates calculated using the model should not be interpreted and used as hard, well-defined values or thresholds, but rather as fuzzy guidelines derived from much data and many best guesses as to the effects of the various factors. Second, while the model allows calculation of nitrogen contributions from specific land cover parcels or sources, interpretation of the significance of such specific small inputs will be difficult. Although it is possible to calculate inputs from small sources, or through small land cover parcels, the model estimates are more suited to examine aggregate results of changes that affect larger areas of watersheds, such as might be the result of different zoning requirements, or shifts from agricultural to residential land uses, rather than to evaluate the effect of change in land use in a specific parcel.

The rather small standard errors, on the other hand, give some reassurance that the model makes fairly precise estimates. This precision makes it possible to challenge model predictions by comparisons to actual data from other sites. Moreover, the small standard errors show that the model predictions are sufficiently fine-grained so that it might be possible to carry out sensitivity analyses in which we can tell if changing specific variables makes a difference to nitrogen loading rates.

Relative contributions from different sources of nitrogen

We can use the calculations of the loading model to evaluate the relative magnitude and fates of contributions from atmospheric, fertilizer, and wastewater nitrogen to the Waquoit Bay watershed–estuary system (Table 10).

Atmospheric deposition is the principal source of nitrogen (56%) *to the watershed* of Waquoit Bay, followed by wastewater (27%) and fertilizers (14%) (Table 10, first and second columns of numbers). The losses of nitrogen from the three sources differ (Table 10, third column of numbers), so that inputs to receiving waters are not directly proportional to inputs to watersheds. Wastewater is the major source of nitrogen *to the estuaries*, providing 48% of the total nitrogen load (Table 10, fifth column of numbers). Atmospheric-derived nitrogen undergoes the greatest losses during transport, but still contributes 30% of the load to estuaries. Fertilizer nitrogen adds 15% of the load to the estuaries of Waquoit Bay. There is a small additional contribution of nitrogen (8%) from ponds upgradient.

The model calculations also provide an idea of the relative importance of different land cover types on nitrogen loads. The effect of land cover is exerted by the different fates that befall nitrogen from each of the three major sources (Table 2), and by the relative area of the different covers. We calculated delivery to and throughput from the principal land cover categories in the Waquoit Bay watershed (Table 10, fifth column of numbers).

Of the nitrogen that makes it to the water's edge in Waquoit Bay (Table 10, fifth column of numbers), the most came through residential parcels with septic tanks (48%, from wastewater), natural vegetation (19%, via atmospheric deposition), agricultural land (6.8%, from fertilizers and atmospheric deposition on bogs and farms), and turf (13%, from fertilizers and atmospheric deposition on lawns, golf courses, and other turf). Other pathways for nitrogen, regardless of source and land cover category, are probably insignificant in the Waquoit Bay system.

Clearly, wastewater from residential land parcels is the key source of nitrogen to Waquoit Bay. Natural vegetation, because of its large area, is the main pathway through which atmospheric inputs enter the estuaries. Agricultural land and turf are less important. While there are many other land cover types (Table 10), their smaller areas limit their role to reception and throughput of less than 5% of the nitrogen, an amount that must be considered insignificant and probably undetectable in view of the uncertainties associated with our estimates.

Data in GIS furnish an abundance of information. Inclusion of many different land cover types may appear to add comprehensiveness, but the level of uncertainty in loading calculations, and the lack of data, suggest limits to the benefits of inclusiveness of land cover detail. We included only a few selected land cover types in our model. To go beyond a few major land cover types, we need to determine the level of geographical detail that simultaneously offers a reasonable estimate of loading and does not exceed the level of uncertainty of the loading estimates. Distillation of complex land use data into fewer, more tractable variables has been proposed (Johnston et al. 1990, Kite and Kouwen 1992). Such approaches, in combination with estimates of uncertainty, could define both the level of detail that should be considered in loading calculations, and the limits of the applications of loading calculations.

We understand the need to make decisions not only about large-scale issues, such as zoning regulations for a whole watershed, but also about whether to approve a single construction project. It should be apparent from our discussion above, however, that knowledge about loading to watersheds is more relevant to whole watershed decisions than to issues dealing with specific projects.

The estuaries of Waquoit Bay have become increasingly eutrophied over past decades (Valiela et al. 1992), largely because residential areas near the Bay have become the dominant land use. Residential land use increases wastewater release to groundwater, and increases fertilizer use on lawns. Moreover, increases in residential land uses inevitably reduce natural vegetation, which is a major interceptor of atmospheric deposition. Recall from Table 10 that 91% of nitrogen falling on forests is intercepted before it reaches the estuary, but only 67% of wastewater nitrogen is lost in transit. Maintenance of accreting green covers on coastal watersheds would reduce nitrogen loading to estuaries.

The model results are best estimates, with broad ranges of variation owing to the intrinsic variability and by insufficient information about many terms in the model. Several topics in need of further study became evident as we developed the terms of the model. These include the assessment of dry atmospheric deposition and cloud droplet capture; the effects of distance and travel time on the denitrification of diffusely distributed nitrogen during transport in vadose zones and in aquifers; nitrogen retention in septic system leaching fields; nitrogen losses during transport in septic system plumes within aquifers; and a critical assessment of the degree of geographical detail needed to estimate nitrogen loading rates that are simultaneously appropriate to the variation of the estimates themselves, and still useful for researchers and managers. Probably the largest and most poorly known of all the issues is the amount of nitrogen loss in aquifers.

As coastal areas become more urbanized, increased nitrogen loading of estuaries will take place. Eutrophication occurs even though about 3/4 of the nitrogen delivered to watersheds is intercepted within them. Although storage, denitrification, and other loss processes occur, their combined effect is unable to keep up with the increasing nitrogen loads that anthropogenic activities deliver to coastal watersheds. The maintenance of estuarine water quality demands a concerted effort at limiting sources of nitrogen. The results of this paper suggest some options for management, but the focus of the effort should be the restriction of nitrogen sources on land.

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