Managing anthropogenic nitrogen inputs to coastal embayments: Technical basis and evaluation of a management strategy adopted for Buzzards Bay.

J. E. Costa[1], B. L. Howes[2], D. Janik[1,3], D. Aubrey[4], E. Gunn[5], A. E. Giblin[6]

Buzzards Bay Project Technical Report

Draft Final

September 24, 1999

1 Buzzards Bay Project, Massachusetts Coastal Zone Management, 2870 Cranberry Highway, Wareham, MA USA

2 UMass Dartmouth, CMAST

3 Massachusetts Coastal Zone Management

4 Woods Hole Oceanographic Institution

5 formerly Coalition for Buzzards Bay

6 Marine Biological Laboratory, Ecosystem Center

Note: This report will shortly be submitted to the journal *Estuaries*.

Current Citation: J. E. Costa, B. L. Howes, D. Janik, D. Aubrey, E. Gunn, A. E. Giblin. 1999. Managing anthropogenic nitrogen inputs to coastal embayments: Technical basis of a management strategy adopted for Buzzards Bay. Buzzards Bay Project Technical Report. 62 pages. Draft Final, September 24, 1999.

ABSTRACT

In 1990, the Buzzards Bay Project, a participant in the U.S. Environmental Protection Agency's National Estuary Program, developed a Total Maximum Annual Loads (TMAL) strategy to manage anthropogenic nitrogen inputs to coastal waters. This strategy, contained in a Comprehensive Conservation and Management Plan for Buzzards Bay, was designed to protect and restore water quality and living resources in more than 30 coastal embayments in Buzzards Bay. The recommended TMAL approach to manage point and non-point sources, was empirically based on a comparison of embayment conditions to estimated nitrogen loads together with a synthesis of previous studies of loading and ecosystem response. Existing nitrogen loads to the Buzzards Bay embayments were based on land use data contained in a Geographic Information System, and a well defined set of nitrogen loading assumptions for different kinds of land uses and sewage disposal. Drainage basins to each embayment were delineated by either land surface topography or groundwater elevations as appropriate. Recommended embayment TMAL limits were established with a tiered system that incorporated existing regulatory water quality classifications, together with embayment area or volume and hydraulic turnover time and depth, so that embayment specific TMALs were established.

The appropriateness of these recommended nitrogen loading limits was evaluated using seven years of data collected through a citizens-based water quality monitoring. Average summertime total nitrogen concentrations and a Eutrophication Index developed by the Buzzards Bay Project show a good correlation with estimates of nitrogen loading derived from land use data. Of the various methods used to characterize nitrogen loadings, a Vollenweider type model incorporating flushing and volume showed the best correlation to water quality parameters. The results of the citizens' water quality monitoring program has led the Buzzards Bay Project to revise its methodology, including reducing recommended TMAL N limits by as much as 50%.

INTRODUCTION

The addition of excessive amounts of nitrogen from anthropogenic sources is contributing to water quality degradation and habitat loss in near coastal waters throughout the world, and in many areas nitrogen loading is one of the most significant long-term threats that must be managed (Jaworski, 1981; Lee and Olsen 1985; Rosenberg 1985; Goldberg 1995; Valiela et al. 1997). In marine waters, nitrogen is typically the nutrient that limits algal primary production (Ryther and Dunstan 1971; Sanders et al. 1987; Boynton et al. 1982; Valiela 1984, 1995). Consequently, the addition of nitrogen to coastal waters from anthropogenic sources (referred to here as "nitrogen loading") can cause conspicuous increases in the growth and abundance of algae (D'Elia et al. 1986; Magnien et al. 1988; Lapointe and O'Connell 1989). Excessive algal production causes, either indirectly or directly, most of the adverse changes in coastal ecosystems attributed to anthropogenic nitrogen inputs (Costa et al. 1991) and is generally termed "nitrogen enrichment," "coastal eutrophication," or "nutrification." Due to the complex nature of the response of coastal ecosystems to nitrogen inputs, and because of the varied sources and pathways of transport, managing anthropogenic sources of nitrogen to coastal waters presents a significant challenge to environmental planners and managers.

In 1991, The Buzzards Bay Project National Estuary Program (BBP) developed a Total Maximum Annual Limits (TMAL) nitrogen management strategy to protect “nitrogen sensitive embayments” using a watershed mass loading approach to manage nitrogen inputs to protect and restore water quality and living resources in the more than 30 embayments that surround Buzzards Bay, Massachusetts, U.S.A. (Fig. 1). This strategy, contained in the Comprehensive Conservation and Management Plan for Buzzards Bay (BBP 1991), identified embayment specific nitrogen loading limits, and was based on scientific information and nitrogen loading assumptions used at that time, as well as practical management needs and considerations. This nitrogen TMAL strategy has been, or is now being used by local, state, and federal managers to address nitrogen loading in several Buzzards Bay watersheds. This paper explains the basis of the Buzzards Bay Project’s nitrogen management strategy, and evaluates it based on seven years of water quality data from 27 embayments in Buzzards Bay.

Responses of shallow coastal systems to nitrogen loading

The response of coastal ecosystems to anthropogenic contributions of nitrogen is complex and varied, and elsewhere we review this subject in more detail (Costa et al. 1992). In general, the response of coastal ecosystems to nitrogen loading are most pronounced in embayments with restricted water exchange, in stratified systems, where the amount of nitrogen added is large compared to the volume of the receiving water, and where large proportions of the bottom are within the photic zone and can sustain macrophytes. These differing responses therefore require different management limits that account for site-specific bathymetry and flushing.

The shallowness and bathymetry of an embayment may be one of the most important factors in defining an embayments response to nitrogen loading. Appreciable declines in seagrass distribution and production, other attached macrophytes, increases in unattached benthic algae and periphyton, as well as increases in phytoplankton have been documented for a wide range of shallow systems where the photic zone extends to the bottom of most of the embayment, which in turn translate into shifts in faunal assemblages (Kemp et al. 1983; Orth and Moore 1983; Brush 1984; Borum 1985; Twilley et al. 1985; Kautskey et al. 1986; Johansson and Lewis 1990; Lapointe and O'Connel 1989; Valiela et al. 1990). Nitrogen loadings to deep embayments also increase phytoplankton abundance and result in shifts in community assemblages (e.g., Benkemma and Cadee 1986; Caddee 1986), but the magnitude of change in the benthic assemblages is unlikely to be as great as the changes associated with shifts in benthic primary producer biomass in shallow embayments, unless of course the bottom waters of the deeper system become anoxic. In Buzzards Bay and surrounding areas, eelgrass (Zostera marina L.) is a dominant feature and nitrogen loading appears to have caused its decline in some embayments (Costa 1988a+b, Costa et al. 1992). The mechanism of decline appears to be increased shading from proliferation of phytoplankton, algal epiphytes, and drift macroalgae, but nitrate toxicity has also been speculated as a mechanism of eelgrass loss. The resulting increases in unattached drift algae may not only reduce eelgrass habitat but may also reduce habitat for shellfish and other invertebrates.

Like deeper systems, shallow embayments may exhibit depletion of water column oxygen concentration due to increase phytoplankton and benthic algal biomass and increased sediment oxygen demand from increased organic matter decomposition. The resulting hypoxic and anoxic events may result in fish kills (Costa et al. 1992; Valiela et al. 1992). Unlike deep stratified systems however, these hypoxic and anoxic events tend to be nocturnal to early morning events and short lived (D'Avanzo and Kremer 1994), typically occur during night time or early morning periods, particularly during cloudy, calm periods when water temperatures are high (Costa et al. 1992; Taylor and Howes 1994). The organic enrichment of sediments, together with the mass mortalities can cause dramatic shifts in the benthic invertebrate community in these shallow embayments.

In Figure 2, we outline some of the ecosystem responses that may occur in shallow Buzzards Bay embayments. This figure does not convey the fact that all the responses shown do not occur concurrently, and that any particular embayment may go through a series of phases or stages. The progression of ecosystem response can be classified into a series of stages (Table 1). This classification system is somewhat artificial and subjective, and considerable variation in ecosystem response exists, but this approach was necessary when the BBP developed its nitrogen management strategy to enable comparisons to be made of specific embayment ecosystem responses and loading rate wherever detailed water quality data was not available, and was used to identify the nitrogen TMALs for Buzzards Bay embayments. These loading limits reflected the belief that whatever the nature of the receiving waters, bay volume, degree of water exchange with offshore waters, and bottom area within the photic zone were principal features that defined ecosystem response and needed to be considered by managers establishing limits for nitrogen loading.

Nitrogen sources and management priorities in Buzzards Bay

In Buzzards Bay, anthropogenic sources of nitrogen include sewage treatment facilities, septic systems, acid rain, and fertilizer used on lawns, golf courses, and fertilizer and animal waste exports from agricultural land (Kelly et al. 1991; Werme et al. 1991). The nitrogen from the watershed sources enters the Bay via streams, groundwater, and direct effluent discharge. Most of the nitrogen entering Buzzards Bay comes from sewage treatment discharges; the next highest amounts are from home septic systems (Table 2).

The central portion of Buzzards Bay does not exhibit bay wide water quality or living resource degradation observed in other estuaries like Chesapeake Bay (Seliger 1985; McCarthy et al. 1988; Jordan et al. 1990) or Long Island Sound (LISS 1993). Instead, the effects of these nitrogen inputs are localized nearest the sites of input.

The New Bedford sewage treatment facility discharges 24 million gallons of effluent per day, or

40% of the annual nitrogen load to Buzzards Bay. Despite the magnitude of this input, the discharge largely affects only the waters within three miles of the outfall (Howes and Taylor 1989; Hampson 1988; Rhoads 1988; Turner et al. 1999). In contrast to central Buzzards bay, many embayments that fringe Buzzards Bay that have already experienced water quality declines or loss of eelgrass due to nitrogen loading (DEP 1989a+b; Costa 1988a+b; Costa et al. 1996). These declines are not the result of the New Bedford outfall or other large point source outside the embayment drainage basins. Rather, the degradation observed is the result of nitrogen loading from surrounding landuse, particularly from the cumulative impacts of so called "non-point sources" of nitrogen within the drainage basins.

In the majority of these embayments, septic systems (irrespective of siting or design) are the largest source of anthropogenic nitrogen. This nitrate travels great distances in groundwater without attenuation and with little chance of uptake by plants (Weiskell and Howes 1992). Fertilizer application on lawns and agricultural land are of secondary importance in these embayments. For example, in Buttermilk Bay, though 25% of the homes are sewered, septic systems still account for more than 45%, and lawn fertilizer use 11% of the nitrogen entering this coastal system (Table 2). When impervious surfaces are considered, residential development non-point sources account for nearly 60% of the nitrogen load to this estuary. In at least two Buzzards Bay embayments, agricultural land use is appreciable, and fertilizer applications and livestock wastes are major non-point sources of nitrogen. In 28 embayments dominated by non-point source pollution, unsewered residential land generally accounts for 20% to 70% of all inputs, with a bay wide mean of 50% (Buzzards Bay Project 1994). These findings illustrate that bay-wide estimates of loadings as in Table 2 can be misleading as to where managers must focus their efforts.

Nitrogen loading scales

Embayment specific water quality models require considerable financial and human resources. Because of these costs, coastal managers must often consider more generalized loading limits or simpler models where immediate action is necessary to protect water quality and living resources before critical impacts occur. If such standards are to be adopted, some consensus must exist as to the appropriate "yardstick" or loading scale to characterize nitrogen loading. In earlier marine ecosystem literature, nitrogen loading to receiving waters was most typically characterized as loading per unit area or loading per unit volume (e.g., Nixon 1983; Nixon et al. 1986). The most commonly used yardstick or loading scale used to evaluate oceanic or large estuary waters was loading per unit area. This approach likens coastal waters to an agricultural system; that is, a given fertilizer application rate results in a corresponding response in primary production. When areal loadings are reported for marine ecosystems, loadings may also be given as nitrogen per volume of water. Such a scale acknowledges that embayments of different depth may have different responses to equivalent areal nitrogen loads. That is, resulting concentrations of DIN, total nitrogen, chlorophyll, and other water quality measures in response to a specific input of nitrogen are fundamentally a function of the volume of the receiving waters.

It has been often observed that water quality improves when flushing increases in a coastal embayment as a result of man-made or natural alteration to the mouth of an embayment, and it is generally believed that better flushed systems can tolerate greater pollutant inputs. That turnover time is an important factor when evaluating the effects of nutrients inputs on marine systems has been suggested by Nixon and Pilson (1980) and others. Valiela and Costa (1988) showed that accounting for flushing can dramatically affect the relative rankings of coastal systems as compared annual aerial or volumetric scales of loadings. If flushing time weighted loadings are chosen as the scale to evaluate nitrogen loading, this scale can be further refined by taking into account the turnover time of the limiting nutrient as well. This approach was established by Vollenweider (1976, 1985) who related ecosystem response of freshwater systems (particularly chlorophyll concentrations) to phosphorus loadings and the turnover time and volume of the system. For marine systems, there is less consensus on the relationship between turnover time and marine ecosystem response, and how a Vollenweider approach might apply. Boynton and Garber (1988) found that the Vollenweider approach did result in a good correlation between loading and total chlorophyll a in the water column of receiving estuaries, and subsequently some scientists and managers have employed such a flushing coefficient.

Each of these four nitrogen loading scales are summarized in a mathematical form in Table 3. Whatever the nitrogen loading scale chosen by managers, nitrogen loading limits to an embayment can be based on empirical observations of ecosystem changes as measured along the selected scale. Managers can then use these observations to predict how an embayment may respond to additional loadings or to establish nitrogen loading limits to meet management goals, and it was this approach that was used by the BBP.

The Vollenweider approach offers an important conceptual framework to think about nitrogen loading in coastal ecosystems, and we consider it as the best scale for establishing limits in coastal embayments. Since the term $(1+\tau_w^{1/2})$ approaches 1 for short turnover times, the Vollenweider term loading scale is nearly identical to a simple turnover time weighted scale for well-flushed embayments. For longer turnover times, the Vollenweider term can affect relative rankings of loading. Table 4 shows that choice of loading scale affects the relative ranking of nitrogen loading for some large estuaries. For example, Narragansett Bay, which does not have bay-wide anoxic events, appears more heavily loaded when flushing time is not considered than Chesapeake Bay and Long Island Sound, estuaries that do exhibit large scale anoxic events.

Recommended nitrogen management strategy in Buzzards Bay

The Buzzards Bay Project's 1991 Comprehensive Conservation and Management Plan included recommendations to manage nitrogen inputs to protect and restore water quality and living resources in the more than 30 embayments that surround Buzzards Bay. Specifically, the BBP recommended a tiered system of embayment Total Maximum Annual Loads (TMALs) for nitrogen that accounted for turnover time or area of the receiving waters, embayment bathymetry, and existing water quality management classifications (Table 5). The recommended loading limits were based on research and monitoring studies in Buzzards Bay and elsewhere (described below), and ease of implementation by managers. Because the recommended nitrogen loading rate limits incorporate site specific hydrographic characteristics, and total maximum annual nitrogen limits for each embayment could be calculated using one of the following equations (areal or volumetric limits) identified in Table 5. The choice of which limit to use depended on depth and flushing time criteria.

Critical annual load (in kg yr^{-1})

$$= \text{Areal limit} \bullet \text{bay area (in } m^2) \div 1000$$

or

$$= \text{Volumetric limit} \bullet \text{volume at half tide (in } m^3) \bullet (1+\tau_w^{1/2})/\tau_w \div 1{,}000{,}000$$

where τ_w is the hydraulic turnover time in years.

Application of the Buzzards Bay Project's nitrogen management strategy requires measurements of embayment volume, summer water turnover times, delineation of the surrounding drainage basin (i.e., identification of nitrogen sources), and quantification of nitrogen from point and non-point sources based on an evaluation of each individually owned land parcels. From a management point of view, these tasks can be accomplished with modest financing and manpower.

The BBP acknowledged that this approach may not always be the most appropriate. These tiered limits were to be used for embayments with not degraded until other more defensible loading limits could be justified using improved ecosystem models. For embayments already critically impacted, it was suggested that historical assessments of loading and water quality and living resources be conducted to identify to what degree existing nitrogen sources must be eliminated to achieve some past desirable environmental conditions.

Figure 3 shows how the recommended loading limits using this approach change with increased flushing time for a hypothetical shallow embayment. At 4.5 days, the loading limit used by managers was to be based on the areal scale equation, rather than the Vollenweider-turnover time scale because of uncertainty at the time as to whether flushing would remain as directly a forcing factor in ecosystem response. The aerial limit was also used because some managers were also concerned that for longer flushing times, recommended loading limits using the volumetric-flushing equation may not be practical or achievable.

In creating this nitrogen management strategy, the BBP borrowed elements of a nitrogen loading strategy adopted in 1985 by the town of Falmouth, MA to protect its coastal embayments. In contrast to the BBP approach, however, the 1985 Falmouth nitrogen bylaws and regulations were based on nitrogen "critical concentration" limits in the water column rather than watershed mass loading limits. The Town regulations specified that in “high priority” sites, water column total nitrogen concentrations limit of 0.32 ppm total nitrogen was established, with 0.50 ppm for medium priority sites, and 0.75 ppm total nitrogen for the lowest priority sites. Implementation of the Falmouth regulations required an analysis of future development potential so that the contribution of nitrogen from the proposed new development alone, or the proposed development together with potential future growth, would not be expected to increase water column total nitrogen concentrations above allowable limits. The increases expected in total nitrogen concentrations in the receiving waters resulting from the proposed development as well conditions at full "buildout" of the watershed were based on a set of loading assumptions adjusted for embayment volume and water turnover times to approximate "steady state" concentrations.

Falmouth's management strategy and similar ones have both technical and management limitations. The Falmouth bylaw established a water quality standard that relied upon an assessment of existing water column total nitrogen concentrations. It is problematic to make permitting decisions based on water column nitrogen concentrations alone because these nitrogen concentrations do not always correlate well with nitrogen contributions to the watershed. For example, because groundwater typically travels 0.3 to 1.6 m per day around Buzzards Bay, nitrogen from septic systems that are kilometers from shores or streams may not reach coastal receiving waters for many years or even decades. Thus, judgments on permits were being based on existing receiving water inputs, and did not take into account recent development whose groundwater conveyed nitrogen was still in transit to the bay. Even if actual loadings to the receiving waters were known, in embayments where benthic macroalgae comprise a significant portion of primary production, nitrogen accumulated by these algae may keep water column nitrogen concentrations relatively low. In these embayments, total nitrogen concentrations in the water may seem unexpectedly low. Finally, managers had to consider practical longstanding problems such as which analytical method was most reliable for measuring low levels of total nitrogen in seawater (D'Elia et al. 1986), and the location, number of sampling stations and sampling frequency required to adequately characterize embayment water quality for regulatory action.

For these reasons, the BBP rejected the use of nitrogen concentrations in the receiving waters as the basis of management decisions and instead adopted theoretical estimates of mass loading as the basis of management action. The BBP's watershed loading approach also made implementation of watershed management strategies simpler for managers to communicate and implement. The BBP retained most other elements of the Falmouth bylaw such as the use of a tiered set of limits for different habitat values, the incorporation of flushing time and volume to establish annual nitrogen limits, and the use of mass loading assumptions. The BBP also included an evaluation of future subwatershed growth potential based on local zoning bylaws. The Buzzards Bay Project's mass loading approach was attractive to managers because estimating annual anthropogenic inputs to coastal waters directly from groundwater and streams was considered too costly to be routinely used for calculating loads. In addition, by the late 1980s, watershed mass loading assumptions were already being used to protect public drinking water supply wells in Massachusetts.

Managers and planners in local and regional government have a variety of tools available to meet nitrogen watershed goals including growth management, sewering, requiring the use of advanced nitrogen removal technology for both public wastewater facilities and private onsite systems, and encouraging smaller lawns, agricultural best management practices, procuring open space in the watersheds of sensitive areas, or even dredging harbor entrances to increase hydraulic turnover rates. These and other options have been addressed in a myriad of management publications and which strategy or strategies are adopted will depend on economic and political considerations. Whatever management options are considered, total watershed planning, rather than relying on the permit process alone, will meet with the most success, especially in addressing the cumulative impact of non-point sources.

<u>Formulation of BBP recommended N limits</u>

To establish nitrogen loading recommendations contained in the 1991 Buzzards Bay Comprehensive Conservation and Management Plan, the BBP classified Buzzards Bay embayments and selected other coastal systems into one of the 5 eutrophication stages (or occasionally an intermediate stage) shown in Table 1. This subjective evaluation of water quality and ecosystem response was used because detailed water quality and living resource information was lacking for Buzzards Bay embayments. Information such as eelgrass abundance or declines (from Costa 1989), occasional water quality studies for selected embayments, shellfish bed closures (stormwater related in most of Buzzards Bay and therefore considered a proxy for development-related nitrogen loading inputs), shellfish catch statistics, qualitative evaluations based on diving excursions, and anecdotal information about ecosystem health. Loading rates and responses of large estuaries as in Table 4 were also considered.

These subjective evaluations of ecosystem response were compared to estimates of nitrogen loading using the loading assumptions and methods described below which are nearly identical to those used in 1991. For the development of specific management recommendation for a watershed, the Buzzards Bay Project recommended that a parcel level land use analysis be conducted (i.e., evaluate each parcel for existing loading and for future development potential). However, in 1990 when the nitrogen loading standards were developed, such a parcel level land use analysis was conducted in only one Buzzards Bay embayment. Therefore the BBP developed algorithms for estimating population, dwelling units, and nitrogen loading in each Buzzards Bay subbasin using 1984 1:25,000 land use data contained in a Geographic Information System maintained by the state of Massachusetts (MassGIS, summarized in Costa et al. 1994) .

These loadings were adjusted for embayment volumes, area, and flushing times. The flushing times used were based on a draft report prepared by Aubrey Consulting Inc. (1991, subsequently finalized as ACI 1995). Other studies where either ecosystem response or nitrogen loadings were characterized were also used by the Buzzards Bay Project. Figures 4a+b (loading as g m^{-2} yr^{-1}) and Fig. 5 (loading as g m^{-3} Vr^{-1}) show how this ecosystem classification system was applied to a number of bays and estuaries to derive the BBP proposed TMALs in 1991. The MERL mesocosm experiments are shown on both figures, but using the Buzzards Bay embayment classification scheme, the aerial loading scale for "deep" embayments would be applied to Narragansett Bay. Other embayments like Waquoit Bay were included on both scales because the estimate of flushing time for the upper 1/3 of the estuary complex is close to the crossover point for use of the aerial and volume-turnover scales. Loadings shown for Waquoit Bay were based on original BBP methodology. In the Discussion we describe new estimates for Waquoit Bay loadings that were developed.

By comparing estimated loadings against ecosystem response as shown in Figs. 3 and 4 and data for other embayments, the BBP developed a tiered system of acceptable total maximum annual loading limits for nitrogen (TMALs) to be used with an existing water quality classification system employed under existing state and federal environmental statutes. These state and federal designations were already in place for Massachusetts bays and harbors as a way of describing the degree of degradation that had already occurred in coastal embayments. These classifications were primarily based on fecal coliform data, which in Buzzards bay was the result of both point and non-point sources of pollution. These classifications consisted “SC” (lowest ranking for salt water), “SB,” “SA” (highest ranking salt water classification), and a special category of SA called “Outstanding Resource Waters” (ORW). Buzzards Bay had no SC designated waters, but two were SB, with the remainder SA, half of which were also designated ORW. The ORW designation has special implications since under the antidegradtion provisions of the US Clean Water Act, this designation can be used to stop any new large pollution discharge to a surface water body.

This water quality classification system was also incorporated in the nitrogen TMAL strategy to provide managers with a mechanism to allow lower standards for coastal ecosystems already severely impacted by eutrophication or other forms of pollution. For example, some systems like the Acushnet River, which had both a sewage treatment plant discharge (Fairhaven sewage treatment plant) and wet and dry weather CSO discharges (from New Bedford), made it one of the most eutrophic of the Buzzards Bay embayments, and was classified as a stage 4 ecosystem (subsequently dry weather CSO discharges have been eliminated and water quality improved). On the aerial loading scale proposed by the Buzzards Bay Project, it is well above the SB water quality standard limit recommended for that estuary, and is intermediate between the MERL 4x and 8x treatments. Similarly, the East Branch of the Westport River characteristic of Stage 3 conditions, was found to have among the highest loading rates of any embayment.

Costa (1988a) reported historical declines of eelgrass in Apponagansett Bay and Waquoit Bay, and loading estimates for different historical periods for the estuary are included in Figure 5. Inner Apponagansett Bay and the upper Wareham River were reported by Costa (1988a) to be devoid of eelgrass, had accumulated drift algae, and occasional summertime hypoxic events. Both estuaries have a depopulated benthos and have black mayonnaise-like sediments typical of stages 3 or 4 (Costa, pers. observ.). Waquoit Bay's collapse of eelgrass beds, accumulation of macroalgae, and increased frequency of anoxic and hypoxic events were characterized as stage 3. Sippican Harbor eelgrass population in the upper harbor have declined (Costa 1988a) and DO concentrations in the upper are occasionally quite low (ENDICO, Marion, MA, unpublished data), and is typical of stage 2. Most other Buzzards Bay embayments, like Buttermilk Bay and Nasketucket Bay have the nominal or localized impacts (such as eelgrass loss in upper portions of the estuary or selected covelets typical of stage 1; Costa 1988a+b). Because most stage 3 embayments were assumed to have loadings above 200 mg m^{-3} during Vr, and most stage 1 and 2 embayments were below this rate, this rate was selected as one of the target loadings in the tiered strategy of Table 5. Similar reasoning was used for the other limits.

Included in Fig. 4 is Cederwall and Elmegren's (1980) observations of macroalgal shifts in Laholm Bay. These authors observed a decline in certain species of phaeophytes at a loading of 13 g m^{-2} yr^{-1} (below the SA limit for shallow poorly flushed systems), and a proliferation of green algae (Cladophora and Enteromorpha) with a loading of 19 g m^{-2} yr^{-1} (above the SA limit, Fig. 4). Comparable ecosystems changes occurred in Apponagansett Bay, but at slightly lower aerial loadings (Fig. 4).

Through this type of evaluation process, the BBP developed and selected the tiered TMAL limits for Buzzards Bay embayments in 1991. Like other areas of pollution management, when these "action limits" needed to be established, they were based on "best available scientific information". The choice of the specific round numbers selected was merely a convenience for managers who require specific standards. At the time, it was believed these limits in Table 5 would be protective for **most** embayments (at least in preventing stage 3 or 4 conditions described above). Creation of these loading limits for Buzzards bay embayments was valuable within the legal and regulatory context of environmental management, since it gave environmental agencies standards and criteria required for defining permitable pollutant loads for point discharges, or needed as the basis for establishing laws and regulations controlling non-point pollution sources.

However, it was apparent from the start that this was not always the case, and not all ecosystems responded similarly to equal loading rates even when adjusted for area, volume or flushing. Because of the limited water quality information available or uncertainties of the loading data, it was unclear if these discrepancies were the because the established limits were too high, nitrogen sources omitted, or flushing rates inaccurate. For example, although the onset of eelgrass decline might have been avoided in Apponagansett Bay if the ORW limit were adopted in the past, this limit would have failed to prevent the near complete collapse of eelgrass in Waquoit Bay. Were assumptions of loading and flushing for Waquoit Bay wrong, or were some bays may be more sensitive to nitrogen? This question could not be answered until better water quality data became available and loadings refined.

As noted above the crossover at 4.5 days from the volumetric-flushing scale to the aerial scale was a management decision made in the face of uncertain information. At the time of adoption, it was believed that the aerial scale had some validity for some poorly flushed Buzzards Bay embayments. Moreover some scientists and managers felt that the turnover time scale alone would result in very difficult to achieve loading limits for less poorly flushed embayments. Because of these uncertainties, it was felt that the more "lenient" aerial scale was a more defensible approach until more water quality data could be obtained. These proposed limits were adopted by the Buzzards Bay Project through a series of meeting and workshops with scientists and managers in 1990 and 1991. At these meetings the scientific community present felt that, despite all the uncertainties inherent in the Buzzards Bay Project's approach, it was a good starting point for management action. One flaw in the criteria however was that for certain shallow but poorly flushed embayments, the aerial based loading limit proved more restrictive.

In 1990, while the nitrogen management strategy was still being finalized, the BBP worked with three Buzzards Bay municipalities to implement the first embayment nitrogen management strategy in Buzzards Bay (Horsley Witten and Heggeman Inc. 1991), and the first of its kind in the country. Because Buttermilk Bay was not yet over the BBP's recommended limits (recommended limits were 54,000 kg yr^{-1}, existing loading was estimated at 41,000 kg yr^{-1}, buildout loading was estimated at 65,000 kg y^{r-1}), the municipalities identified zoning changes as the strategy most likely to be implemented, coupled with sewering already planned. In the spring of 1991, after sewering had been approved to eliminate approximately 18,000 kg yr^{-1}, each municipality adopted at Town Meeting zoning bylaws to increase the minimum size of lots on unsubdivided parcels of land to 70,000 sq. ft., thereby reducing the number of dwellings that could be built in the watershed by 450, equivalent to loading of 9,000 kg yr^{-1}. Since this first implementation case, the BBP's loading recommendations have been used to develop management recommendations for other watersheds, and are also being used to establish nitrogen loading limits for several sewage treatment plants in Buzzards Bay.

When the BBP developed this mass loading watershed nitrogen management approach, it was recognized by managers as a useful management tool, but there was concern as to whether the recommended loading limits were justifiable. The BBP recognized at the outset that water quality data from Buzzards Bay embayments would be needed to both generate support for management action, and to evaluate and refine the recommended nitrogen loading limits. Consequently in 1992, the Buzzards Bay Project partnered with the citizen's group the Coalition for Buzzards Bay and Dr. Brian Howes (who had established a citizen based water quality monitoring program through the Woods Hole Oceanographic Institution) to fund a Buzzards Bay water quality monitoring program. Concurrently, the Buzzards Bay Project funded or conducted additional land use studies to based on parcel land use information and revised watersheds introduced by the Cape Cod Commission in 1994.

In this paper we report data from this water quality monitoring program collected between 1992 and 1998 from 27 Buzzards Bay embayments to help answer the question: "Were the nitrogen loading limits recommended in the 1999 Buzzards Bay Comprehensive Conservation and Management Plan justifiable?"

Methods

<u>Nitrogen loading methodology and assumptions</u>

In this report, we have estimated embayment watershed loadings either via parcel level land use evaluations (the most accurate approach), or by use 1:25,000 land use data contained in a Geographic Information System maintained by the state of Massachusetts (MassGIS). In 1990, the Buzzards Bay Project developed algorithms for estimating population, dwelling units, and nitrogen loading based on this data, and these methods were summarized by Costa et al. 1994, but are also highlighted below. For some watersheds, loadings were based on parcel level land use analysis, but for most subwatersheds, these have not been completed, and we evaluated the MassGIS data with ARCInfo™ and ARCView™ software.

One of the most important estimates for determining nitrogen loadings is ascertaining the number of residential dwellings in a watershed, and the number of those on septic systems. GIS Parcel data enables quick direct counts, but use of MassGIS land use statistics requires extrapolating the number of dwellings from the aerial coverage of 4 residential land use categories that are mapped. The BBP developed housing density coefficients by counting actual housing unit densities for polygons overlain on aerial photographs. These estimates of housing unit and population, also agreed reasonably well with 1990 U.S. and municipal census data. Assumptions were used to estimate road and lawn area in each drainage basin. Table 6 summarizes the loading assumptions that were employed for both MassGIS 1:25,000 land uses coverages and parcel coverages. Sewering maps were used to eliminate septic system loading s from sewered areas.

To estimate future development, the BBP assumes that 40% of existing forested land is unbuildable because of wetlands or need for infrastructure support and open space, and the remainder will be developed under existing municipal zoning laws. For parcel land use coverages, potential new units are calculated for both unbuilt, subdivided and un-subdivided parcels. The number of potential parcels in undeveloped, unsubdivided land is calculated by considering lot size zoning and other local regulations and bylaws, and after subtracting 15% of the land to account for subdivision roads and related infrastructure and open space (see Horsley Witten and Heggeman 1991 for an example of this approach).

Some of the nitrogen loadings estimates used in this paper have been revised somewhat from estimates previously used by the Buzzards Bay Project in earlier reports to formulate the tiered loading system reported in the Buzzards Bay CCMP. For example, we did not adequately account for some sewering in the Apponagansett Bay watershed, and originally overestimated loadings from septic systems. Combined sewer overflows in the New Bedford area have been greatly reduced in the last 5 years, and these reduced loads were used to evaluate the citizen monitoring program water quality data. Similarly, the 1991 Buzzards Bay Project estimates for Waquoit Bay were higher and closer to the ORW limits for that estuary, than is now believed, particularly when septic system loadings adjusted for groundwater transport time (Shaum et al. 1994).

For some other embayments, the Buzzards Bay Project or a municipality has funded a detailed nitrogen loading and land use evaluation. We use those estimates or actual housing units from those studies rather than those based on cruder land use evaluations in this report. Table 7 shows our current best estimates of watershed loadings in each embayment. These loads represent assumed loadings that would eventually enter coastal waters through groundwater and surface water. We recognize these loadings are overestimates of actual receiving water loadings since they do not account for either lag time of transit, and any attenuation that may occur during transit. However, most watersheds have 50 to 75% of development within 5 years transit times to coastal surface waters. Shaum et al. (1994) found that because of the proximity of most homes to Waquoit Bay, average transit time for all septic systems in the watershed was only 10 years.

Although these nitrogen loading rates are implied as loadings "to the embayment," they are really presumed watershed loadings to groundwater and streams that are expected to eventually reach the receiving waters (typically as nitrate) with little attenuation. Because any mass loading study is so dependent on these kinds of loading assumptions, we explain the basis of each below. The management implications of the time lag between watershed inputs and receiving water inputs are addressed in the Discussion.

• Septic Systems

All conventional septic systems, both properly operating or "failing" with respect to pathogen removal, release to groundwater large amounts of ammonia that is rapidly converted to nitrate. These loadings often represent the single most largest source of nitrogen in many coastal watersheds, and assumptions relating to their discharge is a fundamental part of any watershed nitrogen mass loading estimate. The Buzzards Bay Project adopted a per capita annual nitrogen load to groundwater of 2.7 kg from onsite septic systems. This value was based on the various studies described in Table 8 and several lines of inference. The wide range of values reported in Table 8 reflect the fact that methodologies and approaches differed among studies, and not all nitrogen species were examined, nor were water flows always adequately documented, which may explain why nitrogen retention by septic systems ranged was given a range of 10% to 90% in one study (Valiela et al. 1997) .

Many of these early studies characterized effluent from septic tanks, but few studies characterized losses due to biological activity in the soil absorption system (SAS), particularly in the kind of septic systems found in Massachusetts. To better understand reported nitrogen losses from septic systems, water flow and nitrogen concentrations leaving the house, the septic tank, and the SAS (including unsaturated soils over groundwater) must be accounted for. Wilhelm et al. (1994) provide a good review of the potential losses and attenuation of nitrogen in the components of a conventional septic system.

Per capita daily water flow can vary greatly among residences. In a review of data from 71 residences in nine studies, EPA (1980) noted that 10% of the observed flows were less than 87 lpcd (23 gpcd) and 10% were greater than 235 lpcd (62 gpcd). These flows need to be accounted for when estimating mass loading of nitrogen from septic systems, since nitrogen concentrations in both septic tank effluent and leaching system effluent correlate inversely with water flow (Whelan and Titus 1985, BBP unpublished). Thus, mean total nitrogen loading should not be calculated by multiplying mean volumes times mean concentrations reported from different studies, but by averaging loading rates in studies where both concentration and flow were measured.

The dosage rate (volume per unit area) of effluent in a leaching field may also affect the ability of the unsaturated zone to attenuate nitrogen. In sandy coastal soils, Cogger (1988) found that higher than normal dosage rates (4 x), greatly increased DIN concentration in groundwater immediately under experimental leaching fields. The higher concentration was not the result of higher dosage raes with corresponding less dilution by groundwater. Rather, increased nitrogen concentrations at the higher dosage rate was the result of a reduction or elimination of the unsaturated zone with a resultant reduction in biological action. Thus, failing or poorly designed septic systems with high dosage rates may contribute more nitrogen to coastal systems than systems with lower dosage rates.

In the past, septic systems installed in the Buzzards Bay area consisted of a cesspoool or a septic tank and one or more “beehive” leaching chambers. These systems account for the majority of existing onsite wastewater disposal systems. Revisions to the Massachusetts sanitary codes in 1995 now require the use of leach trenches or fields. There have been few studies of nitrogen removal ability of the various leach field designs or old design systems or cesspools. However, the studies by Weiskell and Howes (1991, 1992), which included cesspools in a coastal area, and are described below, show that mass loading predictions of nitrogen export from cesspools and septic systems with saturated leach fields was lower than expected.

Reviews of septic system and household sewage discharges by EPA (1980, 1992) showed that per capita nitrogen load from human urine and feces and household graywater (excluding garbage disposal systems) ranged from 2.2-6.2 kg y^{-1}. In these studies, mean annual nitrogen load per capita was 3.2 kg N in blackwater (toilet discharge) and 0.7 kg N in graywater, or 3.9 kg total per capita N discharge. In reviewing this data, Owen Ayers Associates, Inc. (1991) concurred with these estimates. In another EPA review (1992), annual nitrogen production in human waste was estimated to be about 40.2 g kg^{-1} body weight, which would equal only 3.1 kg per year for a 77 kg adult. Presumably this estimate is lower than actual household loads which includes other household inputs (e.g., ammonia cleansers) and background nitrate concentrations in the water supply which was relevant in a few studies. A comprehensive review of conventional septic system design performance and loadings was recently summarized by Crites and Tchobanoglous (1998).

Of the total nitrogen discharged from a home into a septic tank, 10% to 15% is generally presumed to settle out as septage sludge, denitrified or volatilized before leaving the septic tank and entering the leaching field (Laak and Coates 1978; Andreoli 1980; Laak 1982). Of the effluent discharged from the leaching field, an additional 10% to 40% has been presumed to be typically lost by denitrification, volatilization, or through ammonia binding to sediments before reaching groundwater as nitrate. We believe the likely range of removal for an entire conventional septic system to be 20%-50%. Using the above 3.9 kg per capita mean annual N load for graywater and blackwater sources cited above and a potential 20%-50% total system loss (including losses in the unsaturated zone beneath the leaching field), then the likely range of per capita nitrogen discharge is 2.0 to 3.1 kg of nitrogen are discharged to groundwater.

Several studies have shown good correlations between density of development and groundwater nitrate concentrations (e.g. Persky 1987; Frimpter et al. 1988, Tinker 1991). Seldom have authors of these kinds of studies used their results to estimate nitrogen loads from septic systems, nor are these estimates calculable from published results. Beginning in the late 1980's, the Cape Cod Commission, a county government regulatory agency, required the use of effluent concentrations of 35 ppm N and a system flows of 212 lpcd in its assessments to protect drinking water supplies. The 35 ppm concentration adopted by the Cape Cod Commission has since been criticized as too low based on the literature, but their selected 212 lpcd discharge rates are theoretical maximum system design flows, which are about 25% higher than the typical 167 lpcd system discharge reported by EPA. Thus their not loading rate used for regulatory purposes was 2.7 kg N per capita.

To evaluate these regulatory assumptions, Nelson et al. (1990) estimated average residential septic system nitrogen loads by mapping the 25 year history of well nitrogen concentrations and housing permits in a Cape Cod subdivision with a well defined recharge area and comparing it with a solute transport model by Knokikow and Briederhoft (1980). Nelson found that the best fit to the groundwater model was with a per capita nitrogen load of 2.5 kg yr^{-1}. This model required an estimate of N loadings to groundwater from lawn fertilizers which included a 30% leaching rate of fertilizer for commercial applications and 60% leaching rate for homeowner applications, values which are higher than those generally accepted (see below). Using a more realistic 20% lawn leaching rate, a nitrogen export from septic systems to groundwater of 3.2 kg N per capita gave the best model fit (Nelson, pers. comm.), which is slightly higher than the expected range cited above.

A more thorough land use loading analysis has been conducted by Valiela et al. (1997) on Waquoit Bay in cape Cod, MA. Based on water use records, these authors concluded that water use in their study area was 182 lpcd, but based on a literature review that discharge concentrations were 70 ppm, for a net load of 2.9 kg N per capita load. In this study however, the authors concluded that an additional 43% of N was attenuated within the septic system plume within a short distance of discharge, resulting in a net load of 1.9 kg per capita load for septic systems near the coast. This loading rate gave a good fit between estimated watershed loadings and actual loadings measured in streams and groundwater. If this estimate were true, it would have important implications to nitrogen management.

While mass-loading model studies like Nelson's and Valiela et al., or correlations like Persky's and Frimpter's studies demonstrate the strong link between elevated groundwater nitrogen levels and development. Their calculations of septic loadings are based upon too many assumptions, and should not be used alone to define loadings from septic systems; more direct measurements of septic system discharges and their groundwater plumes are needed, and several studies are now underway. Weiskel and Howes (1991) conducted a detailed nitrogen mass-load analysis for a subbasin of Buttermilk Bay employing per capita nitrogen loads based on census occupancy rates, and actual residence water usage rates. This data was correlated with nitrogen concentrations measured directly within delineated groundwater "stream tubes", coupled with measured rates of groundwater discharge based on household water use. They concluded that per capita annual nitrogen load from septic systems to groundwater was 3.0 kg.

In the Management Plan for Buzzards Bay, the Buzzards Bay Project (1991) used a 2.7 kg per capita loading rate based on an assumed 30% nitrogen reduction and 3.9 kg per capita household N discharge. Adoption of the 2.7 kg per capita loading rate by the BBP was based primarily on the above pre-1991 wastewater literature values, and was adopted in part to ensure regional regulatory consistency in nitrogen management with the Cape Cod Commission, a county government regulatory agency.

To evaluate the BBP's nitrogen TMALs we have employed a 2.7 kg per capita loading in this report. For estimating existing and historical nitrogen loadings from septic systems in each subbasin, we used actual occupancies within each subbasin estimated from 1990 U.S. Census data (Costa et al. 1994). Data from local planning authorities suggest that actual occupancy ranges from 1.7 in communities with an influx of summertime residents to 2.9 for more stable year round populations, with a Buzzards Bay regional average of 2.2. For future-potential conditions, the BBP employs an occupancy rate of 3.0 persons per dwelling unit to account for potential changes in demography within the communities, particularly conversion of summer residences to year round residences. In Massachusetts and elsewhere, septic systems must be designed for 2 persons per bedroom. Since there is average of 2.5 to 3 bedrooms per unit in this area, the theoretical maximum residential occupancy rate is 5 to 6 persons per unit. It is unlikely such average occupancy rates will occur here without dramatic social or economic changes.

To calculate nitrogen loads in sewage from commercial and private structures other than residential units, the Buzzards Bay Project adopted nitrogen loading rates reported in USGS (1988).

• <u>Rainfall on embayments and impervious surfaces</u>

Precipitation contains nitrogen from natural and anthropogenic atmospheric sources that may affect coastal primary productivity (Pearl 1985). DIN concentrations reported for the Northeast U.S. and Buzzards Bay area generally range from 0.3 to 0.7 ppm (Pearl 1985; Stensland et al. 1986; Godfrey 1988; Valiela and Costa 1988). Dry deposition has been documented to account for up to 50% of atmospheric nitrogen deposition in some areas (Hanson and Lindberg 1991), but the estimate of total wet and dry precipitation of DIN in Massachusetts was still equivalent to only 0.69 ppm times total rainfall by Godfrey (1988). Because 1.14 m yr^{-1} of precipitation falls in the Buzzards Bay area (30 year average, Cranberry Experiment Station, courtesy DeMoran), using the wet plus dry estimate of Godfrey, the BBP adopted 7.9

kg ha^{-1} as the precipitation load directly on embayments.

More recent data from the National Atmospheric Deposition Program shows that DIN loadings north of the Buzzards Bay watershed has remained constant during the period 1982 to 1998 at roughly 4 kg ha^{-1} (Fig. 6). Thus the 7.9 kg ha^{-1} precipitation loading rate adopted by the Buzzards Bay Project and other planning agencies appears to be a realistic estimate of wet and dry deposition for the region if dry deposition is presumed to by of equal magnitude to wet deposition (e.g., Valiela et al. 1997). The Buzzards Bay Project ignored the contribution of particulate and dissolved organic nitrogen in precipitation because it was believed that this nitrogen was more refractory and less biologically active. Some, like Valiela et al. (1997) have argued that dissolved organic nitrogen (DON) in precipitation, which equaled 43% of wet deposition total dissolved nitrogen loadings, may not be completely refractory and needs to be considered in evaluations of watershed loadings.

Precipitation falling upon impervious surfaces (roads, driveways, roofs, etc.) is generally directed to surface waters or groundwater with little opportunity for uptake or attenuation by plants or microbes. In addition, stormwater, especially from roads is often contaminated with other nitrogen from animal wastes, car exhaust, and other sources. Koppelman (1982) estimated that nitrogen concentrations in road runoff were 1.5 ppm (107 μM). Valiela and Costa (1988) observed only 0.4 ppm in a limited sampling program around Buttermilk Bay. Valiela et al. (1997) estimated a wet+dry DIN +DON load leaving all impervious surfaces was 15.0 kg ha^{-1}. Since stormwater volume from impervious surfaces is estimated as 90% of rainfall, using a stormwater concentration of 1.4 ppm, we use in this report a loading rate of 15.3 kg DIN ha^{-1} for all road surfaces, and 7.3 kg ha^{-1} for roofs and sidewalks on the assumption that less bioavailable nitrogen accumulates on roofs than on roads. Since dry deposition accumulates on these surfaces, these loading rates take into account wet+dry deposition rate.

In its calculations, the Buzzards Bay Project estimated that impervious surface of roofs and property pavement on private lots is estimated to average 187 m^2 (2,000 ft^2) per unit; road surface area is calculated directly from municipal maps, or occasionally from watershed averages for total road length times average road width.

• Lawns

Reported application rates of lawn fertilizers vary widely, and equally variable are estimates of DIN that leach into groundwater or potentially runoff following application. Consequently, consistent standards have not been adopted for management purposes. For example, the Long Island 208 Plan (1978) adopted an annual application rate of 49 kg ha^{-1} with 60 percent leached to the groundwater, or a 29 kg ha^{-1} loading rate. For nitrogen management around a Buzzards Bay embayment (Buttermilk Bay), Horsley and Witten (1991) assumed 146.7 kg ha^{-1} as the annual application rate and a 30 percent leaching rate, or 73 kg ha^{-1} loading to groundwater.

In a survey of hardware stores in the Town of Orleans, MA, Giblin and Gaines (1990) found that annual fertilizer use was 2.3 kg per lawn (an average of both fertilized and unfertilized lawns). Since typical lawn size has been estimated to be 465 m^2 on Cape Cod (CCPEDC 1979), this equals an application rate 49 kg ha^{-1}. From a survey of home owners in one Cape Cod community, Nelson et al. (1990) determined that homeowner application rates were 137 kg ha$^{)1}$, and that professional lawn care companies applied 227 kg ha$^{)1}$. Because 38% of the respondents used lawn care companies, the weighted application rate was 171 kg ha$^{)1}$. Valiela et al.(1997) assumed an actual average annual application rate of 104 kg ha-1 of lawn for their Waquoit bay watershed evaluation, but assumed only 34% of homeowners applied fertilizer. The Buzzards Bay Project adopted an application rate of 147 kg ha^{-1}, closer to the Nelson et al. study.

Fertilizer leaching rates from lawns have been the subject of increasing debate. Petrovic (1990) suggested that leaching rates are typically less than 10%, but are nonetheless quite variable and may exceed 50%. High leaching rates are typically associated with certain types of fertilizer (e.g. NH_4NO_3 fertilizers), high single dose application rates, porous sandy soils, or winter application. Petrovic presumed that surface runoff was negligible except where slopes were steep or if fertilizer applications coincided with heavy rains and overland runoff. For Waquoit Bay, Valiela et al. (1997) assumed a 24% leaching rate through the vadose zone. Because a large extent of the Buzzards Bay drainage basin has porous sandy soils, often with hilly terrain we used the 20% combined fertilizer leaching and runoff loss rate adopted by the BBP to account for Buzzards Bay conditions, with all of this nitrogen presumably eventually reaching coastal waters. Thus, the effective fertilizer nitrogen loading rate was 29.4 kg ha^{-1} (=147 kg yr^{-1} applied x 20% export). For residential development greater than 1/4 acre (most of Buzzards Bay has 1/2 to 1 acre lots), we assume an average lawn size of 5,000 sq ft contributing 1.37 kg N yr^{-1}.

• Agricultural land

Agricultural cropland in the Buzzards Bay drainage basin is dominated by cranberry bogs. Cranberry bog fertilizer nitrogen releases are often directly to streams which may discharging directly to coastal waters without travel through intervening ponds. In 1991, the Buzzards Bay Project adopted a 18.0 kg/ha loading rate based on Teal and Howes (1995, initially submitted as a 1990 BBP draft report). In the Teal and Howes 1990 draft report, 18 kg/ha represented the output of nitrogen from the bog and 13 kg/ha represented the net flux of nitrogen. The Buzzards Bay Project adopted the higher export value in part because of concerns that certain nitrogen exports were not included. Howes and Teal (1995), subsequently revised their analysis and concluded that net nitrogen losses were 24.7 kg/ha for actual bog production area. This loading represented the loss of loading based on actual production areas, and excluded ditches, berms, retention ponds and surrounding land. Since the MassGIS data includes these supporting and surrounding features, Howes and Teal's revised estimate is not appropriate for direct application to the MassGIS data. However, in one municipality (Wareham) where cranberry bogs are very prevalent, we have actual bog surface area from Massachusetts Wetland Conservancy maps (1990 coverage, base map 1:2,500 scale base map) and MassGIS data (1985 coverage, 1:25,000 scale base map). In Wareham, land in cranberry bog production in the MassGIS data layer totals 926 ha whereas actual bog growing area on Wetland Conservancy maps show total bog surface area to be 660 ha, or 71.2% of the MassGIS coverage. Using the Howes and Teal published estimate of nitrogen export, cranberry bog loading from using the Mass GIS coverage should be revised to 17.6 kg/ha.

In 1991, The Buzzards Bay Project adopted 10 kg N ha^{-1} for fertilizer export from other common crops in the region which include corn, strawberries, orchards, and nursery plants. These agricultural land estimates were from USGS (1988) and SCS (1990 draft report, finalized in 1992). Valiela et al. (1997) used an application rate of 136 kg N/ha for all cropland, but after subtracting losses to volatization, and vadose zone and during aquifer transport, and adding precipitation inputs to cultivated lands (1.44 kg ha), concluded that croplands contributed 22.5 kg/ha to receiving coastal waters.

In some Buzzards Bay embayment watersheds, concentrated animal feedlot operations like dairy farms with up to 20 animals per acre are an important nitrogen source. For these watersheds, dairy cows were assumed to produce 75 kg N per "animal unit" (454 kg of animal) per year (SCS 1992). The amount of this nitrogen reaching groundwater depends on many factors including manure management practices and vegetative buffers along streams. For the subwatershed evaluation presented here, we assumed 25% of this animal waste nitrogen reached groundwater or surface as inorganic nitrogen, except for open feed lots immediately along the shore of a stream or bay where 50% of the nitrogen is assumed to reach the bay.

• Other non-point sources, and point sources

The contribution of nitrogen from other types of development and non-point pollution sources used by the Buzzards Bay Project were adopted from USGS (1988) and EPA (1980). Any permitted point sources of pollution discharging to an embayment, such as sewage treatment facilities or industrial outfalls, were included in the nitrogen loading assessment. The amount of nitrogen from these sources generally can be determined by on site measurements of flow and concentration where available, but where this data is unavailable, discharge limits of flow or concentration are used. For planning purposes it is appropriate to use discharge limits established by the regulatory permit since this maximum allowable discharge is often eventually met.

Physical and hydrological features of embayments and drainage basins

The Buzzards Bay Project, in conjunction with the U.S. Geological Survey created subdrainage basins (see Figure 1) based on either land surface topography on 1:25,000 scale USGS topographic maps or groundwater contours . Land surface topography was used largely on the western shore of Buzzards Bay where granite bedrock underlies sandy glacial sediments. Under these conditions, groundwater elevations approximately match surface topography and groundwater flow is approximately perpendicular to land surface contour lines. In these watersheds, surface water flow often exceeds groundwater flow, although much of the river flow is groundwater fed. On the northern and eastern shores of Buzzards Bay, groundwater permeates through a glacial till and outwash and drainage basins were delineated from 10 ft. groundwater elevations. In these embayments, groundwater discharge generally exceeds surface water flow. We believe these delineations are suitable for the embayment loadings reported here because few large nitrogen sources were found near the boundary of any embayment subbasin, but we recognize that local managers may require more detailed subbasin delineations before management action is adopted.

A preliminary estimate of hydraulic turnover times of Buzzards Bay embayments was prepared for the Buzzards Bay Project embayments (ACI 1991, 1995), and these estimates were used for many of the Buzzards Bay Project's preliminary evaluations of nitrogen loading to many embayments, and many of these values are used in this report. ACI used at least two, and in some cases three approximations of flushing. These methods were a simple volume to tidal prism ratio "box model" to give what in most cases would be a theoretical lower limit of turnover time if there were no return of any portion of outgoing tidal water. The second method was a "spatial" model using a plausible range of dispersion coefficients for the upper 1/3 of the embayment, one of the criteria for defining flushing in the BBP methodology. The third approach was a "numerical model", but this was done on only 5 bays, and not necessarily for the upper third of the embayment, and in at least two bays the method proved inappropriate.

Generally, for consistency, we used the average of the minimum and maximum range for the box model estimates of the upper 1/3 of the estuary as cited in the Aubrey 1995 study. In four unenclosed, "open" well flushed embayments (Clarks Cove, Aucoot Cove, Mattapoisett, and Megansett Harbor) we felt the spatial model did not apply because tidal currents sweeping around these embayments and dispersing N inputs were an important factor in flushing and a simple tidal prism estimate of the box model were more appropriate. The numerical model was used for Wareham River estuary, and averaged with the spatial model for Apponagansett Bay because of the range of values. Where available, we used more recent and better defined flushing estimates. For example, hydraulic turnover time for West Falmouth Harbor were based on ACI (1995_, using the segment weighting in Costa 1996. Flushing in Allens Pond, Nasketucket Bay, and Onset Bay were based on Geyer et al. 1997. Flushing time for Onset Bay, Little Bay, Allens Pond, Pocasset River, Hen Cove were based on Geyer et al. 1997 and 1998. These authors used both dye studies and freshwater residence times. Because there was in general a good agreement with the two approaches, those values were averaged. Woods Hole Group (1999) calculated flushing for areas of Falmouth and Bourne using a new methodology identifying "local" and "system" hydraulic turnover times. The local hydraulic turnover time is closest to the methods used in the other reports, but because it measures flushing into the immediate adjoining area, these flushing estimates were faster than earlier ones. For the three Woods Hole Group studied systems included in this report (HenCove, Red Brook, Pocasset Harbor-Barlows Landing, and Squeteque Harbors), we used the average of the Woods Hole Group (1999) estimate and the ACI (1995) report estimates if available, except for Hen Cove where the Geyer et al. (1997) estimate was used because it was based on a dye study and freshwater residence time.

Nitrogen loading for Buzzards Bay embayments were adjusted for these flushing times using the Vollenweider equation in Table 3. Loadings and flushing times for other embayments were taken from other publications. The volume of each embayment used in the loading calculations was based volumes at mid-tide estimated from nautical chart bathymetric isopleths and reported tidal range (ACI 1995). To calculate the tidal prism volume, the mean tidal range can be multiplied by the bay area, unless the bay had extensive tidal flats, or if the tidal range varies considerably in different parts of a shallow bay, in which case these areas were accounted for. A summary of all assumed flushing times, embayment volumes, mean depths at half-tide, and estimated watershed loadings used elsewhere in this report are summarized in Table 7.

Water quality monitoring

Estimates of existing nitrogen loading for each embayment (using the different loading scales) were related to existing summertime water quality through the use of site specific data collected by the Buzzards Bay Citizens' Water Quality Monitoring Program, a joint project of Buzzards Bay Project, the Coalition for Buzzards Bay (a local citizens group), and Howes' laboratory. The citizens measure dissolved oxygen concentrations with Hach Kits™, secchi depth, salinity, and temperature) approximately 15 times between June 1 and September 30. Water samples in each embayment were collected in July and August on four dates. These water samples were analyzed for dissolved and particulate organic nitrogen, nitrate+nitrite, ammonia, orthophosphate, and chlorophyll *a*. The analytical methodologies are described by Howes and Goering (1994). Data for seven years (1992 to 1998) are used in this report.

Generally two to four sites within were monitored in each embayment for both oxygen and nutrients. In some smaller embayments only one site was monitored; in some larger embayments, five or more sites were sampled. Generally only water quality data from stations in the innermost half of the embayment were compared to our estimate of loadings. Samples for nutrient analysis were taken on outgoing tides, while oxygen and secchi data included both incoming and outgoing tides because the oxygen measurements were needed in the early morning hours, generally taken between 6-9 AM, as indicated by Taylor and Howes, (1994). Secchi depths were not attainable at many monitoring stations because of shallow depths, and these data often depended on occasional sampling in deeper areas.

Mean summertime values of dissolved oxygen percent saturation, secchi disk depth, chlorophyll a, total organic nitrogen (TON), and dissolved organic (DIN) were combined in a Eutrophication Index modeled after a similar index developed by Hillsborough County, Florida, U. S. A. (Hillsborough County 1991). The Hillsborough County Index included 7 water quality parameters (% dissolved O_2 saturation, Chlorophyll ***a***, total coliform, light penetration, total phosphorus, TKN, and BOD). Because the focus of the Buzzards Bay Eutrophication Index was on the effects of nitrogen loading, BOD and coliforms were omitted from our index, and DIN and DON were included instead of TKN because of problems associated with measuring low levels of nitrogen in seawater using the TKN methodology (D'Elia et al. 1985).

As noted by Harkins (1974), it is acceptable for parameters used in a water quality index to show interaction or interdependence, but there must not be any direct redundancy in the parameters. Thus, it is acceptable to have DIN and TON included in a water quality index as we have done, but inappropriate to include both TKN and DIN because these parameters have redundancy in that the both include the direct measure of NH_4 concentrations. On the other hand we felt that TON and chlorophyll a were not redundant because, even though a large amount of particulate organic nitrogen, which is often greater than 50% of TON, is composed of phytoplankton. This is because PON also includes zooplankton and detrital material as nitrogen reservoir, and because algae have some storage capacity for nitrogen independent of their chlorophyll concentration.

Like the Hillsborough County Index, the Buzzards Bay Eutrophication Index evaluated water quality parameters against a scoring curve like the one we used for mean secchi depth shown in Figure 7. As shown, if the summertime mean secchi depth was less than 0.5 m, then a score of 0 was received for that parameter. Conversely, a secchi depth greater than 3.0 m received a score of 100. If Secchi depth was between 0.5 and 3.0 m, the score was calculated using the following equation:

$$\text{Score}=(\ln(\text{value})-\ln(0\text{ pt. value}))/(\ln(100\text{ pt. value})-\ln(0\text{ pt. value}))$$

The 100 and 0 point values for each parameter is shown in Table 9. All summertime means of the five parameters were applied to the equation above. These end points shown in Table 9 were chosen for Buzzards Bay based on the authors' knowledge of conditions typically found in a range of southern New England embayments. During the course of the water Citizen Monitoring Program, 100 and 0 point values changed somewhat, and Table 9 reflects the current values employed. Most notably, at inception, mean summertime oxygen concentration were used in the index. This was later changed to the mean of the lowest 1/3 of all summertime values with a concurrent change in the 0 point and 100 point values.

The Eutrophication Index equaled the mean of the scores for the five parameters (i.e., all parameters were equally weighted). In this paper we also show an Alternate Eutrophication Index scoring without oxygen scores included. When several sites were monitored in an embayment, we averaged only those data from sites in the upper half of the estuary because of the steep gradient in water quality near the mouth to Buzzards Bay conditions. Correlation coefficients shown were calculated based on summertime means and unless specified were calculated with the log of the loading scale and log of the water quality parameter, except for the Eutrophication Index.

Eelgrass index

Eelgrass habitat area was obtained from Costa (1988a+b) for each embayment was also compared to nitrogen loading. In the embayments of Buzzards Bay, eelgrass typically grows in between 0.3 m MLW and 2.5 m MLW. In the more eutrophic embayments eelgrass grows to only 1 m MLW, and in the less eutrophic, better flushed embayments of Buzzards Bay, eelgrass grows to 2.5 m or more, with a maximum depth exceeding 6.0 m along the outer shores of the most flushed portions of Buzzards Bay (Costa 1988a, Costa et al. 1992). To use eelgrass cover as an indicator of nitrogen loading we examined the ratio of eelgrass habitat area to "potential" eelgrass habitat area. To approximate a minimal “potential” habitat area we digitized (in ArcView™) contours on USGS 1:25,000 quadrangle maps from the 6 foot (1.8 m) to a depth of 0.3 m. Because there is no 0.3 m (1 foot) contour line on these maps, this depth was approximated by digitizing 20 m from the highwater mark (i.e., the coastal boundary) or around areas indicated as “tidal flats” (stippled areas) on the maps. One embayment (Squeteague Harbor) was not included in our eelgrass evaluation because more than half the embayment is of a depth of less than 0.3 m.

Results

Figure 8 shows TN concentrations compared to a) aerial, b) volumetric c) simple flushing-volumetric, and d) Vollenweider-term flushing-volumetric loading scales. The aerial and volumetric scales correlated more poorly with mean summertime total nitrogen concentrations than when flushing time is employed. The correlation coefficient was nearly identical (0.70) for both the simple flushing model and the Vollenweider model.

As noted above, the Buzzards Bay Project did not account for nitrogen inputs to the embayments from offshore (the reasons for which are discussed below). Buzzards Bay waters are low in total nitrogen, and generally contain less than 1 µM DIN--the most bioavailable nitrogen to most likely to cause the adverse ecosystem changes of concern to coastal managers. However, annual tidal volumes are immense, and can actually account for a sizeable portion of the annual nitrogen load to an embayment. Some embayments were influence by eutrophic river discharges as shown in Fig. 8 and subsequent figures. Specifically, Marks Cove (MRK), a small covelet at the mouth of the Wareham River (WAR), and Little River (LIR), near the mouth of the Slocums River have small watersheds with relatively small nitrogen inputs, but both have large river inputs with elevated DIN at the entrance to the embayment. As a result, conditions in Marks Cove and Little River are very similar to those of the Wareham and Slocums River respectively, and higher than would be expected compared to other embayments with comparable loading. Thus in the case of Marks Cove and Little River, our estimate of nitrogen loading is inappropriate since its water quality is dominated by conditions in their estuarine system, and correlation coefficients werre not calculated with water quality data from these two embayments.

Estimates of embayment total nitrogen loading using the Vollenweider-term flushing adjusted loading scale (log transformed) were compared to the five Eutrophication Index parameters (secchi depth, DIN, chl a, TON, and oxygen saturation) measured between 1992 and 1998 through the Citizen's Water Quality Monitoring Program (Figs. 9-13 respectively). For nearly every nitrogen related variable monitored, there was a significant correlation of water quality with nitrogen loading estimates ($P>0.05$), albeit the degree of correlation varied markedly, with oxygen saturation showing the worst correlation ($r^2=0.21$) and TON giving the best correlation ($r^2=0.70$) among the measures used in the Eutrophication Index. Part of the bad correlation with oxygen percent saturation may have been the result of considerable variability in temperatures and cloudiness among the seven year period that could have canceled embayments specific differences evident during any specific summer, and individual years (e.g. Fig. 14) of oxygen percent saturation data showed a better correlation that the 7 year average.

When the five Eutrophication Index parameters are combined to obtain the Eutrophication Index score (Table 9), a far better correlation was obtained with the Vollenweider term flushing scale (Fig. 15, r^2=0.80) than with any of the five individual parameters in Figs. 8-12. Because oxygen shows the worst correlation with our estimates of nitrogen loading, we have also considered an “alternate” Eutrophication Index that is the mean of the scores of 4 parameters (without oxygen). This Alternate Eutrophication Index shows a slightly better correlation with Vollenweider term-volume loading than the Eutrophication Index with oxygen (Fig. 16).

In Figure 17, we add offshore inputs of nitrogen from Buzzards Bay using an assumed 1 uM DIN (typical summertime concentration we have observed just offshore in Buzzards Bay) times the annual tidal exchange (annual tidal prism volume estimated from ACI 1991, 1995) to calculate anthropogenic loadings on the x-axis and compare these loading estimates to actual summertime total nitrogen concentrations in the water. For Marks Cove, instead of 1 μM, we use a typical lower Wareham River DIN concentration of 2.8 μM, and 2.0 μM for Slocums River inputs to Little River (7 year mean). With these “offshore” loadings, two trends are apparent. First, Marks Cove and Little River mean TN now falls more in line with the response of other estuaries to similar loading. These extra loadings do not explain completely the observed water quality because the water quality in these two embayments remain dominated by water quality from their larger neighboring estuaries. Second, the “noise” around the loadings for undeveloped watersheds is considerably dampened and these points are more tightly clustered on the graph. This is due to the fact that anthropogenic loadings in undeveloped watersheds accounts for only a small percentage of the background loadings from offshore.

While tidal prism loadings improved the relationship between loadings and total nitrogen concentration, the correlation between Eutrophication Index and loadings with prism did not markedly improve (not shown). This is because total nitrogen concentration in Buzzards Bay embayments tends to be similar to total nitrogen concentrations in central Buzzards Bay (about 0.28 ppm), and modest anthropogenic inputs do not appreciably alter these concentrations in the receiving waters, and therefore do not appreciably change the scores.

Eelgrass as an indicator

The ratio of existing eelgrass habitat area to potential eelgrass habitat area (0-2 m bottom area) is shown in Fig. 18 as it relates to nitrogen loading (Vollenweider-term scale). Included in this is eelgrass cover and loading estimates in Waquoit Bay on three dates, and loadings to Apponagansett Bay on two dates. Like the other indicators of nitrogen, there is considerable variability in response among the embayments, but a clear trend of declining eelgrass coverage with loadings. Some of the variability in response may be also due to the fact that we have not excluded tidal flats and high energy areas in our estimate of potential eelgrass habitat. The inclusion of eelgrass coverage in the Buzzards Bay Project's embayment Eutrophication Index is planned, but was not included here.

DISCUSSION

When the various measures of ecosystem response were compared to loading adjusted to flushing time, correlations were variable. DIN correlated weakly with our loading estimates, but this was expected because DIN is so reactive in coastal waters. The relationship between early morning oxygen saturation and loading did not correlate as well as we expected suggesting that other ways of characterizing oxygen concentrations or demand are needed. One of the problem faced in a citizen-based monitoring program is that volunteers collect samples at convenient site such as along shore or on a dock, whereas low oxygen are most likely to be observed in deeper central areas. Also, low oxygen concentrations in shallow embayments is very intermittent and depend on weather and other factors. We believe oxygen saturation remains a useful tool, but continuos monitoring equipment in embayments would likely result in better characterizations of water quality and nitrogen loading impacts.

The best correlations were observed with more comprehensive measurements such as total nitrogen or the BBP's Eutrophication Index. Because of the poor correlation between loading and oxygen, our "alternate" Eutrophication Index (without oxygen scores) show the best correlation with nitrogen loading. This observation suggests that a comparative water quality evaluation of coastal water quality can be based on a relatively small number of summertime water quality sampling dates.

Some of the observed "noise" or poor correlations observed among all the parameters could be due to a variety of factors and limitations in our data. In some embayments we had too few sites or they were sampled too infrequently, or in only a handful of years. We may have relied on crude flushing estimates, or our estimates of actual loadings may have been flawed or contained omissions. Despite these obstacles, and the variability of the data, overall this monitoring program did show a consistent pattern of declining water quality with increased nitrogen loading and generally support the BBP's 1991 approach with some exceptions.

The water quality monitoring program also indicated that aspects of the Buzzards Bay Project's nitrogen management approach needed to be revised, and more fundamental technical issues need to be addressed as discussed below.

• Nitrogen calculations and sources and losses not included in the loading estimates

The estimates of embayment loading used in this report are actually estimates of loadings to the watershed that we believe will eventually reach the embayment, and not actual loadings now discharging to the embayment. For most embayments we believe the difference between the estimated watershed loadings and actual inputs to the receiving waters are modest. For example, overall 43% of residential development in Buzzards Bay is within 0.8 km of the coast (BBP 1991), and for some Buzzards Bay embayments we estimate up to 80% of development is within 1 km of the coast of groundwater fed streams. These distance represent a transit time of 5 to 10 years for groundwater discharges like septic systems. Thus most development older than 10 years is already reaching coastal waters. This is especially true since most Buzzards Bay communities are experiencing growth rates of 5% to 10% per decade. Another important factor is that for most embayments, our estimate of nitrogen loading was based on 1984 or 1990 land use data (which are compared to 1992 to 1996 water quality data) and these estimates do not include construction after those dates. For small watersheds, the errors of loading resulting from these transit lags or recent development may cancel each other, but for embayments with large watersheds that have had considerable inland development during the 1980s, we have overestimated actual receiving water nitrogen inputs.

Accounting for these lags and losses can be important. Based on land use and actual occupancy rates, we estimate that annual loading from the drainage basin to Buttermilk Bay was about 23,500 kg in 1990 (Horsley Witten and Heggeman 1991, revised with actual occupancy rates and sewering and using 1985 land use data). In contrast, Valiela and Costa (1988) found that actual loadings in 1987 from streams and groundwater to be 15,600 kg annually. The Buttermilk Bay drainage basin is large, and we estimate loadings in the upper watershed to take 20 years or more to reach the bay. This time lag, together with unaccounted for losses (from wetlands and other sources) may account for the 33% discrepancy in estimates. Chi et al. (1994) estimated that existing loadings to Waquoit Bay are 64% of existing watershed loadings (in part because of a building boom on Cape Cod in the 1980s and 1990s), simply based on theoretical lag times without any attenuation.

In our loading estimates, we did not include loss terms for DIN transport through the watershed. While it is generally believed that nitrate from sewage and other sources can travel great distances in groundwater with little attention because of the absence of conditions favorable for denitrification (Gilliam et al.1974; Hagedorn et al. 1981; Kimmel 1984; Caezan et al. 1986; Trudell et al. 1986), we recognize that even very minor annual losses (e.g. Parkin 1987) could become significant if groundwater transit times are decades long. Valiela et al. (1997) used a 30% loss of nitrogen (even as nitrate) in their model for transit through groundwater, but this generalization remains controversial, and whether losses of nitrate can occur in deep groundwater high in oxygen and that apparently lacks carbon as an energy source remain a topic of debate.

In many Buzzards Bay embayments, some groundwater discharges feed first into wetlands, ponds, and rivers before entering coastal waters, and these may be sites of additional nitrogen attenuation (Peterjohn and Correll 1984). In developing management strategies for these portions of watersheds, managers may need to assume some diminished loading coefficient for these areas when estimating watershed loading from land use. The estimated loadings used in this paper did not include a loss term for wetlands uptake (except the Wareham River estuary), but the Buzzards Bay Project has begun to use a 30% attenuation coefficient as an average for upper watershed landuse regions in larger watersheds to better estimate actual losses due to transit through streams, wetlands and ponds. In the case of the Wareham River, loadings were reduced by about 15% using the loss coefficient. Actual losses in these areas are probably wide ranging, and is under review by a number of investigators.

In 1991, the Buzzards Bay Project did not use precipitation on unvegetated land in its loading calculations. This was done for several reasons. First, nitrogen in acid rain has remained constant or has declined in the northeast U.S. during the past 20 years (Figure 6), yet water quality has declined appreciably in many estuaries in the region during the same period, suggesting that acid rain could not alone explain water quality declines. Many studies show that accreting forests, grasslands, and subsoils effectively capture most nitrogen in precipitation (Heil et al. 1988) and nitrogen concentrations in groundwater under forested land around Buzzards Bay is very low, sometimes approaching analytical limits of detection. For example, in the Buttermilk Bay watershed, DIN concentrations under an accreting forest was somewhat less than 2 μM, but concentrations were elevated to 400 μM when this groundwater passed under a densely developed area near shore (Weiskell and Howes 1992). The 2 μM DIN concentration suggests an effective nitrogen loading rate of 0.17 kg ha^{-1} for forested land. When this loading rate is included in the Buzzards Bay Project's nitrogen loading evaluation, forested land, which often account for 40-70% of the GIS land use coverage of these embayment drainage basins, accounted for an average of only 1.2% of the nitrogen load among the 30 embayment watersheds evaluated. Even when this background is assumed to be 5 μM DIN, “forest loading” averages of only 3.0% of total loads, and only 4.2% of loading if it is added as a constant to all watershed land area. These relative contributions are lower than precipitation contributions already included in our loading model such as impervious surfaces (mean=7.3% of total loadings for 30 subwatersheds) and precipitation directly on an embayment (mean=11.0%). Fig 19 shows the addition of this precipitation to forested land. Considering the plot is at a log scale, the shifts in loading are imperceptible from figure 8d. Although this loading represents a small background noise, we believe that future loading estimates should incorporate a loading coefficient 0.17 kg ha^{-1} (equivalent to 2 μM DIN) to better account for background nitrogen input from undeveloped lands in large undeveloped watersheds where this term may have more relevance.

Similarly, we have not included nitrogen from offshore waters in our loading assessment, nor is this factored into the Buzzards Bay Project's tiered loading limit strategy. On an annual basis, total nitrogen inputs from offshore can be quite high. However, nitrogen inputs from offshore waters tend to be considerably higher in organic nitrogen than inorganic nitrogen. Although this organic nitrogen (mostly particulate, e.g., plankton) is important in the embayment nutrient cycling, it is DIN inputs from land sources that appear to cause the excessive algal production we observe in eutrophic bays. Certainly the low DIN in offshore waters is a sizeable portion of any estuary's budget-- e.g., in the exercise of adding a 1 μM DIN load in tidal prism water shown in Fig. 17 we found that this prism annual DIN loading ranged from 10 times anthropogenic DIN inputs for the most undeveloped watershed down to 12% of anthropogenic land inputs for the most heavily developed watersheds. In a practical sense, however, estuaries do not become "polluted" by clean offshore waters as found in Buzzards Bay. Because of the time, expense, and uncertainties of accurately calculating tidal prism N loading, it is probably reasonable for managers to ignore in most instances these "background" DIN loadings and concentrate on land based anthropogenic inputs. If a simplified offshore loading factor is employed as we have done in Fig. 17 (which was very relevant for interpreting the water quality in Marks Cove at the mouth of the eutrophic Wareham River estuary and Little River near the mouth of the Slocums River), a new set tiered loading limits would need to be defined.

Denitrification by estuary sediments (e.g., Jensen et al. 1988) is another variable that can be considered in evaluating nitrogen loading impacts, but these losses are so variable and difficult to quantify, it is not practical for managers to consider this loss in evaluating ecosystem response among embayments.

From a management point of view, discrepancies among loading models are often not appreciable since management conclusions are robust if a consistent methodology is used. The BBP's methodology in 1991 was based on a particular nitrogen loading paradigm, and the empirical relationships between existing water and habitat quality and the presumed watershed loadings. If the Buzzards Bay Project had adopted a 50% watershed attenuation coefficient, the recommended nitrogen loading limits for embayments would have been half of those proposed because it would have taken only half the loading to cause the observed ecosystem changes. For scientists, it is critical to understand the exact pathways and loss terms for all nitrogen sources, but for managers and the public the question may be as simple as knowing how many septic system inputs must be eliminated to restore water quality. For example, in Waquoit Bay, eelgrass beds and shellfish habitat had appreciably declined by 1978. At that time there were approximately 1800 residential dwellings in the watershed together with other land uses. Depending on which nitrogen loading model is used, a different conclusion would be made as to how much nitrogen was causing these adverse effects. However, in a practical sense, the management objective becomes reducing loading by the equivalent of 1800 dwellings, with whatever loading model is used. In other words, if nitrogen loading standards are developed based on a certain set of assumptions, those same assumptions must be used by planning and regulatory agencies in their decision making process.

<u>Which nitrogen loading "scale" is most appropriate?</u>

In 1991 the Buzzards Bay Project believed that it was unlikely that any single nitrogen loading scale or limit on a loading scale was appropriate in all situations, and that both area-based and volumetric-turnover time scales needed to be considered for managing coastal embayments. The results of the Citizens Water Quality Monitoring Program formed an independent data set to evaluate which scale of loading measurement is most appropriate. Clearly the response of embayment water quality shows that hydraulic turnover time is a fundamental factor in correlating ecosystem response to anthropogenic loading, and accounting for flushing time in establishing loading limits appears well justified.

Although Figure 8 suggests that the BBP's 1991 ORW limit of 5 g m^{-2} y^{r-1} is in fact a transition point for ecosystem response, the limited data from Buzzards Bay embayments alone does not support that a limit based on area alone is any more important beyond 4.5 days than the Vollenweider scale. Other assessments like those of Nixon (1986) showing a correlation between loading on an aerial basis and water quality parameters were biased toward larger and deeper systems like Chesapeake Bay and Long Island Sound where parcels of water remain may remain out of the photic zone for prolong periods because of stratification. Even in the deepest embayments in Buzzards Bay, mixing is considerable so that light limitation is not important in phytoplankton production. As shown in Figure 2, the aerial loading scale will generally result in more "lenient" loading limits when water turnover times in the embayment are longer than 4.5 days for embayments with mean depths of 1 m, but for embayments of 2 m mean depth (closer to the depth of more poorly flushed Buzzards Bay embayments), the crossover point is 10 days. Seven years of monitoring has shown that the aerial loading scale does not correlate as well with water quality data as a scale incorporating flushing. From a management point in the few instances that the aerial scale is triggered in Buzzards Bay, the result is a somewhat more lenient allowable nitrogen load than required under the Vollenweider-term flushing equation, resulting in easier to achieve nitrogen loading goals. However, strict application of the tiered limit criteria has resulted in a more restrictive aerial scale, which was not the original intent of the managers. Because of this confusion and the lack of support from the water quality data, the areal scale for defining loading limits should be abandoned by managers in favor of a volumetric-Vollenweider flushing term scale of nitrogen loading.

While flushing time is an important forcing function of ecosystem response, we acknowledge that strict application of the Vollenweider approach in marine systems needs to be investigated further. For Buzzards Bay embayments, the Vollenweider term adjustment to flushing represents a very small reduction in relative nitrogen loadings [= $(1+\tau_w^{½})^{-1}$ where τ is in years] in rapidly flushed systems. For an embayment with a 3 day flushing time, the adjustment is only 9%. While estimated loadings adjusted for flushing time show a better correlation observed water quality than loadings adjusted for embayment area or embayment volume alone, the use of the modest Vollenweider term adjustment to flushing cannot be discerned from a simple flushing term without the Vollenweider expression. It use, however, does result in poorly flushed embayments being allowed more nitrogen than if a management loading limits were simply inversely proportional simple flushing (e.g., 20% higher at 15 days). Vollenweider noted that the relationship he defined did not extrapolate linearly to shallow lakes (<20 m) with turnover times less than 1 year. These characteristics are very different from the shallow well-flushed embayments we studied in Buzzards Bay. Vollenweider's approach required an estimate of phosphorus turnover time. Because calculating phosphorus turnover times in lakes is complex, Vollenweider argued that <u>phosphorus replacement time approximately equaled the total amount of P in a lake divided by the annual P loading rate</u>. Such an approach cannot be applied to a marine system without also accounting for tidal export of nitrogen. For example, we estimated that nitrogen loading to Buttermilk Bay currently is 25,197 kg yr^{-1}.

Assuming that the average total nitrogen concentration was 0.42 ppm (1992-1998 mean), the half-tide mass of nitrogen (volume x concentration) would be 1,500 kg. These numbers would suggest that the nitrogen replacement time for Buttermilk Bay is 1,260 kg / 24,000 kg yr^{-1}, or 19.1. However, the 0.42 ppm value is actually a steady state condition with tidal losses (0.42 ppm leaving) and tidal gains (0.27 ppm in offshore waters). The 0.15 ppm difference due to land-based loading would suggest that the half tide mass of nitrogen from land-based loading is really 450 kg, suggesting a nitrogen replacement time of 6.8 days.

In marine ecosystems, nitrogen losses are more complex than phosphorus losses in lakes. Not only is a portion of nitrogen buried (just as phosphorus is in lakes), but 40-50% may be denitrified and thus lost completely from the system (Seitzinger 1988; Jensen et al. 1988; Koike and Sorrenson 1988). Thus the above theoretical exercise in calculating nitrogen replacement time is an underestimate because not all added nitrogen remains in the water column in measurable form. For comparison nitrogen replacement time, in Buttermilk Bay, Valiela and Costa (1987) estimated a hydraulic replacement time of 5 days based on measurements of salinity and tidal prism volumes.

Phosphorus turnover time and hydraulic turnover in lakes often exceed a year and may be many years. The interdependence of these two variables suggested to Vollenweider that the ratio of mass phosphorus in the lake to hydraulic turnover time equaled the ratio of the P concentration in the lake to P in the inflow concentration. As illustrated in the example above, both the hydraulic turnover time and nitrogen replacement time can both be less than a week. Would the same principals apply?

Despite the lack of empirical supporting evidence, we have adopted the Vollenweider approach because of its theoretical framework and because the Vollenweider adjustment diminishes the role of flushing in estuaries with very long hydraulic turnover times. This may be valid because with longer residence times, burial in sediments and denitrification become relatively more important as a mechanism for removing anthropogenic loading. This latter point has important implications for managers since at a residence time of 20 days, about 23% more nitrogen would be “allowed” with this Vollenweider adjustment factor than if a simple flushing term alone were used.

Use of hydraulic turnover times

If hydraulic turnover times are used to establish nitrogen loading limits for embayments, there must be some consensus among managers as to what methodologies should be used since for short turnover times, acceptable loading rates are nearly inversely proportional to hydraulic turnover times. Unfortunately there is probably no one single methodology to estimate a turnover rate for nitrogen management because of the remarkably varied physical natures of estuaries. The distribution and concentration of watershed inputs and their potential impact, will depend upon the time needed to exchange freshwater and saltwater in receiving coastal systems. This exchange period is usually termed "turnover time,” "residence time,” or "flushing time" depending upon how the period is calculated or defined (Bowden 1967; Emery 1969; Isaji et al. 1985; Pilson 1985). Sometimes these terms are used interchangeably, but each has a specific definition. Zimmerman (1976) defines the mathematical basis for each of these terms, and these definitions are adopted here. Flushing time has been defined as the length of time necessary to replace the fresh water contained within an estuary, whereas turnover time defines the length of time necessary to remove 63% ($1-e^{-1}$) of marine water within an estuary. Geyer et al. (1997) similarly defined flushing time in dyes studies as the length of time necessary to reduce the mass of added dye by 63% ($1-e^{-1}$). Because turnover times for coastal embayments reflect the duration of conservative species dissolved or suspended in seawater, turnover time, in most instances, may be the most appropriate method to be used in adjusting annual nitrogen loading rates. Residence time, which is the average age of a water particle in an estuary, yields very similar values to turnover time, and the two methods are approximately equivalent at equilibrium conditions. However near the mouth of bays, or when turnover times are less than 2 days they may be substantially different (Zimmerman 1976). Consequently the use of mean residence time is less desirable than turnover time for assessing nitrogen loading rates for small rapidly flushed embayments.

Flushing time (sensu Ketchum 1951) is a freshwater budget approach that can be an appropriate measure of water exchange and hydraulic turnover to establish nitrogen loading limits in V-shaped estuaries that receive most freshwater and nitrogen via riverine discharge in the headwaters. This method is inappropriate in embayments that receive low freshwater inputs or receive most freshwater via groundwater because the necessary upstream gradients of salinity for model calculations are not observed. In these latter embayments, turnover time or residence time should be calculated. In bays where freshwater inputs are very large relative to seawater volumes, low salinity surface waters may leave before thoroughly mixing with the water in the embayment, although residence time of seawater can greatly affect patterns of fresh and seawater mixing (Cifuentes et al. 1990). Even where freshwater inputs are not especially large, embayments that are stratified may exhibit more rapid flushing of nitrogen because the fresh water (which contains more nitrogen) may be the first waters to exit on an outgoing tide (Garcon, et al. 1986). If less nitrogen of the nitrogen mixes in the receiving salt waters in salinity stratified systems, either different management loading limits or the use of freshwater residence times might be appropriate for these sites.

Because most of the adverse effects of nitrogen loading are observed in summer, mean summer water exchange rates should be used in nitrogen loading assessments for temperate waters. It is more important to characterize flushing during this period where freshwater inputs or prevailing winds show distinct seasonal differences because each may affect water exchange rates. Because most embayment ecosystem responses are integrated over weeks or longer, turnover time during a mean tidal cycle (rather than neap or spring tides) should be used.

The Buzzards Bay Project recommended that hydraulic turnover time for the upper third of an estuary, because it was felt that these areas were most likely to be affected by nitrogen loading. These turnover times (from early drafts of the ACI report) were used by the BBP and the tiered nitrogen loading limits shown in Table 5 and Figs. 5 and 6. We recognize that the estimates of hydraulic turnover time of Buzzards Bay embayments estimated by ACI (1991 1995) are rudimentary and represent considerable uncertainty in our analysis, and that more detailed models and field studies should be conducted before management action is adopted for most Buzzards Bay embayments. This process has already begun for Buzzards Bay. Managers and politicians must decide what level of effort (and money) should be expended to characterize water exchange, and ultimately must make decisions based on the best available information at hand. Because hydraulic turnover time is a crucial component of in establishing nitrogen embayment TMALs (allowable loading is nearly inversely proportional to flushing time), it is important that scientists and managers define criteria and conditions under which the various methodologies are employed.

<u>Did the 1991 recommended loading limits make sense?</u>

The results of the water quality monitoring program suggest that nitrogen loading limits based on flushing times and water volumes are scientifically defensible. The use of existing tiered water quality classifications to create tiers of allowable loading is merely a tool for environmental managers and regulators to make decisions that reflect existing conditions, uses, and management priorities. But were the specific limits proposed defensible?

The results of the citizen water quality monitoring program, especially broader measures such as total nitrogen concentrations and the Eutrophication Index (Figs. 13-16) suggest that coastal ecosystems were showing appreciable changes in water quality or eelgrass coverage with as little as 50 or 60 mg m^{-3} Vr^{-1}, or half of the BBP's recommended volumetric-flushing limit for Outstanding Resource Waters. In fact, eelgrass cover, which seemed to be the most sensitive of these indicators (as might be expected from other studies), declined with loadings as low as 40 to 50 mg m^{-3} Vr^{-1} (Fig. 17).

The fact that the Buzzards Bay Project recommended loading limits would not always be protective was also demonstrated by observations in Waquoit Bay where the near complete loss of eelgrass beds and the bays first anoxic event occurred at approximately 20% below the ORW limits recommended by the BBP in 1991, and significant losses occurred at around 40 mg m^{-3} Vr^{-1}. At the time it was felt that Waquoit Bay may have been "hypersensitive" to nitrogen loading. This sensitivity may be due to the fact that most of the bottom of the bay was near the photosynthetic compensation point for the growth of eelgrass so that slight declines in water transparency and light availability to eelgrass resulted in large scale losses of eelgrass as observed in that bay during the 1970's and 1980s (Costa et al. 1993). These eelgrass beds were replaced by unattached benthic algae that has accumulated in thick layers in some parts of the bay.

Another interpretation of Waquoit Bay's sensitivity may be the result of the fact that Waquoit Bay is interconnected to an adjoining embayment (Eel Pond) at its head, that is heavily loaded with nitrogen from coastal development, and a considerable portion of this nitrogen enters Waquoit Bay. Eel Pond is a much more heavily nitrogen loaded system with poorer water quality, and each day a major portion of the fresh water and nitrogen flow to Eel Pond actually flows into Waquoit Bay. Thus the Waquoit Bay and Eel Pond complex may fit the Buzzards Bay Project's model much better than Waquoit Bay alone. Using the loading assumptions presented here, Waquoit Bay was receiving 68 mg m^{-3} Vr^{-1} in 1978 and 99 mg m^{-3} $Vr^{-1,}$ loading rates consistent with embayments with similar ecosystem responses.

Science vs. Management

The TMALs recommended for Buzzards Bay in 1991 were a product of management needs and available scientific information. Embayment-specific limits were needed but it was too costly to develop ecosystem response models for a large number of embayments. Rather, an empirically derived relationship was defined between estimates of nitrogen loading a subjective evaluation of ecosystem response. In 1991 this was a novel approach, but with time, this management strategy has been increasingly used by Massachusetts state and regional agencies and municipalities for planning and permitting decisions. This tiered limits enabled managers to develop nitrogen loading rates specific to each embayment, gave planners and regulators an objective and consistent mechanism to manage nitrogen.

Based on the results of the monitoring program in Buzzards Bay, changes in the Buzzards Bay Project methodology and TMALs appear appropriate, and the BBP is proposing the simplified TMAL strategy in Table 11. When these total nitrogen or the Eutrophication Index scores (not shown) are compared to embayment loadings defined as a percentage of these new recommended TMALs for

Buzzards Bay embayments (Fig. 19), the results suggest that these new limits will be protective using the current loading assumptions.

In other regions, managers and scientists have also taken lar action to protect or restore coastal systems based on the best available scientific information. For example, a 50% nitrogen reduction goal has been established for the North Sea, despite difficulties in tracking ecosystem response (Lanne et al. 1990). Similarly, the states around Chesapeake Bay have adopted a goal of reducing nitrogen to 40% of 1987 levels (Jordan et al. 1990) to restore water quality and living resources in that estuary, and a 30% reduction goal has been adopted for Long Island Sound. This Chesapeake Bay remediation target was based on computer models that predicted the extent and duration of hypoxic conditions in the mainstem of the estuary under different nitrogen loading scenarios. The Long Island Sound goals were similarly based on a model to predict spacial coverage of low dissolved oxygen concentrations. In both cases, data and information on loading and ecosystem response continue to be gathered both to ascertain whether loading targets are appropriate and will result in the desired ecosystem benefit.

Elsewhere, attempts to manage nitrogen to coastal waters has remained a significant challenge to environmental managers and policy makers for a number of reasons. The relationship between nitrogen loading and ecosystem response is complex and no single model or loading approach can suit all situations. For Chesapeake Bay and Long Island Sound, considerable expense and manpower was needed to develop the embayment specific models. Even when there is a high certainty as to what is the acceptable loading rate is appropriate for an embayment, developing solutions to mange the cumulative impacts non-point sources such as septic systems and agricultural land through runoff and leaching to groundwater is difficult for all levels of government. Changes in coastal ecosystems and marine habitat are gradual, often spanning decades, making it difficult to quantify, and reducing public perceptions as to the magnitude of the problem until severe effects are observed such as fish kills. Equally problematic, the beneficial effects of management action will take years or decades to document and evaluate because of the slow transit time of nitrogen in groundwater. Thus, even if development in some embayments were to cease, actual nitrogen loading to the marine receiving waters will continue to rise for years. For nitrogen management to succeed in protecting or restoring near coastal waters, good estimates of nitrogen inputs to the receiving waters and reliable assessments of water quality and living resources are needed. Both these tasks are expensive and time consuming, and have been undertaken for few bays and estuaries.

The TMALs established for Buzzards Bay were meant to be a starting point until more sophisticated computer models and simulations were developed that could be tailored to the specific conditions in each embayment to establish more appropriate loading targets. However, these alternative approaches are still under development, and these models will generally require embayment specific water quality and biological monitoring to calibrate and validate. This process may take years and considerable funding. In the interim, managers can use the Buzzards Bay Project’s tiered system of loading limits, and the modifications suggested here as a starting point to establish goals for watershed management plans. For coastal managers, such nitrogen TMALs are needed immediately because many watersheds are already developed, and it is far more difficult and costly to remediate existing nitrogen than to manage future impacts of growth.

Acknowledgments
This paper evolved from a draft Buzzards Bay Project technical report first developed in 1991. Thanks to Ivan Valiela who reviewed early drafts of this report. Thanks also to Neil MacGaffey who compiled some of the original landuse coverage statistics within the embayments used in this study. We are indebted to the 100's of volunteers that participated in Buzzards Bay Citizen's Water Quality Monitoring Program coordinated by the Coalition for Buzzards Bay. The help of the present coordinator for this program, Tony Williams, is much appreciated. Thanks also to the Coalition for the use of their 1997 and 1998 data. Many thanks also to Dale Goerenger-Toner for coordinating and overseeing the laboratory analyses. Jim Kremer provided valuable guidance and comments on a 1997 draft of this manuscript. Portions of this work was funded through several US EPA Cooperative Agreements including CX812872, CE001510, X001514, X991504, X991540, X991540.

LITERATURE CITED

ACI (Aubrey Consulting Inc.). 1991. Determination of flushing rates and selected hydrographic features of selected Buzzards Bay embayments. EPA Contract No. 68-C8-0105 Battelle Work Assignment 2-132. September 13, 1991. 56 pp.

ACI (Aubrey Consulting Inc.). 1995. Estimation of flushing rates in selected Buzzards Bay embayments. January 1995. 46 pp.

Anderson, D. L., A. L. Lewis, and K. M. Sherman. 199x. Unsaturated Zone Monitoring below subsurface wastewater infiltration systems serving individual homes in Florida.

Benkema, J.J. and G.C. Cadee. 1986. Zoobenthos responses to eutrophication of the Dutch Wadden Sea. Ophelia 26:55-64.

Borum, J. 1985. Development of epiphytic communities on eelgrass (Zostera marina) along a nutrient gradient in a Danish estuary. Marine Biology 87:211-218.

Bowden, K. P. 1967. Circulation and diffusion. In Lauff, G. H. (ed.), Estuaries, Publication No. 83, AAAS, Washington,

Boynton, W. R. and J. H. Garber. 1988. Nutrient budgets for the Northern Chesapeake Bay and three tributary estuaries. EOS Transactions A. G. U. 69:1096.

Boynton, W. R., W. M. Kemp, and C. W. Keefe. 1982. A comparative analysis of nutrients and other factors influencing estuarine phytoplankton production. P. 69-90. In V. S. Kennedy (ed.), Estuarine Comparisons. Academic Press.

Brush, G. S. 1984. Stratigraphic evidence of eutrophication in an estuary. Water Research 20:531-541.

Buzzards Bay Project (U. S. Environmental Protection Agency and Massachusetts Executive Office of Environmental Affairs). 1991. Buzzards Bay Comprehensive Conservation and Management Plan, 8/91 Final. 246p.

Cadee, G. C. 1986. Increased phytoplankton primary production in the Marsdiep area (Dutch Wadden Sea). Netherlands Journal of Sea Research 20:285-290.

Caezan, M. L., E. M. Thurman and R. L. Smith. 1987. The role of cation exchange in the transport of ammonium and nitrate in a sewage contaminated aquifer. Toxic Waste and Groundwater Contamination Program, Third Technical Meeting, B53-B58.

Cederwall, H. and R. Elmgren. 1980. Biomass increase of benthic macrofauna demonstrates eutrophication of the Baltic Sea. Ophelia Supplement 1:287-304.

Cederwall, H. and R. Elmgren. 1990. Biological effects of eutrophication in the Baltic Sea, particularly the coastal zone. Ambio 19:109-112.

Chi et al. 1994

Cifuentes, L. A., L. E. Schemel, and J. H. Sharp. 1990. Qualitative and numerical analyses of the effects of river inflow variations on mixing diagrams in estuaries. Estuarine Coastal and Shelf Science 3):411-427.

Costa, J. E. 1988a. Distribution, production, and historical changes in abundance of eelgrass (Zostera marina L.) in Southeastern MA. Ph. D. Thesis, Boston University, 352 p.

Costa, J. E. 1988b. Eelgrass in Buzzards Bay: Distribution, production, and historical changes in abundance. U. S. Environmental Protection Agency Technical Report. EPA 503/4-88-002, 204 p.

Costa, J. E., B. L. Howes, A. Giblin, and I. Valiela. 1992. Monitoring Nitrogen and indicators of nitrogen to support management action in Buzzards Bay, p. 497-529. In McKenzie et al.(eds) Ecological Indicators, Elsevier, London.

Costa, J. E., D. Janik, N. MacGaffey, and D. Martin. 1994. Use of a geographic information system to estimate nitrogen loading to coastal watersheds. Buzzards Bay Project Technical Report. Draft, March 2, 1994, 21 p.

Costa, J. E., B. Howes, and E. Gunn. 1996. Use of a geographic information system to estimate nitrogen loading to coastal watersheds. Buzzards Bay Project Technical Report. Draft, March 2, 1994, 21 p.

Crites, R. and G. Tchobanoglous. 1998. Small and decentralized wastewater management systems. McGraw-Hill, Boston.

D'Avanzo, C. and J. N. Kremer. 1994. Diel oxygen dynamics and anoxic events in an eutrophic estuary of Waquoit Bay, Massachusetts. Estuaries 17:131-139.

D'Elia, C. F., J. G. Sanders, and W. R. Boynton. 1986. Nutrient enrichment studies in a coastal plain estuary: Phytoplankton growth in large-scale, continuous cultures. Canadian Journal of Fisheries and Aquatic Science 43:397-406.

DEP (Massachusetts Department of Environmental Protection Division of Water Pollution Control). 1989a. Water quality survey data, 1985. Buzzards Bay Project Technical Report, BBP-89-14, 147 p.

DEP (Massachusetts Department of Environmental Protection, Division of Water Pollution Control). 1989b. Water quality survey data, 1986. Buzzards Bay Project Technical Report, BBP-89-16, 104 p.

DEP (Massachusetts Department of Environmental Protection, Division of Water Pollution Control). 1991. State Surface Water Quality Standards.

Emery, K. O. 1969. A coastal pond studied by oceanographic methods. American Elsevier.

EPA (U. S. Environmental Protection Agency). 1980. Design Manual for Onsite Wastewater Treatment and Disposal Systems. EPA 625/1-80-012. 412 p.

EPA (U. S. Environmental Protection Agency). 1992. Wastewater treatment disposal for small communities. EPA/625/R-92/-005. xxx p.

Frimpter, M. H., J. Donohue, and M. V. Rapacz. 1988. A mass balance nitrate model for predicting the effects of land-use on groundwater quality in municipal wellhead protection areas. Cape Cod Aquifer Management Project.

Garcon, V.C., K. D. Stolzenbach, and D. M. Anderson. Tidal flushing of an estuarine embayment subject to recurrent dinoflagellate blooms. Estuaries 9:179-187.

Geyer, W. R., P. Dragos, and T. Donoghue. 1997. Flushing study of three Buzzards Bay harbors: Onset Bay, Little Bay, and Allen's Pond. Prepared for the Buzzards Bay Project. April 15, 1997. 39 pp.

Giblin, A. E, and A. G. Gaines. 1990. Nitrogen inputs to a marine embayment: the importance of groundwater. Biogeochemistry 10:309-328.

Gilliam, J. W., R. B. Daniels, and J. F. Lutz. 1974. Nitrogen content of shallow groundwater in the North Carolina coastal plain. Journal Environmental Quality 3:147-151.

Godfrey, P. J. 1988. Acid rain in Massachusetts. Water Resources Research Center, University of Massachusetts, 54 pp.

Hagedorn, C., E. L. McCoy and T. M. Rahe. 1981. The potential for groundwater contamination from septic effluents. Journal of Environmental Quality 10:1-8.

Hampson, G. P. 1988. Ground truth verification of a REMOTS survey of Buzzards Bay. Buzzards Bay Project Draft Report.

Harkins, R. D. An objective water quality index. Journal of the Water Pollution Control Federation 46:588-5.

Heil, G. W., M. J. A. Werger, W. De Mol, D. Van Dam and B. Heijne. 1988. Capture of atmospheric ammonium by grassland canopies. Science 239:764-765.

Horsley Witten Hegeman, Inc. 1991. Quantification and control of nitrogen inputs to Buttermilk Bay. Volume 1. Buzzards Bay Project Technical Report. January 1991. 66 p.

Howes, B. L. and C. D. Taylor. 1989. Nutrient regime of New Bedford Outer Harbor: Infaunal community structure and the significance of sediment nutrient regeneration to water column nutrient loading. Final Report, New Bedford Sewage Treatment Facilities Plan. Volume IV, Appendix R.

Howes, B. L. and D. D. Goehrenger. 1993. Water quality monitoring of Falmouth's coastal ponds: Report from the 1992 season. Technical Report Woods Hole Sea Grant Program. 89 pp.

Isaji, T., M. L. Spaulding and J. Stace. 1985. Tidal exchange between a coastal lagoon and offshore waters. Estuaries 8:203-216.

Jaworski, N. A. 1981. Sources of nutrients and the scale of eutrophication problems in estuaries, p. 83-110. In B. J. Neilson and L. E. Cronin (eds.), Estuaries and nutrients. Humana Press. Clifton, NJ.

Jensen, M. H., T. K. Anderson, and J. Sorensen. 1988. Denitrification in coastal bay sediments: regional and seasonal variation in Aarhus Bight, Denmark. Marine Ecology Progress Series 48:155-162.

Johansson, J. O. R. and R. R. Lewis. 1990. Recent improvements of water quality and biological indicators in Hillsborough Bay, a highly impacted subdivision of Tampa Bay, Florida, U. S. A. Proceedings of the International conference on marine coastal eutrophication, Bologna Italy, March 21-24, 1990.

Jordan, S, K. Mountford, C. Stenger, R. Batiuk, M. Olson, D. Forsel and L. Platt, 1990. Chesapeake Bay dissolved oxygen restoration goals. October 1990 Draft report, Chesapeake Bay Program, CBP/TRS 53/90.

Kautsky, N., H. Kautsky, U. Kautsky, and M. Waern. 1986. Decreased depth penetration of Fucus vesiculosus (L.) since the 1940's indicates eutrophication of the Baltic Sea. Marine Ecology Progress Series 28:1-8.

Kelly, J. T., D. J. Hersh, and I. Valiela. 1991. Nitrogen pollution in Buzzards Bay. Buzzards Bay Project Report.

Kemp, W. M., W. R. Boynton, R. R. Twilley, J. C. Stevenson, and J. C. Means. 1983. The decline of submerged vascular plants in Upper Chesapeake Bay: Summary of results concerning possible causes. Marine Technology Society Journal 17:78-89.

Ketchum, B. H. 1951. The exchanges of fresh and salt waters in tidal estuaries. J. Mar. Res. 10:18-38.

Kimmel, G. E. 1984. Non-point contamination of groundwater on Long Island, N. Y., p. 120-126. In National Academy Science, Groundwater Contamination.

Kristiansen, R. 1981. Sand-filter trenches for purification of septic tank effluent: II. The fate of nitrogen. Journal Environmental Quality 10:358-361.

Koike, I., and J. Sorensen. 1988. Nitrate reduction and denitrification in marine sediments. In Blackburn, T. H., and J. Sorensen (eds.) Nitrogen cycling in coastal marine environments. John Wiley & Sons.

Laak, R. 1982. On-site soil systems, nitrogen removal, p.129-143. In A. S. Eikum and R. W. Seabloom (eds.), Alternative Wastewater treatment. D. Reidel Publishing Co.

Laak R. and Coates. 1992.

Lanne, R. W. P. M., J van Der Meer, A. DeVries, and A van Der Gressen. 1990. Monitoring the progress of attempts to reduce nutrient load and input of certain compounds in the North Sea by 50%. Environmental Management 14:221-227.

Lapointe, B. E. and J. O'Connel. 1989. Nutrient-enhanced growth of Cladophora prolifera in Harrington

Sound, Bermuda: Eutrophication of a confined, phosphorus limited marine ecosystem. Estuarine and Coastal Research Shelf Science 28:347-360.

Lee, V., and S. Olsen 1985. Eutrophication and management initiatives for the control of nutrient inputs to Rhode Island coastal lagoons. Estuaries 8:191-202.

LISS (Long Island Sound Study). 1993. Comprehensive Conservation and Management Plan. 1099p. Draft January 1993. EPA 902-D-93-001.

Magnien, R. E., K. G. Sellner, P. A. Vaas. 1988. Nutrient control of phytoplankton production in the Chesapeake Bay mainstream an tributaries. EOS Transactions A. G. U. 69:1097.

McCarthy, J. J, J. L. Taft, L. E. Cronin, M. A. Tyler and W. B. Boynton. 1988. Chesapeake Bay anoxia: Origin, development and significance. Science 223:22-27

Nixon, S., and M. Pilson. 1983. Nitrogen in estuarine and coastal marine ecosystems, p. 565-648. In E. J. Carpenter and D. G. Capone (eds.), Nitrogen in the Marine Environment. Academic Press, NY.

Nixon, S. W. 1983. Estuarine ecology - A comparative and experimental analysis using 14 estuaries and the MERL microcosms. EPA Chesapeake Bay Program. 59 p.

Nixon, S. W., C. A. Oviatt, J. Frithsen, and B. Sullivan. 1986. Nutrients and the productivity of estuarine an/d coastal marine ecosystems. Journal Limnological Society of South Africa 12:43-71.

Orth, R. J. and K. A. Moore. 1983. Chesapeake Bay: An unprecedented decline in submerged aquatic vegetation. Science 222:51-53.

Parkin, T. B. 1987. Soil microsites as a source of denitrification variability. Soil Science Society American Journal 51:1492-1501.

Pearl, H. W. 1985. Enhancement of marine primary production by nitrogen enriched acid rain. Nature 315:747-749.

Persky, J. H. 1986. The relation of ground-water quality to housing density, Cape Cod, Massachusetts. U. S. G. S. Water-Resources Investigations Report 86-4093. 28 p.

Peterjohn, W. T. and D. L. Correll. 1984. Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. Ecology 65:1466-1475.

Petrovic, A. M. 1990. The fate of nitrogenous fertilizers applied to turfgrass. Journal of Environmental Quality 19:1-14.

Pilson, M. E. 1985. On the residence time of water in Narragansett Bay. Estuaries 8:2-14.

Reneau, P. B. 1979. Changes in concentrations of selected chemical pollutants in wet, tile drained soil systems as influenced by disposal of septic tank effluents. Journal Environmental Quality 8:189-196.

Rhoades, D. 1988

Rosenberg, R. 1985. Eutrophication - the future marine coastal nuisance. Marine Pollution Bulletin 16:227-2311.

Ryther, J. H., and W. M. Dunstan. 1971. Nitrogen, phosphorus, and eutrophication in the coastal marine environment. Science 171:1008-1031.

Werme, C. 1991. Characterization of pollutant inputs to Buzzards Bay. Buzzards Bay Project Technical Series, Draft report, May 1991.

Sanders, J. G., S. V. Cibik, C. F. D'Elia, and W. R. Boynton. 1987. Nutrient enrichment studies in a coastal plain estuary: changes in phytoplankton species composition. Canadian Journal of Fisheries and Aquatic Science 44:83-90.

Schindler, W. H. and A. E. Hartley. 1992. The biosphere as an increasing sink for atmospheric carbon: estimates from increased nitrogen deposition. Global Biogeochemistry 15: 191-211.

SCS (USDA Soil Conservation Service). 1992. Agricultural Waste Management Field Handbook. (210-AWMFH, 4/92) 150 p.

Seitzinger, S. P. 1988. Denitrification in freshwater and coastal marine ecosystems: ecological and geochemical significance. Limnology and Oceanography 33:702-724.

Seliger, H. H., J. Bugg and W. H. Biggley. 1985. Catastrophic anoxia in the Chesapeake Bay in 1984. Science 228:70-73.

Shaum 1994

Sikora, L. J. and R. B. Corey. 1976. Fate of nitrogen and phosphorus in soils under septic tank waste disposal fields. Transactions of the American Society of Engineers?. xx:866-875

Smith, R.L., B. L. Howes. and J. H. Duff. 1991. Denitrification in nitrate contaminated groundwater: Occurrence in steep vertical geochemical gradients. Geochimie Cosmochim Acta 55:1815-1825.

Stensland, G. J., D. M. Whelpdale, and G. Oehlert. 1986. Precipitation Chemistry. In Acid Deposition Long-Term trends. National Research Council, National Academy Press, pp 128-185.

Taylor, C. D. and B. L. Howes. 1994. Effects of sampling frequency on measurements of seasonal primary production and oxygen status in near-shore coastal ecosystems. Marine Ecology Progress Series.

Trudell, M. R., R. W. Gillham, and J. H. Cherry. 1986. An in-situ study of the occurrence and rate of denitrification in a shallow unconfined sand aquifer. Journal of Hydrology 83:251-268.

Twilley, R. R., W. M. Kemp, K. W. Staver, J. C. Stevenson, and W. R. Boynton. 1985. Nutrient enrichment of estuarine submersed vascular plant communities. 1. Algal growth and effects on production of plants and associated communities. Marine Ecology Progress Series 23:179-191.

Valiela, I. 1984. Marine Ecological Processes. Springer-Verlag, New York. NY.

Valiela, I. and J. Costa. 1988. Eutrophication of Buttermilk Bay, a Cape Cod coastal embayment: Concentrations of nutrients and watershed nutrient budgets. Environmental Management 12:539-551.

Valiela, I., J. Costa, K. Foreman, J. M. Teal, B. Howes, and D. Aubrey. 1990. Transport of groundwater-borne nutrients from watersheds and their effects on coastal waters. Biogeochemistry 10:177-197.

Vollenweider, R. A. 1976. Advances in defining critical loading levels for phosphorus in lake eutrophication. Memoirs 1st Italian Idrobiologie. 33:53-84.

Vollenweider, R. A. 1985. Elemental and biochemical composition of plankton biomass; some comments and explorations. Archives of Hydrobiology 105:11-29.

Walker, W. G., J. Bouma, D. R. Keeney and P. G. Olcott. 1973. Nitrogen transformations during subsurface disposal of septic tank effluent in sands: II. Groundwater quality. Journal Environmental Quality 2:521-525.

Weiskel, P. K. and B. L. Howes. 1991. Quantifying dissolved nitrogen flux through a coastal watershed. Water Resources Research 27:2929-2939.

Weiskel, P. K. and B. L. Howes. 1992. Differential transport of sewage-derived nitrogen and phosphorus through a coastal watershed. Environmental Science and Technology 26:352-360.

Whelan, B. R. and Z. V. Titamis. 1982. Daily chemical variability of septic tank effluent. Water, Air, and Soil Pollution 17:131-139.

Wilhelm, S.R, Schiff, S.L. and J. A. Cherry. 1994. Biogeochemical evolution of domestic waste water in septic systems: 1. Conceptual Model. Ground Water 32:905-916.

Zimmerman, J. T. F. 1976. Mixing and flushing of tidal embayments in the western Dutch Wadden Sea

Part I. Distribution of salinity and calculation of mixing time scales. Netherlands Journal of Sea Research 10:149-191.

Table 1. Simplified sequence of ecosystem response typical of coastal embayments used to develop BBP loading limits in 1991.

Pristine: "Normal" ecosystem conditions without anthropogenic N loading.

Stage 1 (nominal response): Localized eelgrass losses and macroalgae accumulation near major entry points of nitrogen, alterations in Benthic community structure, increase in denitrification rates, chlorophyll concentrations may become somewhat elevated in portions of the estuary.

Stage 2 (moderate response): Onset of system-wide eelgrass loss, and macroalgae accumulation within photic zone. Progression of benthic community response which may include further increase in benthic biomass. Chlorophyll concentrations elevated, especially in embayments not dominated by macroalgae production. Denitrification as a % of N loading begins to decrease.

Stage 3 (severe response): Loss of most eelgrass. Macroalgae may dominate benthos in shallower systems resulting in widespread loss of shellfish and other typical macroinvertebrates. Further decrease in denitrification as percentage of loading. Benthic biomass may begin to decline. Chlorophyll concentrations, total nitrogen appreciably higher in systems not dominated by macroalgae. Water transparency considerably reduced.

Stage 4 (catastrophic response): Occasional diel hypoxia or anoxia in shallow embayments, possible prolonged bottom hypoxia in deeper stratified systems. Collapse of macrobenthic community. Sediments have appearance and consistency of "black mayonnaise". Summertime DIN may remain elevated (above 10 μM) and N:P well above Redfield ratio indicating nitrogen may no longer be limiting.

Table 2. Sources of nitrogen to Buzzards Bay and Buttermilk Bay, a well studied Buzzards Bay embayment. About 25% of Buttermilk Bay homes are sewered.

	Buzzards Bay[a]		Buttermilk Bay[b]	
Source	(mt N y^{-1})	(% of total)	(mt N y^{-1})	(%of total)
Precipitation				
- runoff from developed land	37	2	2.4	10
- directly on Bay	217	12	1.6	7
Sewage treatment facilities (includes CSOs)	1210	62	0.0	0
Septic systems	276	15	11.2	46
Fertilizer				
- on lawns	70	4	4.3	19
- agricultural use	76	5	4.4	19
Other	360		nil	
Total	2246		24.3	

[a] Based on Kelly et al. (1993) and Werme et al. (1993).

[b] Based on Valiela and Costa (1988), Horsley, Witten and Hegeman, Inc. (1991), and revised BBP sewering and actual occupancies.

Table 3. Four scales for evaluating nitrogen loading expressed in common terms to assist in comparisons.

N loading scale	loading units	comments
1) L_A	mg N $\cdot$ m^{-2} y^{-1}	areal scale
2) $\dfrac{L_A}{z}$	mg N $\cdot$ m^{-3} y^{-1}	volumetric scale [a]
3) $\dfrac{L_A \cdot \tau_w}{z}$	mg N $\cdot$ m^{-3} during τ_w	volumetric-turnover time scale [b] (i.e. loading during turnover time)
4) $\dfrac{L_A \cdot V_r}{z}$ Or $\dfrac{L_A \cdot \tau_w}{z \cdot (1+\tau_w^{½})}$	mg N $\cdot$ m^{-3} during V_r	volumetric-Vollenweider term adjusted turnover time scale [c] (i.e. loading during turnover time adjusted for the Vollenweider term.)

[a] z is the mean depth of the embayment in meters

[b] τ_w is the turnover time of the receiving waters in years

[c] Vollenweider defined a critical phosphorus loading limit (Lc) with the equation $Lc = k_3 \bullet q_s \bullet (1+(z/q_s)^{½})$, where q_s is "hydraulic load" of the lake, and k_3 is an empirically derived constant in the range of 10-20 which based on spring overturn P concentrations (in mg/m$^{)3}$) observed in eutrophic freshwater systems. Because Vollenweider defined hydraulic load $q_s = z/\tau_w$, his equation can be re-written Lc (as mg m^{-2} y^{-1})= $k \cdot z/\tau_w \cdot (1+\tau_w^{½})$. Since Vollenweider' critical limit for lake (Lc) was defined in terms equivalent to the term L_A above, and dropping the constant k or in general terms Vollenweider's equation is $La = z/\tau_w \cdot (1+\tau_w^{½})$. Moving all the terms to one side, $1= (L_A \cdot \tau_w)/(z \cdot (1+\tau_w^{½}))$, showing that Vollenweider's assessment of loading is equivalent to loading to the receiving waters during the hydraulic turnover time divided by the term $(1+\tau_w^{½})$. We refer to $(1+\tau_w^{½})$ as the Vollenweider-term and this loading scale is expressed as "mg m^{-3} during the Vollenweider-term adjusted turnover time (V_r)".

Table 4. Loading rates of some large temperate embayments by unit area, volume, volume during hydraulic turnover time, and volume during Vollenweider term adjusted hydraulic turnover time. Choice of loading scale affects relative ranking. Data modified from Valiela and Costa (1988).

	Loading Scales					
Embayment	g m^{-2} yr^{-1}	mg m^{-3} yr^{-1}	turnover times (d)	mg m^{-3} r^{-1}	mg m^{-3} V_r^{-1}	Comments
Buzzards Bay	3.6	428	9.1	11.7	10.1	No bay scale anoxia
Narragansett Bay	13.3	1400	26	100	79	No bay scale anoxia
Chesapeake Bay	8.2	902	56	138	99	Central bay anoxic events
Long Island Sound	5.6	420	166	191	114	Central bay anoxic events
South San Francisco Bay	22.4	4340	320	3805	1965	Bay-wide anoxic events

Table 5. Nitrogen loading rate limits to coastal waters recommended for Buzzards Bay embayments in the Buzzards Bay Comprehensive Conservation and Management Plan (Buzzards Bay Project, 1991) based on either area of the embayment or flushing and volume as specified by the depth and flushing time criteria[a].

Embayment type	SB Waters[b]	SA Waters[b]	Outstanding Resource Waters[b]
Shallow[c]			
-flushing: ≤4. 5 days	350 mg m^{-3} Vr^{-1}	200 mg m^{-3} Vr^{-1}	100 mg m^{-3} Vr^{-1}
-flushing: >4. 5 days	30 g m^{-2} yr^{-1}	15 g m^{-2} y^{r-1}	5 g m^{-2} yr^{-1}
Deep			
-lesser of	500 mg m^{-3} Vr^{-1} or 45 g m^{-2} y^{r-1}	260 mg m^{-3} Vr^{-1} or 20 g m^{-2} y^{r-1}	130 mg m^{-3} Vr^{-1} or 10 g m^{-2} yr^{-1}

[a] Vr = Vollenweider flushing term, $=\tau_w/(l+\mathrm{sqrt}(\tau_w))$ where τ_w is the hydraulic turnover time of the receiving waters in years.

[b] SA, SB, and ORW are Massachusetts water quality classifications contained in Massachusetts Surface Water Quality Standards (DEP, 1991). Outstanding Resource Waters are a special designation within the Surface Water Quality Standards that can be applied to SA waters under the Anti-degradation Provision of the federal Clean Water Act.

[c] Shallow is defined as having a mean depth of 2 m or less, or having 40% or more of area less than 1 m.

Table 6. Nitrogen loading assumptions used by the Buzzards Bay Project for characterizations and management purposes. The loadings shown are the contributions eventually expected to reach coastal waters after transit through the watershed.

Specific N loading source		**units and rates**
Septic systems	2.7	kg yr^{-1} $capita^{-1}$
Occupancy rate (area average)	3.0	persons per residential unit; use actual census data [a]
Lawns	29.3	kg yr^{-1} per hectare (1.4 kg yr^{-1} per typical lawn)
Precipitation	1.19	m yr^{-1}
Road surface runoff	15.3	kg ha^{-1} yr^{-1} [b]
Roof, other impervious runoff	7.3	kg ha^{-1} yr^{-1} [c]
Natural landscapes	0.0	kg ha^{-1} yr^{-1} (or 0.42 kg ha^{-1} yr^{-1})
Precipitation to bay	7.1	kg ha^{-1} yr^{-1} [d]
Dairy Cows	75.0	kg animal $unit^{-1}$ yr^{-1} (454 kg of animal)
Mass GIS Land use statistics 1:25,000 coverages:		
1: Cropland (corn, nurseries)	20.0	kg ha^{-1} yr^{-1}
2: Pasture (hay, dairy)	10.0	“
3: Forest	0.0	“
4: Non-forested wetland (freshwater marshes)	0.0	“
5: Mining (sand and gravel pits)	7.3	“
6: Open land(includes cleared land)	0.0	“
7: Participatory recreation (incl. golf courses)	29.3	“
8: Spectator recreation (incl. baseball fields)	29.3	“
9: Water Based recreation (incl. Beaches)	0.0	“
10:R0-Residential-multi-family[a] (condominiums, dormitories, apartment buildings)	106.5	“, =12.36 units ha^{-1}, 8.62 kg $unit^{-1}$, occupancy=3
11: R1-Residential-<¼ acre lots[a]	82.6	“=9.27 units ha^{-1}, 9.03 kg $unit^{-1}$, occupancy =3
12: R2-Residential-¼ - ½ acre lots[a]	46.4	“=5.41 units ha^{-1}, 9.57 kg $unit^{-1}$, occupancy=3
13: R3:Residential->½ acre lots[a]	23.2	“=2.57 units ha^{-1}, 9.57 kg $unit^{-1}$, occupancy =3
14: Salt marsh	0	“
15: Commercial (business districts, unsewered)	121	“
15: Commercial (business districts, sewered)	15.8	“
16: Industrial	15.8	“
17: Urban open (parks)	0	“
18: Transportation (interstate highways)	15.8	“
19: Waste disposal (incl. landfills)	15.8	“
20: Water (freshwater ponds and rivers)	0	“
21: Woody perennial (cranberry bogs, orchards) (If bog production surface used alone, then 24.7 kg ha^{-1} yr^{-1})	17.6	“revised

[a] For residential land use, actual occupancy rates from federal or local census statistics were used to characterize existing loadings rather than the examples cited. These occupancy rates included unoccupied homes and were annualized by include seasonal rentals, and typically ranged between 2.0 and 2.7 for most Buzzards Bay communities. To characterize potential buildout conditions for management assessments, an occupancy rate of 3.0 was used to account for trends in seasonal to year round dwelling conversion occurring in many surrounding municipalities.

[b] Assumes 90% recharge volume, and DIN = 1.5 ppm.

[c] Same as note c, but DIN = 0.75 ppm.

[d] 100% recharge, DIN = 0.75 ppm.

Table 8. Reported or calculated nitrogen loadings for septic systems and system components. Important study assumptions or approaches in brackets [], loading estimates per system components.

Component reference; (study type[1])	per capita flow ($l \cdot d^{-1}$)	effluent concentration ($mg \cdot l^{-1}$)	per capita N ($kg \cdot yr^{-1}$)	component loss
Household effluent				
EPA (1980, 1992); (L)	167	35-100, x= 61	3.9	0%
[values without garbage grinders]				
Laak (1982); ((L)		40	(4.4)	
Walker et al. (1973); (E)			8.2	
Sikora and Corey (1976); (L)		40-80		
Valiela et al. (1997) (L)	188	70	4.8	
Septic Tank effluent				
Andreoli, 1980; (L)				10%-20%
Canter and Knox, 1985; (E)		40-60		
EPA 1980; (L)		45, 36		
[means of 2 studies, 26 sites total]				
Magdoff et al. , 1974; (E)		40-50, x=43		
Hardisty, 1974 in Laak, 1982; (E)				10%
Piluk, 19 ;E)		69.4		
Whelan and Titus, 1985; (E)			4.6	
Wilhelm et al. , 1994; (E)	274	40	4. 0	33%
[high water use; leaching field in glacial soils with high carbonate, 2 m separation to groundwater]				
Costa and Howes, study in progress	141	68.6	3.1	
[per capita N represents mean of flow x concentration during study]				
Leaching field effluent				
Magdoff et al. , 1974; (E)		25-35		2% aerobic 32% anaerobic
[septic effluent , 8 cm d^{-1} dosage on mound system columns, above and below anaerobic layer]				
Kristiansen, 1981; (E)				nil to 11%
[accounted for NH_4^+ absorption on soils]				
Lamb et al. , 1978; (E)		49		1-6%
[comparative system design study]				
Alhajjar, 198 ;E)				
Assumed overall performance				
Cape Cod Commission, 1993; (A)	212	35	2.7	31%[2]
Buzzards Bay Project, 1993; (A)	NA	NA	2.7	31%[2]
Weiskell and Howes, 1991; (M)	~168	NA	3.0	30%[2]
[water use model, total water use =198 lpcd, assumed 15% for outside use (lawns, etc.)]				
Nelson et al. 1990; (M)	NA	NA	3.2	18%[2]
Gold et al. 1990; (E)	128	68	3.2	21%
[flows assumed from water surveys, lysimeter data, 4.0 kg pc load]				
Valiela et al. (1997) (L)	188	42.3	2.9	41%
[additional loss in plume, combined short term=]		1.9	60%	

[1] L=literature review, E= Experimental study, direct measurements; M=Mass loading model for recharge area; A=adopted for planning or regulatory purposes
[2] Based on an assumed 3.9 kg per capita annual load from EPA (1980 and 1992).

Table 7. Summary of embayment hydrological features and watershed loadings used in this report[1].

Figure Label	BUZZARDS BAY EMBAYMENT 09/20/99	Water area km2	Depth MLW (m)	Depth HTL (m)	Vol HTL $m^3 x 10^6$	Tidal Range (m)	Prism $m^3 x 10^6$	Prism % of MW Volume	selected turnover Time (d)	<2 m depth ha"	Basin land area(km^2)	1990 C Basin units	1990 C Basin Popul.	basin occupancy	Existing N Load/ (kg yr^{-1})	with prism loading	with forest 5uM loading
NBH	New Bedford Harbor (Acushnet River)	4.27	3.2	3.8	16.08	1.2	4.95	31	26.5	75	69.5				173521	224110	174926
ALL	Allens Pond	0.77	0.5	1.1	0.82	1.1	0.87	106	3.0	77	9.1	66	147	2.2	7088	15969	7237
APP	Apponanset,inner1990	1.54	0.7	1.3	1.95	1.1	1.73	89	7.4	87	21.6	4516	11380	2.5	27866	45547	28245
AUC	Aucoot Cove	1.29	2.2	2.9	3.69	1.3	1.65	45	0.4	35	10.5	327	629	1.9	7578	24441	7924
BIC	Brant Island Cove	0.34	0.8	1.4	0.47	1.1	0.39	83	0.7	33	1.7	72	154	2.2	662	4628	704
BMR	Broad Marsh River														88364	119433	92379
AGA	Agawam River	0.47	0.5	0.8	0.40	0.7	0.33		5.8						46439	49773	47545
BUT	Buttermilk Bay (combined)	2.17	1.2	1.7	3.71	1.0	2.32	63	3.4	165	26.0	3314	5575	1.7	24077	47787	24948
CLA	Clarks Cove	2.86	3.6	4.1	11.77	1.1	3.14	27	1.3	43	7.6				35078	67169	35139
EPB	Eel Pond, Bourne														ERR	ERR	ERR
EPM	Eel Pond, Mattapoisett	0.10	0.7	1	0.10	0.6	0.06	60	1.0	5					4949	5545	ERR
HEN	Hen Cove	0.26	0.8	1.5	0.37	1.2	0.31	84	0.1	21	4.4	596	966	1.6	5889	9088	6070
LIB	Little Bay Fairhaven	0.74		1.3	0.98	1.1	0.83	84	1.5						15281		
LIR	Little River	0.5	0.4	1.0	0.40	1.1	0.40	100	0.1	16	5.3	92	228	2.5	2556	10732	2685
MRK	Marks Cove	0.46	0.8	1.4	0.62	1.2	0.52	84	0.6	14	1.5	528	708	1.3	1801	16546	1830
MAT	Mattapoisett upper Harbor	1.59	2.8	3.4	5.47	1.2	1.89	35	1.8	66	NA				44954	64270	47581
MEG	Megansett Harbor	1.7	4.6	5.2	8.84	1.2	2.04	23	1.6	68	5.2				8412	29261	8552
NAS	Inner Nasketucket Bay	2.05	1.6	2.1	4.41	1.1	2.31	52	1.6	208	14.2	895	3176	3.5	35770	59378	36019
ONS	Onset Bay	2.39	1.3	1.8	4.33	1.0	2.48	57	3.9	121	12.6	2650	3846	1.5	22363	47709	22672
PHI	Phinneys Harbor	2.17	2.0	2.6	5.68	1.2	2.65	47	1.8	55	9.5		2251	ERR	12927	40010	13191
POH	Pocasset Harbor	1	6.0	2	2.00		1.00		4.0	30	3.7				6761	16981	6831
POR	Pocasset River	0.80	0.9	1.5	1.08	1.2	0.67	62	0.0	76	8.6		1538	ERR	8696	15543	8963
QUI	Quisset Harbor	0.47	1.6	2.2	1.02	1.2	0.56	55	0.4	17	2.1	157	177	1.1	2494	8227	2537
RBH	Red Brook Harbor	0.61	1.7	2.4	1.43	1.2	0.74	52	4.5	28	10.3	291	439	1.5	5575	13148	5920
SIP	Sippican Harbor upper harbor	1.70	1.4	2.0	3.47	1.2	2.03	59	12.3	114	6.1	663	1668	2.5	10500	31247	10635
SLO	Slocums River	1.97	0.7	1.3	2.53	1.1	2.16	85	11.3	106	95.7				92450	114526	95161
SQU	Squeteague Harbor	0.30	0.8	1.4	0.42	1.2	0.36	85	0.6	20	9.5	577	1245	2.2	7073	10762	7386
WAR	Wareham River	2.49	1.0	1.6	3.92	1.2	3.04	78	5.8	152	114.4	5200	10009	1.9	88364	119433	92379
WEE	Weweantic River	2.38	1.1	3.5	3.29	1.2	1.14	35	4.1	191	217.3	7539	19671	2.6	149126	160777	155582
WFH	West Falmouth Harbor	0.80	0.6	1.2	0.93	1.2	0.93	100	2.4	58	9.0	1180	1446	1.2	15154	24679	15390
WID	Widows Cove	0.54	0.9	1.5	0.83	1.2	0.65	78	0.8	37	1.4	23	38	1.7	626	7218	674
WIL	Wild Harbor	0.49	1.2	1.8	0.87	1.2	0.59	68	0.4	10	2.6	634	546	0.9	10811	16810	11078
WNG	Wings Cove	0.88	1.4	2.0	1.76	1.2	1.07	61	2.0	30	3.3	190	302	1.6	1993	12928	2097
WRE	Westport River, East Branch	8.02	0.8	1.0	8.17	0.5	3.91	48	49.7	740	154.8	4840	15136	3.1	133408	173368	138199
WRW	Westport River, West Branch	5.32	0.8	1.2	6.36	0.8	4.38	69	16.2	327	44.0	679	1163	1.7	33047	77811	34413
WAQ90	Waquoit Bay (1990 GIS)	6.56	0.9	1.5	6.02	1.0	0.23		5.0	286	49.44			ERR	49318		50628
WQE-90	Waquoit Bay-Eel Pond Complex	7.43	0.9		6.69	1.0			5.0	373							

[1]General notes to Table 7: Mnemonic code in column 1 is used in selected figures in this report. Abbreviations in table column headers as follows: MLW=Mean Low Water, HT= Half Tide conditions, Cen. =US Census Bureau, Acushnet River is synonymous with New Bedford Inner Harbor.

Table 9. Parameter Scale endpoints currently employed by the Buzzards Bay Project for the Buzzards Bay Eutrophication Index. End Point values have been modified somewhat since inception.

Parameter	0 point value		100 point value	
Oxygen saturation (mean of lowest 33%)	40	%	90	%
Transparency	0.6	m	3.0	m
Chlorophyll	10.0	µg/l	3.0	µg/l
DIN	10.0	µM	1.0	µM
Organic N	0.60	ppm	0.28	ppm

Table 10. Revised nitrogen loading rate limits to coastal waters for Buzzards Bay embayments proposed by the Buzzards Bay Project. See definitions in Table 5.

Embayment type	SB Waters	SA Waters	Outstanding Resource Waters
Shallow	300 mg m^{-3} Vr^{-1}	150 mg m^{-3} Vr^{-1}	50 mg m^{-3} Vr^{-1}
Deep	400 mg m^{-3} Vr^{-1}	200 mg m^{-3} Vr^{-1}	75 mg m^{-3} Vr^{-1}

Figure Legends

Fig. 1. Buzzards Bay, its drainage basin, and nitrogen management sub-basins surrounding its major embayments. This figure summarizes the nitrogen loading evaluation for the subwatersheds completed by the Buzzards Bay Project in 1993 and were based on the TMALs identified in Table 1.

Fig. 2. Generalized ecosystem responses of a) "shallow", well-mixed and b) "deep", stratified estuaries to nitrogen loading. Modified from Costa et al. (1993). Shallow System have most of their bottom above the light compensation point for seagrasses and algae)

Fig.3. Application of Buzzards bay Project proposed loading limits to a hypothetical 1 km^2 embayment with a 1 m mean depth with different flushing times.

Fig. 4. Comparison of nitrogen loadings using areal based loading scales and Eutrophication Stage for Buzzards Bay area estuaries and published studies as compared to Buzzards Bay Project recommended loading limits.

Fig. 5. Comparison of nitrogen loadings using Vollenweider Scale and Eutrophication Stage for shallow (<2.0 m MLW) estuaries in Buzzards Bay and other published studies as compared to Buzzards Bay Project recommended loading limits.

Fig. 6. National Atmospheric Deposition Program annual DIN loading in precipitation data for a Massachusetts station closest to the Buzzards Bay watershed (Station 13, Quabbin Reservoir).

Fig. 7. "Parameter transformation scale" used for evaluating Secchi Depth to create a score for incorporation into the Buzzards Bay "Eutrophication Index".

Fig.8. Scatter plots showing correlation between nitrogen loading, expressed using the volume Vollenweider-term flushing scale and the seven year mean (92-98)+/- std. errors of summertime secchi depth.

Fig.9. Scatter plots showing correlation between nitrogen loading, expressed using the volume Vollenweider-term flushing scale and the seven year mean (92-98) +/- std. errors of summertime DIN.

Fig. 10. Scatter plots showing correlation between nitrogen loading, expressed using the volume Vollenweider-term flushing scale and the seven year mean (92-98)+/- std. errors of summertime chlorophyll a.

Fig. 11. Scatter plots showing correlation between nitrogen loading, expressed using the volume

Vollenweider-term flushing scale and the seven year mean (92-98) +/- std. errors of summertime total organic nitrogen.

Fig. 12. Scatter plot showing correlation between nitrogen loading, expressed using the volume Vollenweider-term flushing scale, and the seven year mean (92-98)+/- std. errors of summertime oxygen saturation.

Fig. 13. Scatter plot showing correlation between nitrogen loading, expressed using the volume Vollenweider-term flushing scale, and the 1992 summertime oxygen saturation.

Fig. 14. Scatter plot showing correlation between nitrogen loading, expressed using the volume Vollenweider-term flushing scale, and the four year mean (92-95)+/- std. errors of summertime Eutrophication Index.

Fig. 15. Scatter plots showing correlation between nitrogen loading, expressed using the volume Vollenweider-term flushing scale, and 92-98 mean +/- std. errors of the Alternate Eutrophication Index scoring (without oxygen scores).

Fig. 16. Scatter plot showing correlation between nitrogen loading, expressed using the volume Vollenweider-term flushing scale, and the 1992-1998 mean +/- std. errors of summertime Total Nitrogen concentration. Correlation coefficients did not include data for Marks Cove (MRK) or Little River (LIR).

Fig. 17. Scatter plot showing correlation between nitrogen loading, expressed using the volume Vollenweider-term flushing scale with tidal prism DIN loadings added, and the seven year mean (92-98)+/- std. errors of total nitrogen.

Fig. 18. Scatter plots showing correlation between total nitrogen and three alternative measures of characterizing nitrogen.

Fig. 19. Scatter plots showing correlation between total nitrogen and watershed loading, expressed as a % of new Buzzards Bay Project recommended limits.

Fig. 20. Ratio of eelgrass habitat area to potential habitat area versus nitrogen loading, expressed using the volume Vollenweider-term flushing scale (see text).

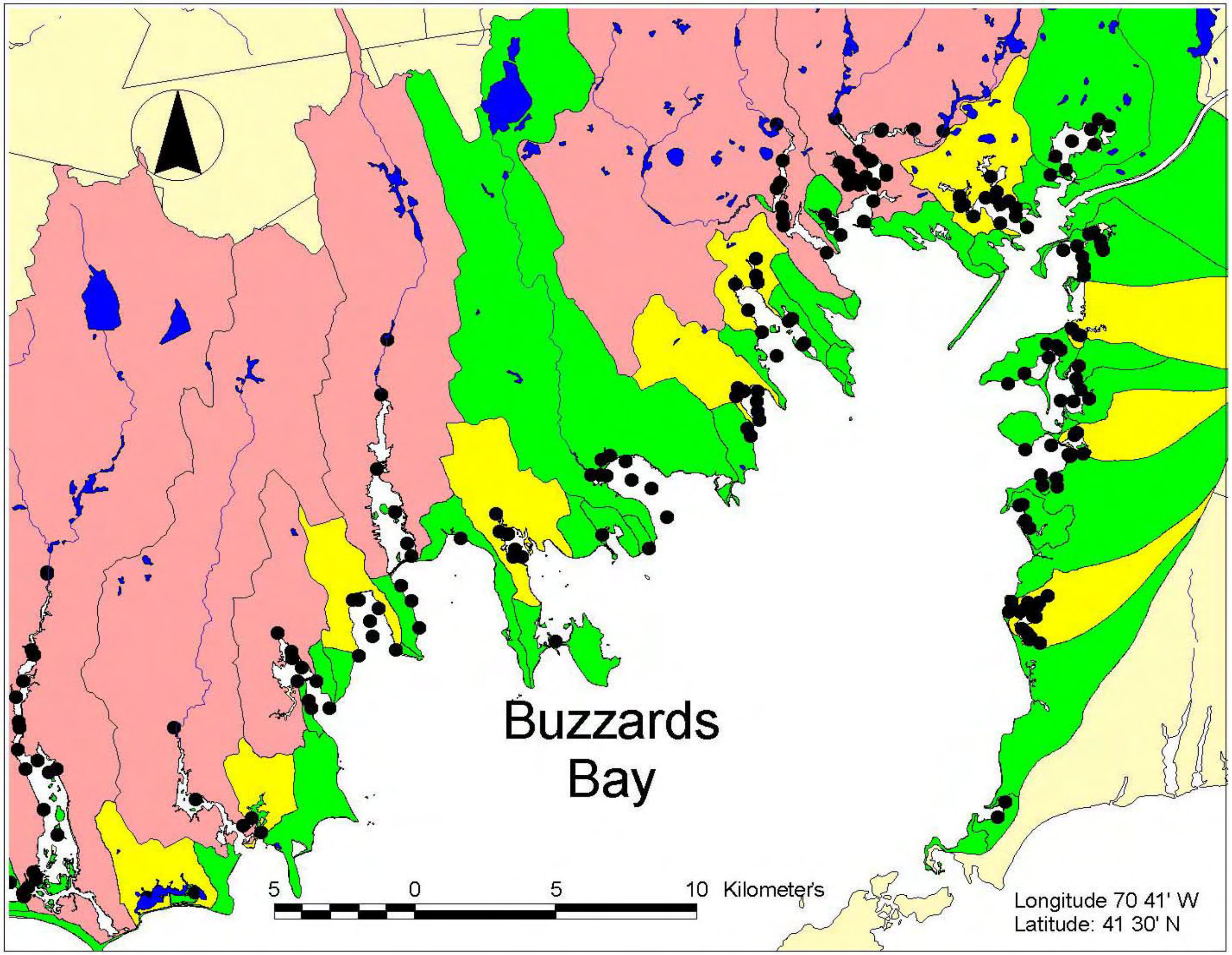
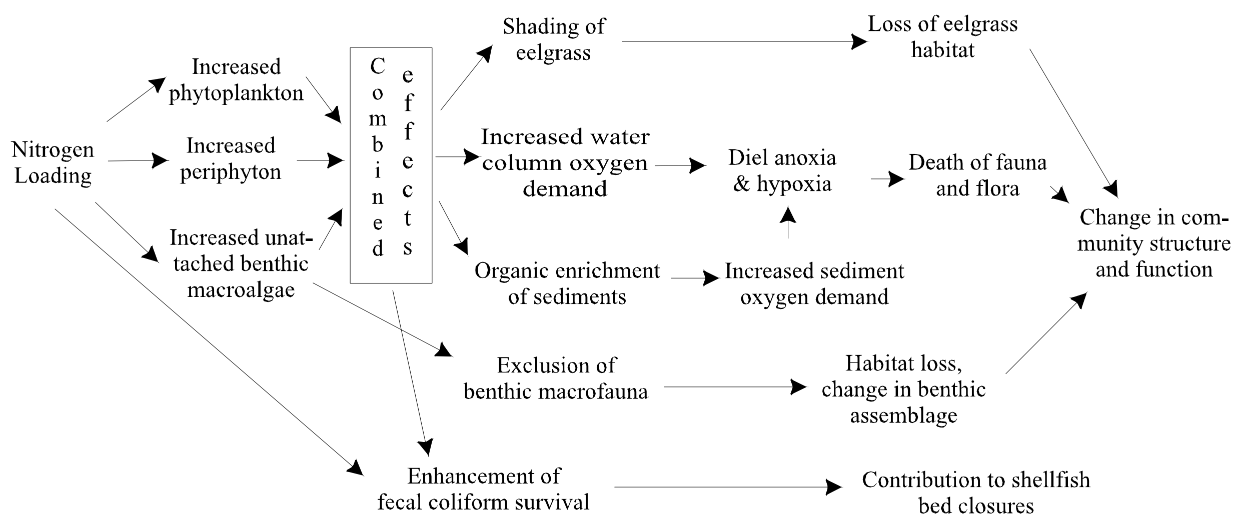
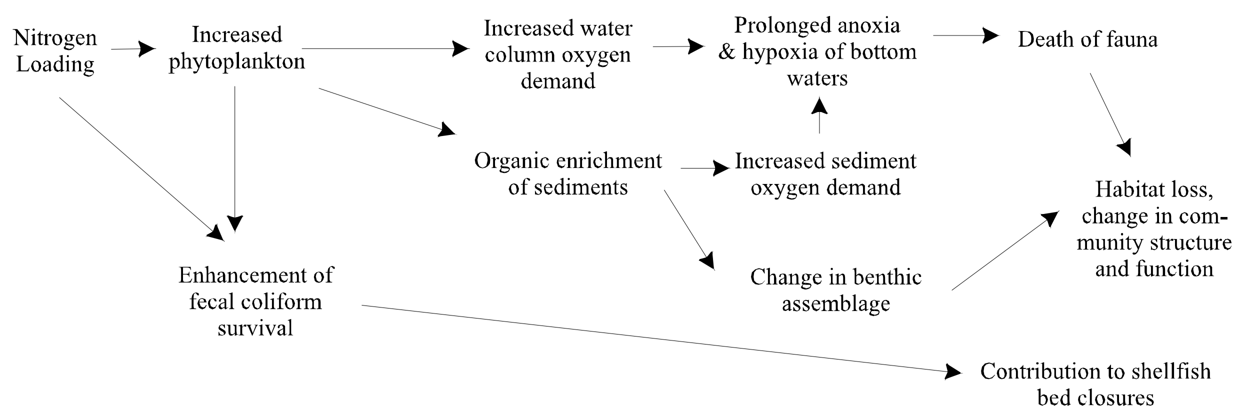


Fig. 1. Buzzards Bay, its drainage basin, and nitrogen management sub-basins, and various water quality monitoring stations surrounding its major embayments.

Fig. 2. Generalized ecosystem responses of a) "shallow", well-mixed and b) "deep", stratified estuaries to nitrogen loading. Modified from Costa et al. (1993). Shallow System have most of their bottom above the light compensation point for seagrasses and algae).





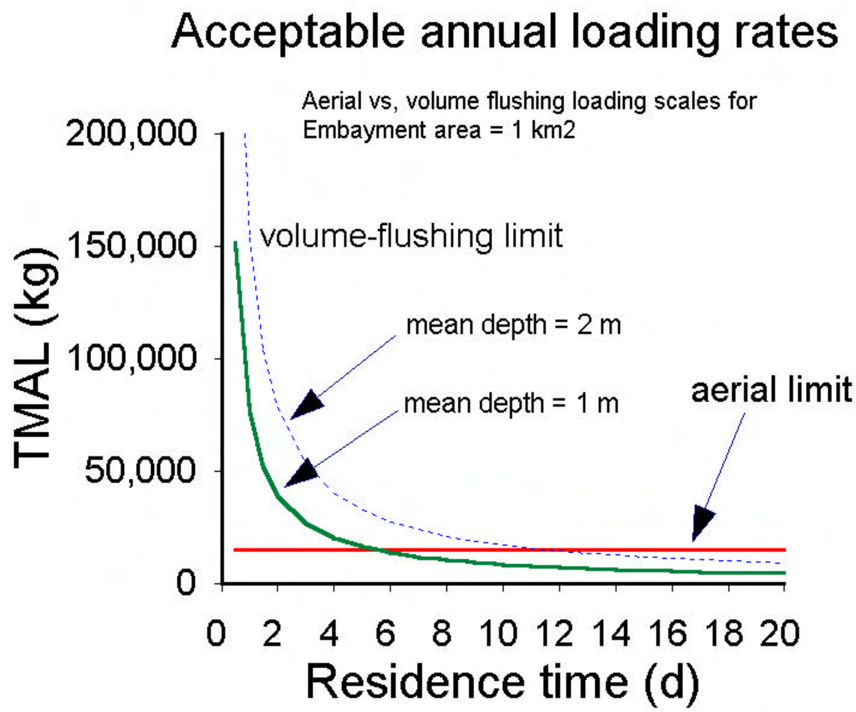
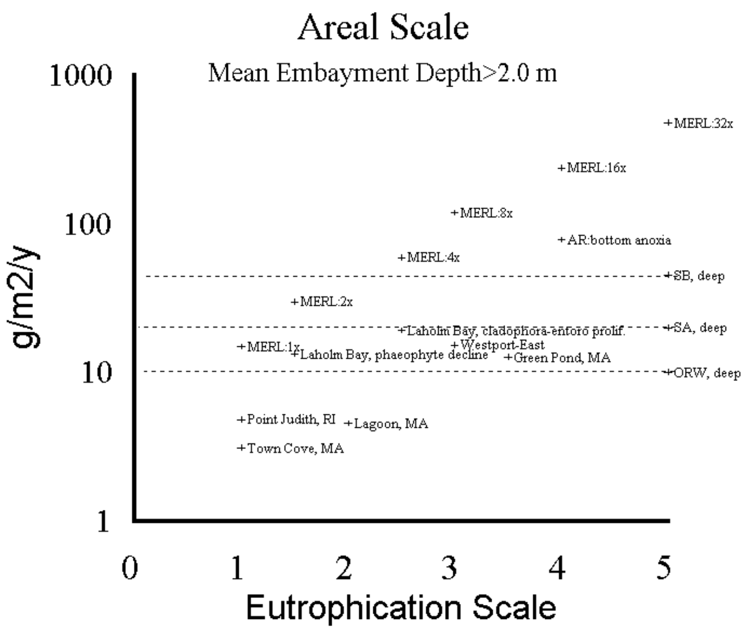
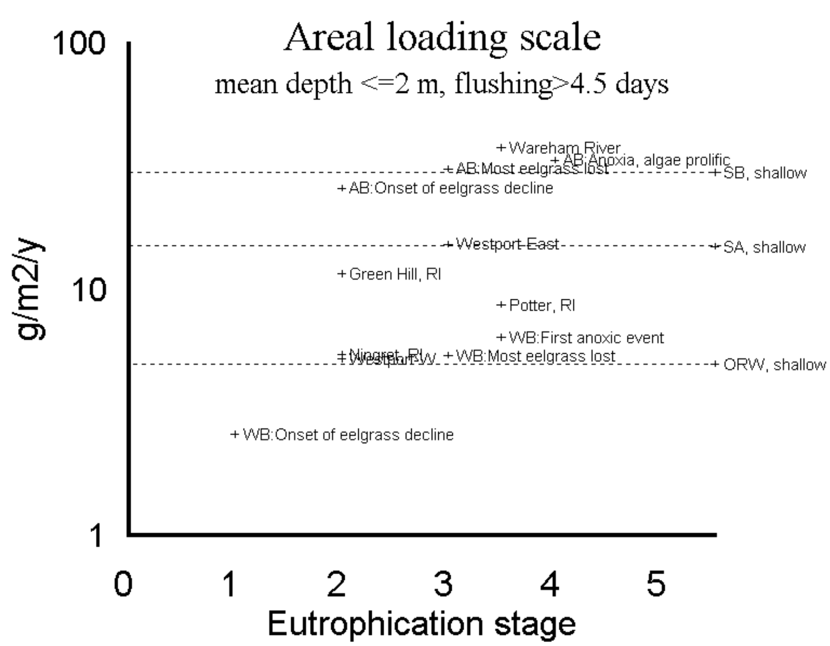


Fig.3. Application of Buzzards bay Project proposed loading limits to a hypothetical 1 km^2 embayment with a 1 m mean depth with different flushing times.

Fig. 4. Comparison of nitrogen loadings using areal based loading scales and Eutrophication Stage for Buzzards Bay area estuaries and published studies as compared to Buzzards Bay Project recommended loading limits. Fig 5a. Areal scale limits for embayment with a mean depth> 2.0 m MLW, especially results of MERL mesocosm experiments. Fig 5b. Areal scale for shallow embayments with flushing times greater than 4.5 days. The assumed hydrologic characteristics and loading to each embayment were based on assumptions used by the Buzzards Bay Project circa 1991(as in BBP, 1991 and Buzzards Bay Project draft reports). Some loadings and flushing times have since been revised (c.f. Table 4) based on new data or changed conditions. See text for explanation.





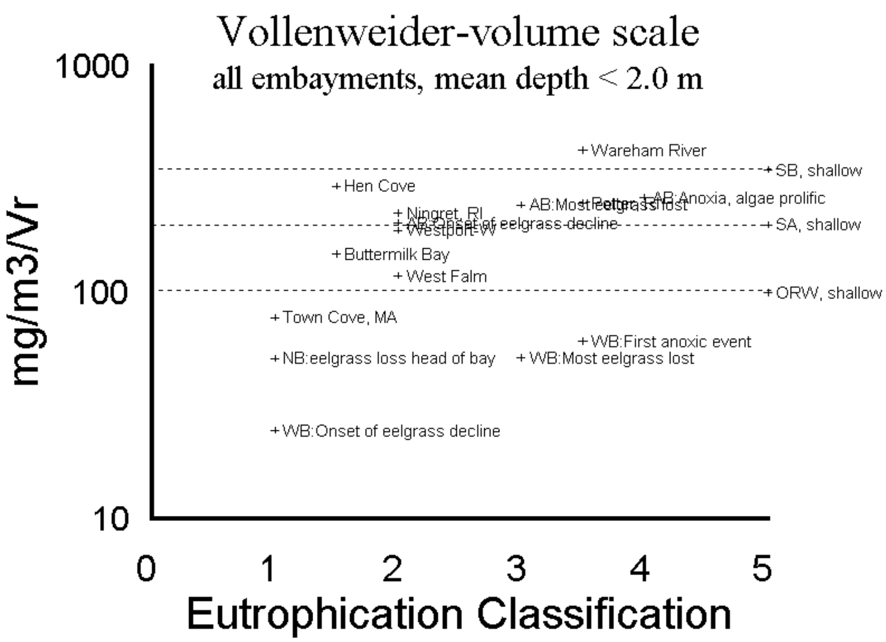


Fig. 5. Comparison of nitrogen loadings using Vollenweider Scale and Eutrophication Stage for shallow (<2.0 m MLW) estuaries in Buzzards Bay and other published studies as compared to Buzzards Bay Project recommended loading limits.

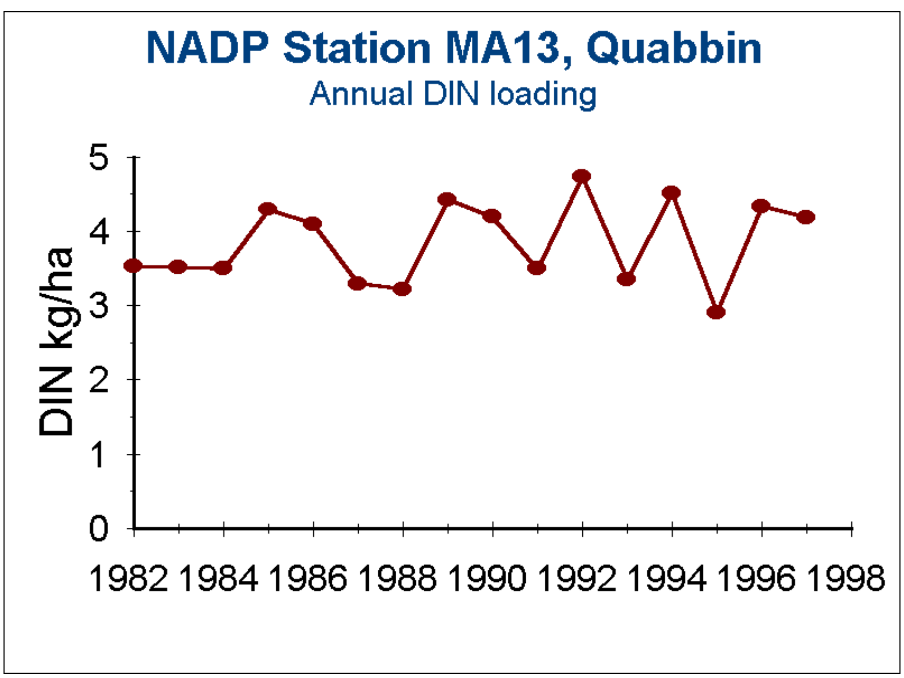


Fig. 6. National Atmospheric Deposition Program annual DIN loading in precipitation data for a Massachusetts station closest to the Buzzards Bay watershed (Station 13, Quabbin Reservoir).

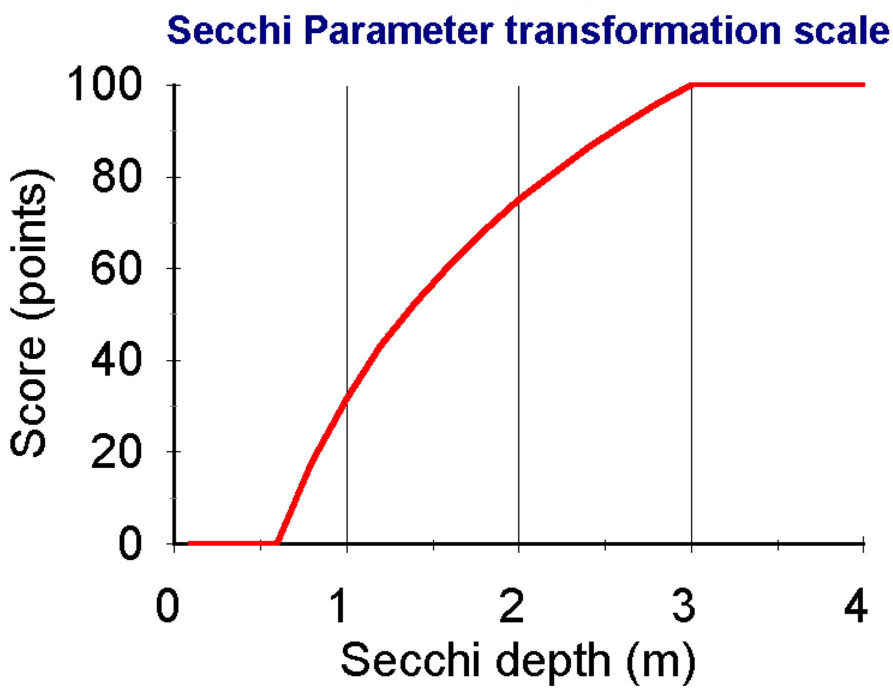


Fig. 7. "Parameter transformation scale" used for evaluating Secchi Depth to create a score for incorporation into the Buzzards Bay "Eutrophication Index".

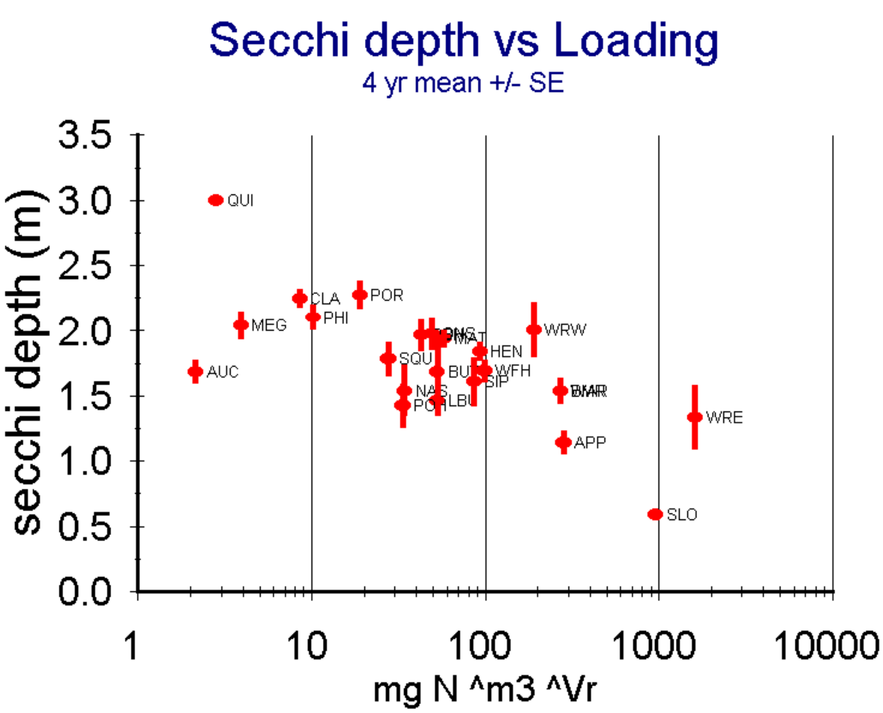


Fig.9. Scatter plots showing correlation between nitrogen loading, expressed using the volume Vollenweider-term flushing scale and the seven year mean (92-95)+/- std. errors of summertime secchi depth.

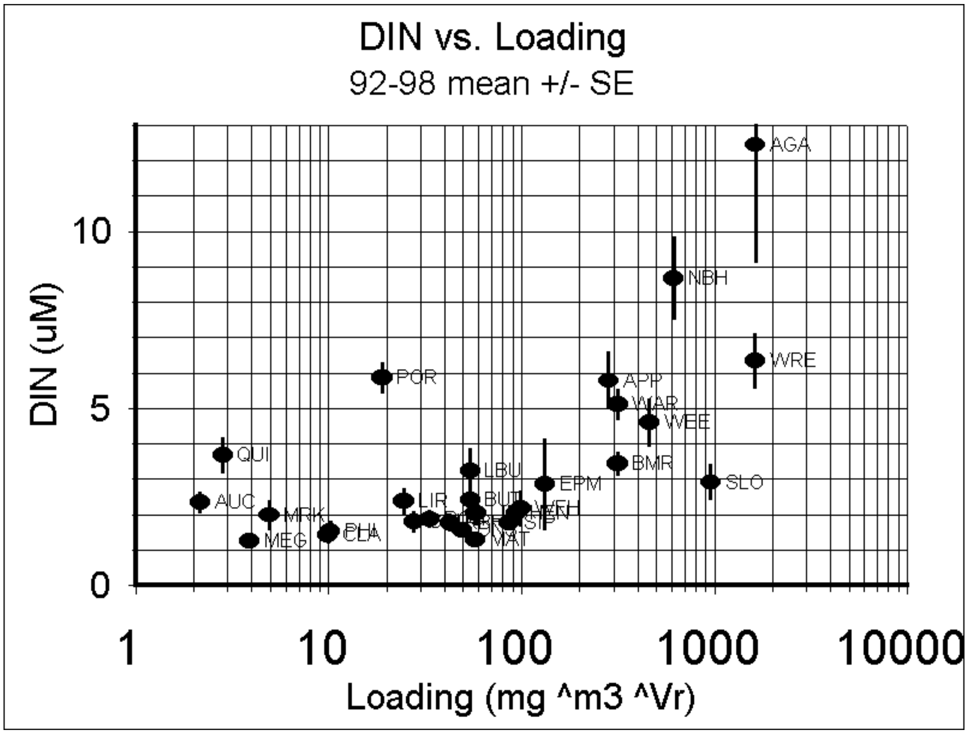


Fig.10. Scatter plots showing correlation between nitrogen loading, expressed using the volume Vollenweider-term flushing scale and the seven year mean (92-98) +/- std. errors of summertime total organic nitrogen.

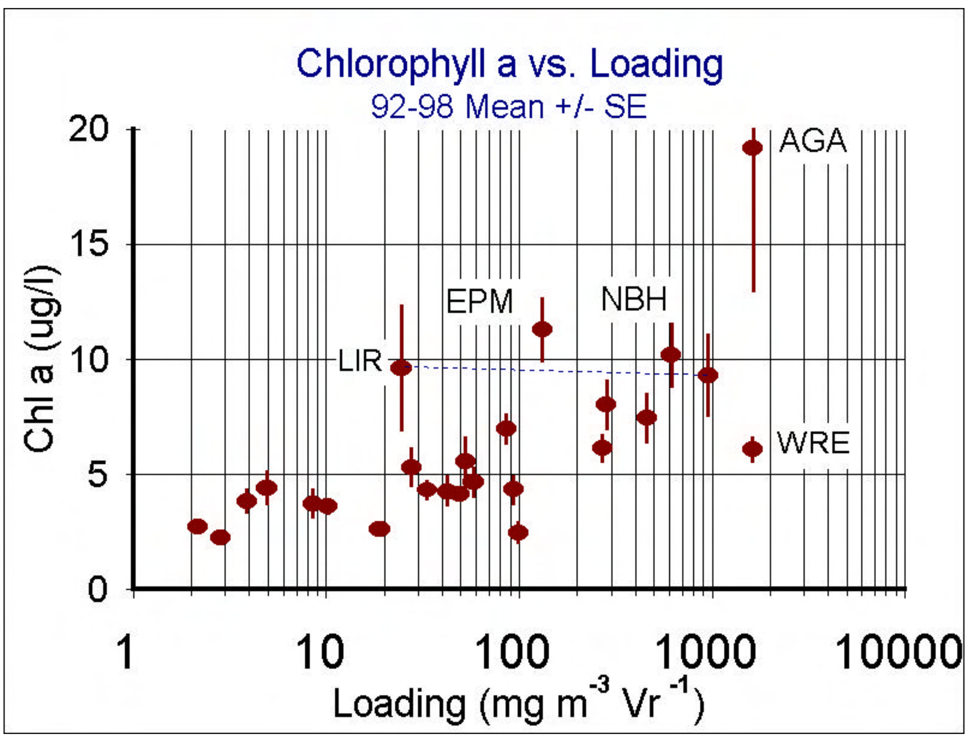


Fig. 11. Scatter plots showing correlation between nitrogen loading, expressed using the volume Vollenweider-term flushing scale and the six year mean (92-98, 1997 unavailable)+/- std. errors of summertime chlorophyll *a*.

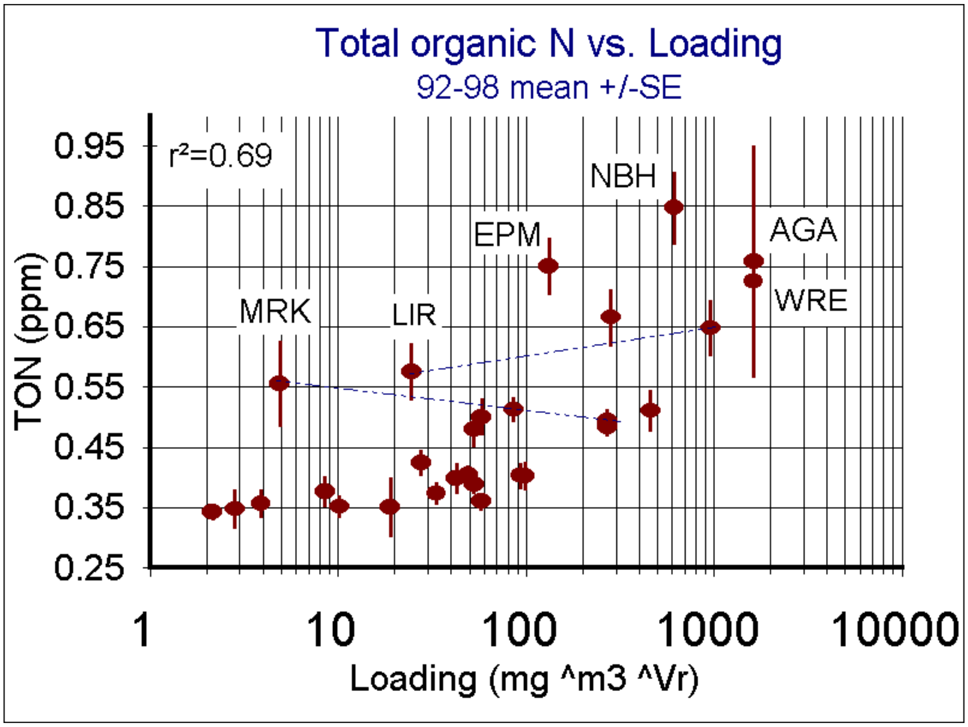


Fig. 12. Scatter plots showing correlation between nitrogen loading, expressed using the volume Vollenweider-term flushing scale and the seven year mean (92-98) +/- std. errors of summertime total organic nitrogen.

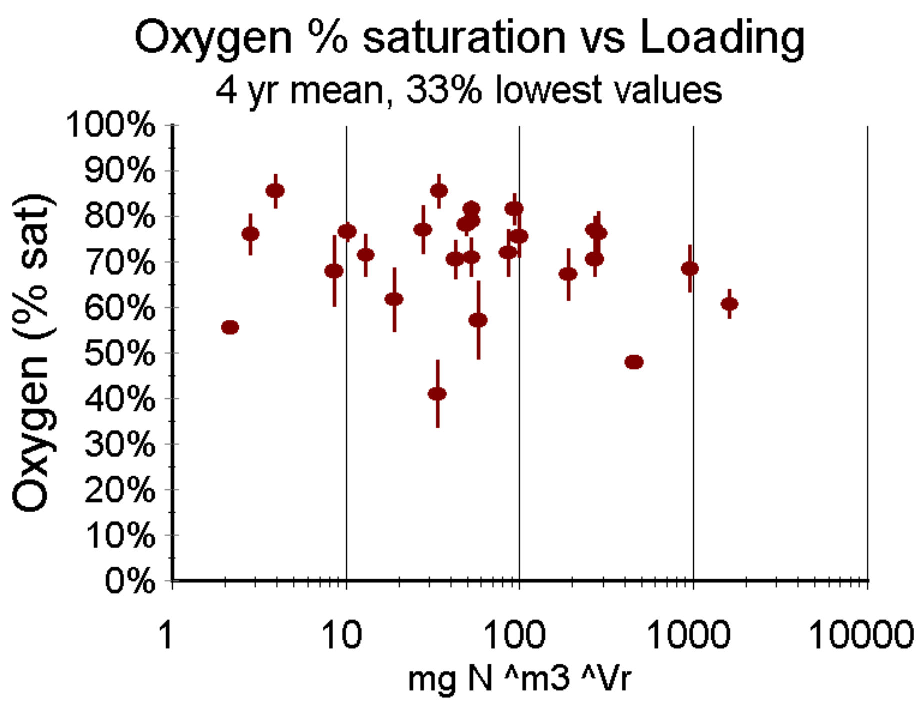


Fig. 13. Scatter plot showing correlation between nitrogen loading, expressed using the volume Vollenweider-term flushing scale, and the four year mean (92-95)+/- std. errors of summertime oxygen saturation

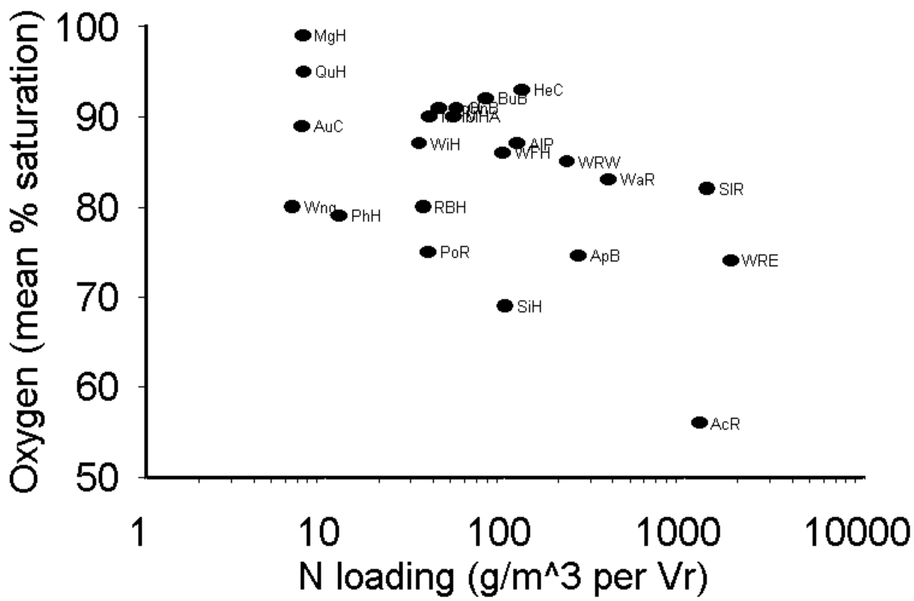


Fig. 14. Scatter plot showing correlation between nitrogen loading, expressed using the volume Vollenweider-term flushing scale, and the 1992 summertime oxygen saturation.

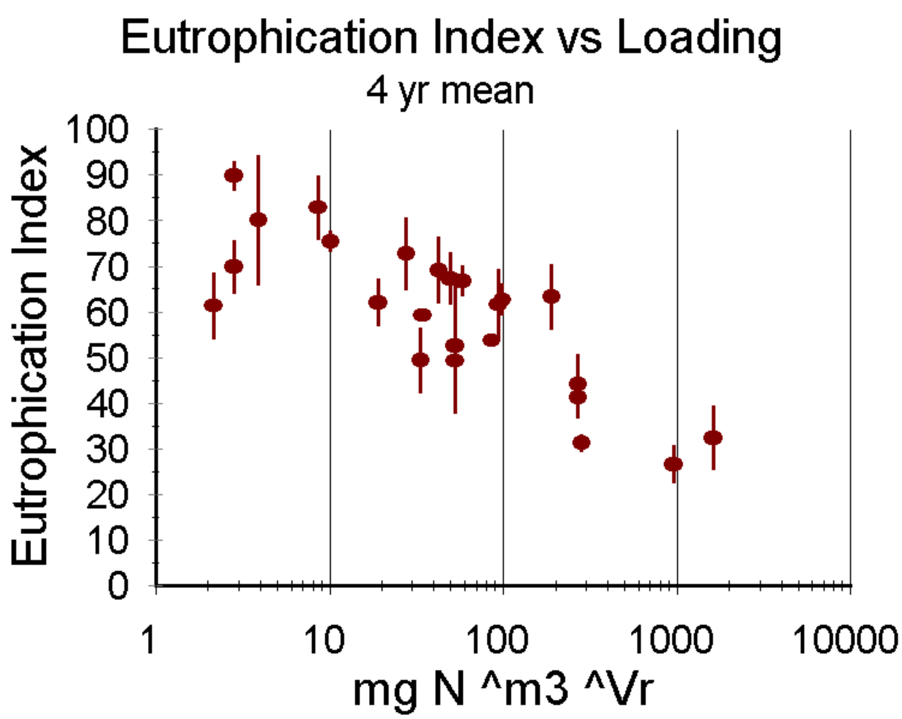


Fig. 15. Scatter plot showing correlation between nitrogen loading, expressed using the volume Vollenweider-term flushing scale, and the four year mean (92-95)+/- std. errors of summertime Eutrophication Index.

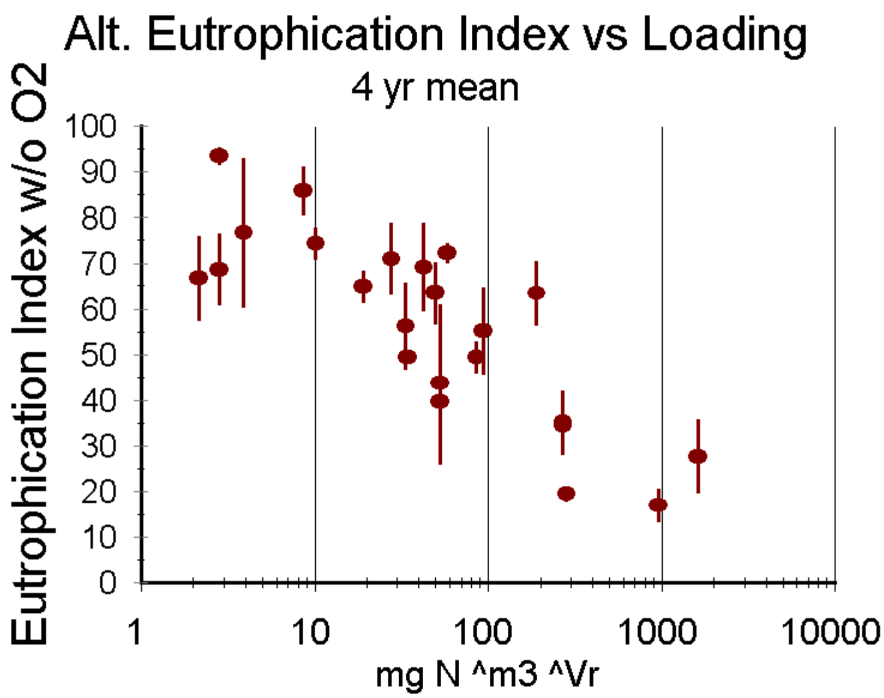


Fig. 16. Scatter plots showing correlation between nitrogen loading, expressed using the volume Vollenweider-term flushing scale, and 92-98 mean +/- std. errors of the Alternate Eutrophication Index scoring (without oxygen scores).

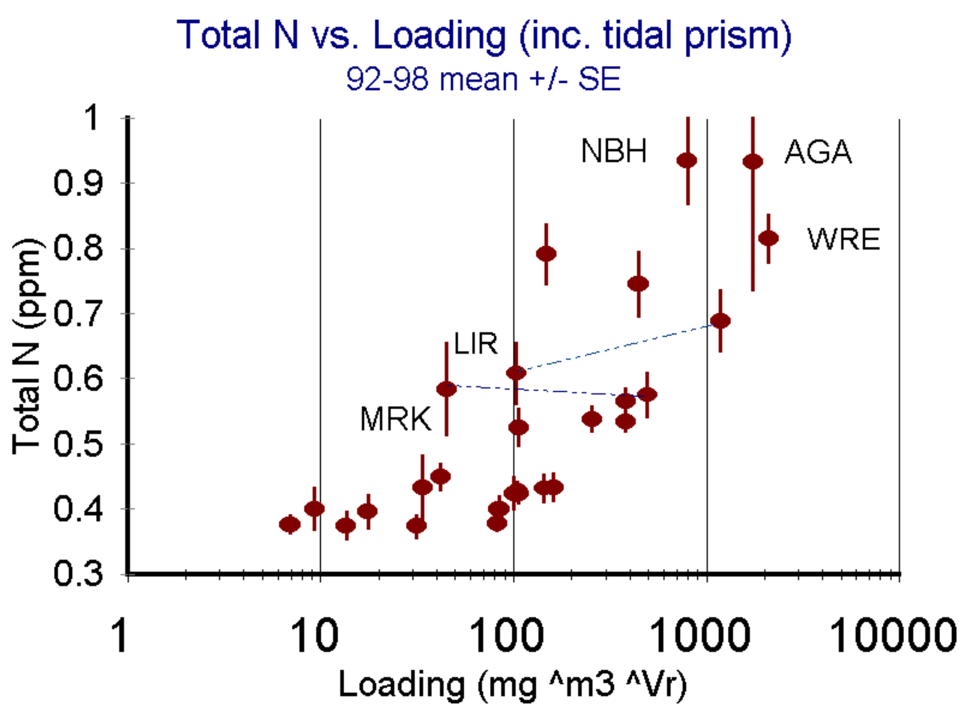


Fig. 17. Scatter plot showing correlation between nitrogen loading including tidal prism inputs, expressed using the volume Vollenweider-term flushing scale, and the 1992-1998 mean +/- std. errors of summertime Total Nitrogen concentration.

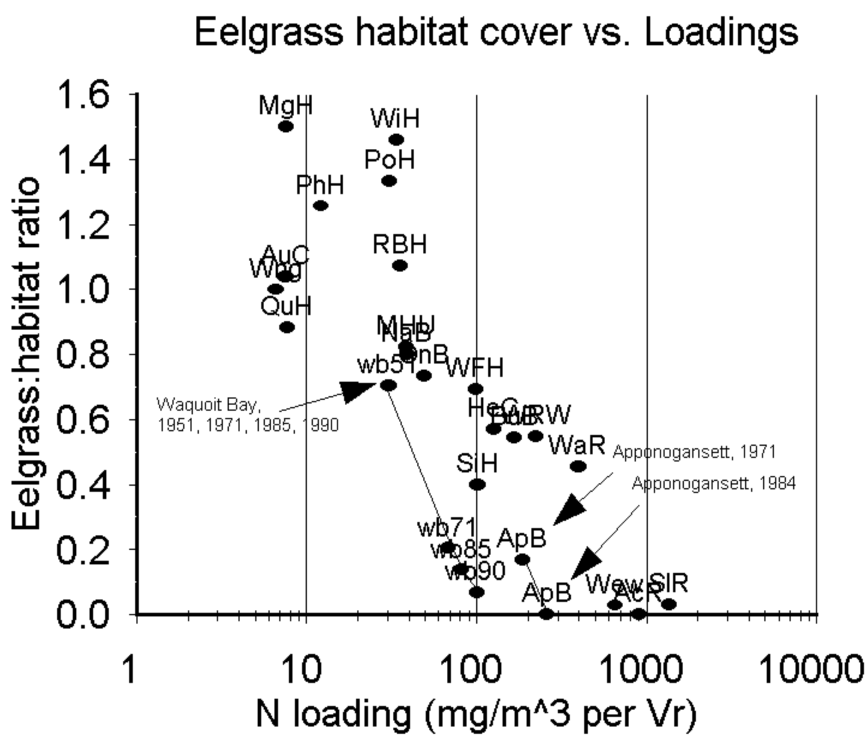


Fig. 18. Ratio of eelgrass habitat area to potential habitat area versus nitrogen loading, expressed using the volume Vollenweider-term flushing scale (see text).

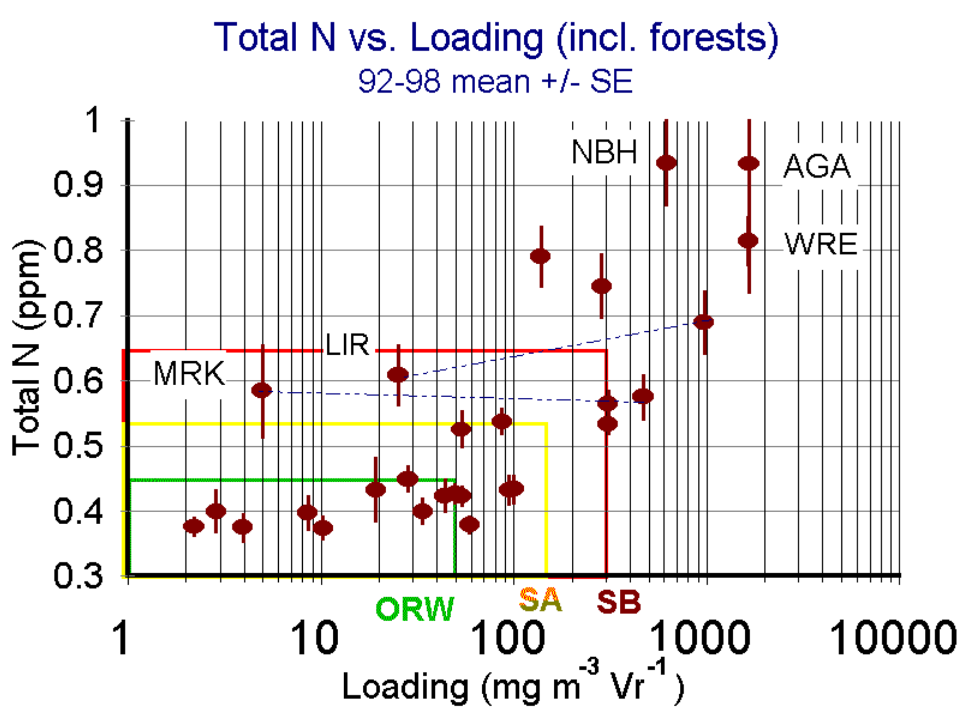


Fig. 19. Scatter plots showing correlation between total nitrogen and watershed loading, expressed as Vollenweider flushing volume scale, including background inputs of forests and other undeveloped lands. New proposed BBP TMALs also shown

Fig. 8. Scatter plots showing correlation between total nitrogen and four alternative measures of characterizing nitrogen loading. Selected stations identified (see Table 7). R^2 on Ln transformed loadings scales and TN. Embayments MRK and LIR not included in correlations.