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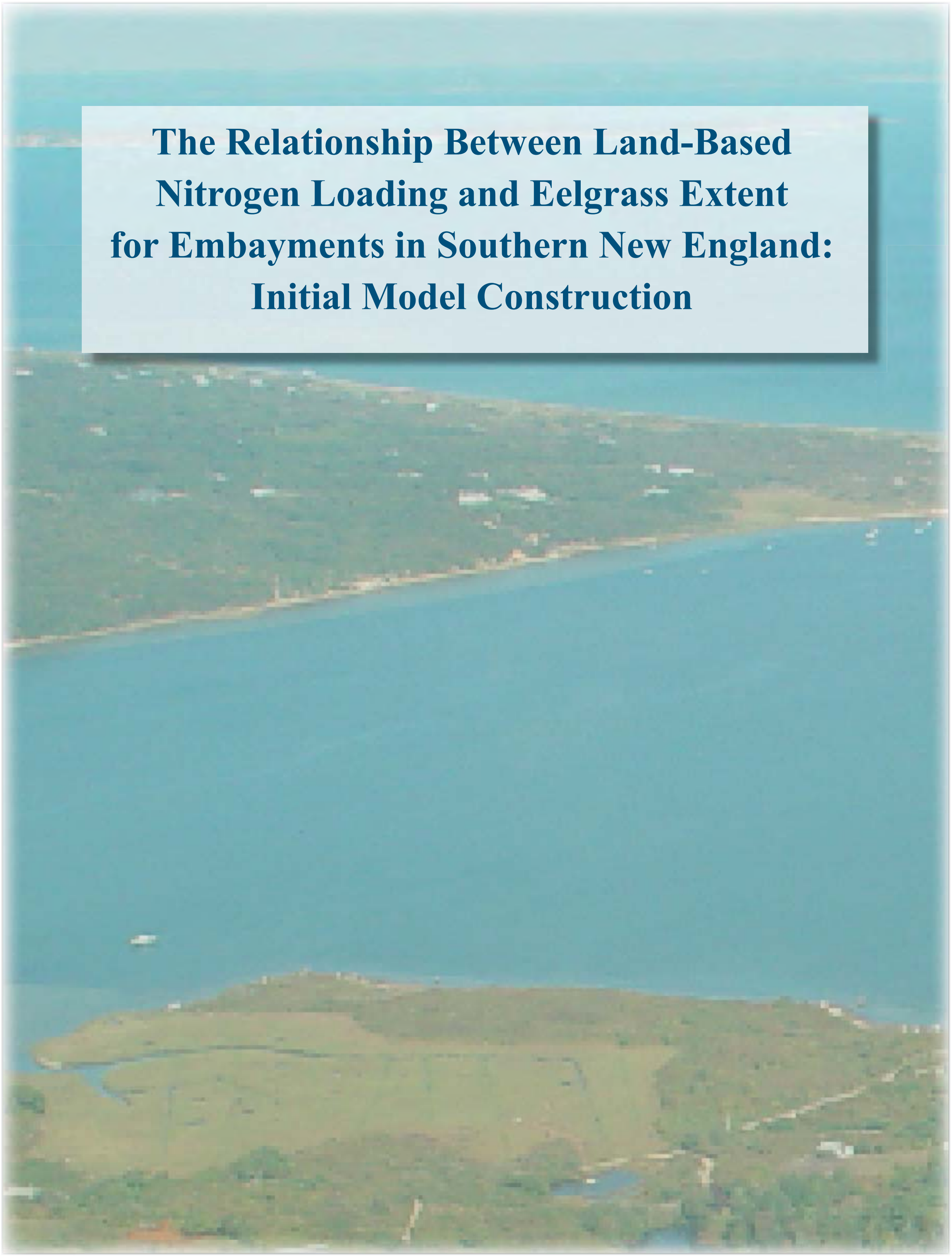

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The Relationship Between Land-Based Nitrogen Loading and Eelgrass Extent for Embayments in Southern New England: Initial Model Construction

James S. Latimer, Steven Rego, Giancarlo Cicchetti, Carol Pesch, Edward H. Dettmann, and Richard McKinney

U. S. Environmental Protection Agency
Office of Research and Development
National Health and Environmental Effects Research Laboratory
Atlantic Ecology Division
Narragansett, RI 02882 USA

Michael Charpentier
Computer Sciences Corporation
Narragansett, RI 02882

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Abstract

This report outlines research results of the US EPA Atlantic Ecology Division in fulfilling the National Health and Environmental Effects Laboratory's Aquatic Stressors Nutrient Program's charge to develop nutrient load-ecological response models useful in setting loading limits protective of estuarine designated uses. The results reveal that eelgrass extent is significantly related (r^2 = 0.82, $p<0.0001$) to land-based nitrogen loading when estuarine volume and flushing time are considered. Once this preliminary model is revised and validated, it can be used by local, state and tribal resource managers as part of the weight of evidence required to set nitrogen loading thresholds protective of eelgrass habitat for the class of estuaries defined as southern New England shallow embayments.

Key Words: New England, eutrophication, estuary, shallow embayment, nitrogen, water quality criteria, biocriteria, response variability, eelgrass, SAV, *Zostera marina*

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Background

Aquatic Stressors Research Plan

The National Health and Environmental Effects Research Laboratory's (NHEERL's) Aquatic Stressors research implementation plans focus on the effects on estuaries of four specific stressors: habitat alteration, nutrients, suspended and bedded sediments, and toxic chemicals (EPA 2002a). This focus is consistent with recent scientific consensus, recognizing that these stressors have the greatest potential for causing adverse effects to aquatic ecosystems (EPA 2002a). The ultimate goal of the Aquatic Stressors research is to develop scientifically valid approaches for protecting aquatic ecosystems from the impacts of these stressors and restoring those systems that have been degraded. The immediate focus is to develop and improve assessment methodologies, diagnostic capabilities, and ecological criteria to guide management options for protection, restoration and remediation efforts to meet designated uses.

Excess Nutrients as Aquatic Stressors

It is well established that human activities have dramatically changed the amounts, distributions, and movements of major nutrient elements (nitrogen and phosphorus) in the landscape and have increased nutrient loading to receiving waters (Howarth *et al.* 2002). Some of these changes affect use of the nation's aquatic resources and pose risks to human health and the environment (NRC 2000). EPA is in the process of developing guidelines that states and tribes can use to set nutrient criteria for the nation's waters. For waters failing to meet water quality standards, states and tribes are required to develop total maximum daily loads (TMDLs) to eliminate the causes of non-attainment. Our current understanding of aquatic ecosystem function is inadequate to allow extrapolation of relationships between nutrient load and ecological effects for systems with extensive data to prediction of adverse impacts in those with more limited data. NHEERL research will provide the load-response relationships for classes of estuaries around the country that will facilitate the translation of effects-based numeric criteria to nutrient loading limits that are protective of aquatic life.

Ecological Response to Excess Nutrients

Ecological responses to excess nutrients generally fall into two categories: primary and secondary (Cloern 2001). The primary response is an increase in algal production (or carbon supply as defined by Nixon (1995)) and/or shifts in the algal community composition at the base of the food web. Secondary responses include increases in extent and duration of hypoxia, losses of submerged aquatic vegetation including eelgrass, and changes and losses of biodiversity including changes in fish abundance and species composition.

Seagrasses enhance estuarine habitats by acting as food sources for consumers and by providing sediment enrichment through decomposition, stabilization and erosion control. Seagrasses are highly valuable nurseries and forage grounds for many kinds of fish, shellfish and wildlife, and are considered one of the most productive ecosystems in the world (Duarte 2001). They provide both food and shelter from predators for a variety of shallow water nekton (Orth *et al.* 1984). These habitats are areas of high organic matter production, typically between 500-1000 g C m^2/yr, and their plant morphology and growth complexes create micro-habitats for nekton (Hoss & Thayer 1993). Benthic and pelagic diversity is also higher in seagrass habitats than in other habitat types (Hughes *et al.* 2002). In recent literature reviews, researchers found that fish abundance increased with seagrass extent (Beck *et al.* 2001, Heck Jr. *et al.* 2003). Seagrasses are an important location for attachment of juvenile bivalves such as the bay scallop (*Argopecten irradians*) and the hard clam (*Mercenaria mercenaria*), and these habitats provide protection for these juveniles (Orth *et al.* 1984). Although juvenile shellfish and fish species can use other types of habitat, seagrasses provide the bulk of shelter in many estuaries, and the loss of these vegetated habitats may produce declines in juvenile fish (Wyda *et al.* 2002). Fragmentation of vegetated habitats can also have important implications for species such as the bay scallop, since patchy seagrass beds (with a high edge to interior ratio) can lead to enhanced rates of predation on juvenile scallops (Irlandi *et al.* 1995).

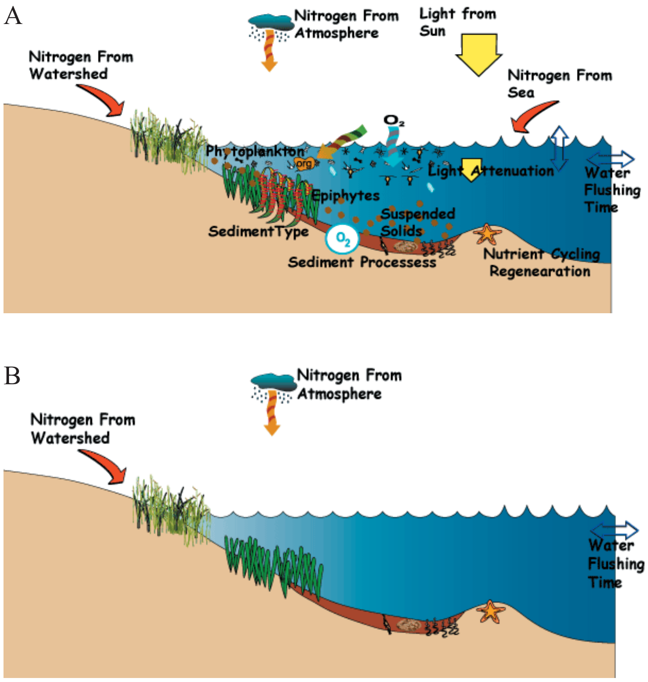

Figure 1A. Conceptual diagram of some of the processes that affect the health and extent of eelgrass. B. Simplified conceptual diagram utilized for the comparative systems empirical approach.

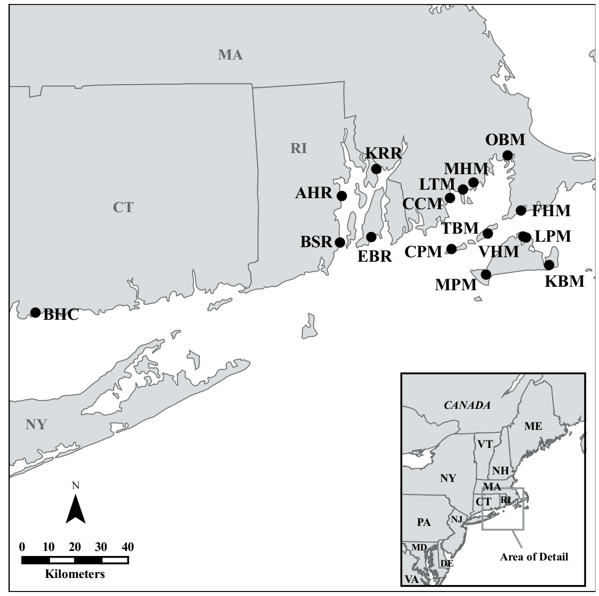

Figure 2. Study systems for the 2002 aerial eelgrass survey.

Anthropogenic loading of nutrients has been implicated in the loss of seagrasses (Short *et al.* 1995). The major cause of seagrass decline is not thought to be via nitrate toxicity, but rather through light limitation by planktonic, macro-algal or epiphytic algal shading (Duarte 1995, Hauxwell *et al.* 2001, Hauxwell *et al.* 2003) (Figure 1A). Geochemical processes that are associated with high organic loading such as hypoxia and sulfide production in the sediment have also been shown to affect the health of seagrasses (Eldridge *et al.* 2004).

There is a strong consensus among estuarine ecologists that excess nitrogen, not excess phosphorus, is the main cause of eutrophication in most estuaries (Howarth & Marino 2006). Therefore, in this document we report preliminary relationships between watershed-based nitrogen loading rate and eelgrass (*Zostera marina)* extent in southern New England shallow embayments. Additional reports are forthcoming on the relationship between nitrogen loading rate and primary productivity and benthic condition in similar coastal embayments.

Methods

Overview

The development of empirical models that relate nitrogen load to ecosystem response is based on a comparative systems approach in which loading and ecological responses are determined for a number of study embayments along a nitrogen load gradient within one class of estuary. The program is divided into three major components: 1) determination of nitrogen loading rate to coastal embayments from the watershed and atmosphere; 2) assessment of ecosystem response (in this case eelgrass extent) and 3) application of normalizing factors to refine the load-response models. Figure 1B describes the components of the conceptual model that are measured or estimated using the empirical comparative systems approach.

Study Systems

Sixteen (16) estuarine embayments, reflecting a gradient of watershed-derived nitrogen loading in southern New England, were evaluated in this study (Figure 2).

The study estuaries are considered to be representative of a specific class, one that will be defined using the Coastal and Marine Ecological Classification Standard (CMECS) conceptual classification scheme (Madden *et al.* 2005). The CMECS is an ecosystem-oriented, science-based framework for the identification, inventory, and description of coastal and marine habitats and biodiversity. The study systems fall into the following Level 1 and 2 CMECS Hierarchical Classification categories: Level 1 Regime: Estuarine, Level 2 Formation-Geoform: Embayment. Thus, at the highest class levels of the CMECS, the study systems are defined as estuarine embayments. In this context, estuaries are defined as enclosed or semi-enclosed coastal water bodies that are influenced by fresh water input that reduces salinity to below 30 psu during at least two months of the year. Moreover, an embayment is an estuary that is partially enclosed or surrounded by a landmass but that has a significant open connection to the sea (Madden *et al.* 2005).

The CMECS scheme includes additional chemical, physical, biological and biogeographic descriptors to further refine the upper level class designations. Table 1 contains the descriptors and the applicable values for the study systems of this research summary. Table 2 provides some additional specific data for the individual embayments. For simplicity, throughout this report, we will refer to the class of estuaries under study as southern New England shallow embayments. Nevertheless, the reader should keep in mind the additional components of the study systems noted in Table 1.

Table 1. CMECS descriptors to define the class of estuarine embayment for the study systems (Madden *et al.* 2005).

Descriptors	Magnitudes
Chemical	
Temperature Class:	Cold (0-10°C) to temperate (10-20°C)
Salinity Class:	Mesohaline (5-18 psu) to euhaline (30-40 psu)
Oxygen Class:	Variable (anoxic, hypoxic, oxic, saturated, supersaturated)
Turbidity Class:	Moderately turbid (2-4 m)
Turbidity Type:	Mixed (chlorophyll, mineral, colloidal, dissolved color, detrital)
Turbidity Provenance:	Mixed (allochthonous, autochthonous, resuspended, terrigenous, marine)
Physical	
Energy Type:	Wind/tide/current
Energy Intensity:	Moderate (moderate currents and wave action, 2-4 kn)
Energy Direction:	Mixed
Depth Class:	Very shallow (0-5 m)
Tide Class:	Small (0.1-1 m) to moderate (1-5 m) tidal range
Primary Water Source:	Watershed, local estuary, local marine (non-river dominated)
Enclosure Status:	Partially-enclosed (50-75% area encircled by land)
Biological	
Trophic Status:	Oligotrophic (<5 ug Chl-a/L) to eutrophic (>50 ug Chl-a/L)
Ecological	
Region:	Eight (8); Virginian Atlantic Region. The region extends along the eastern North American continent from Cape Hattaras northward to Cape Cod. The region lies within the temperate limatological zone, and is interposed between the east coast and the Northern Gulf Stream Transition Region offshore (Region 9).
Additional[1]	
Embayment Size:	Small (0.1 km^2) to medium (6 km^2)
Watershed Size:	Small (0.5 km^2) to medium (73 km^2)
Ecoregions:	Northeastern Coastal Zone and Atlantic Coast Pine Barrens (Shirazi *et al.* 2003)
Geographic:	Southern New England Region (CT, RI and southeastern MA coastal)

[1]These descriptors are important in further refining the class of estuarine embayments included in this report.

Table 2. Characteristics of the eelgrass study estuarine embayments.

Embayment	State	ID	Larger Systems	Direct WWTF Sources	Total Embayment Area[1] (km^2)	Embayment Volume[2] (m^3x10^3)	Watershed Area (km^2)[3]	Flushing Time (d)[4]
Branford Harbor	CT	BHC	Long Island Sound	Yes	1.41	1,450	71.3	3.27
Clarks Cove	MA	CCM	Buzzards Bay	No	2.79	10,700	7.08	4.11
Cuttyhunk Pond	MA	CPM	Buzzards Bay-Outer	No	0.40	622	0.50	2.15
Falmouth Inner Harbor	MA	FHM	Nantucket Sound	No	0.12	301	1.58	1.46
Katama Bay	MA	KBM	Atlantic Ocean	No	5.88	10,900	9.58	5.26
Lagoon Pond	MA	LPM	Nantucket Sound	No	2.12	7,690	12.6	3.75
Little Bay	MA	LTM	Buzzards Bay	No	0.88	710	10.4	2.80
Mattapoisett Harbor upper	MA	MHM	Buzzards Bay	No	2.81	8,190	73.2	4.11
Menemsha Pond	MA	MPM	Vineyard Sound	No	2.76	6,130	4.20	4.09
Onset Bay	MA	OBM	Buzzards Bay	No	2.59	3,840	11.8	4.01
Tarpaulin Cove	MA	TCM	Vineyard Sound	No	0.78	3,420	1.88	2.69
Vineyard Haven-Inner	MA	VHM	Vineyard Sound/Harbor	No	0.18	673	0.84	1.66
Allen Harbor	RI	AHR	Narragansett Bay	No	0.31	913	4.62	1.99
Bonnet Shores	RI	BSR	Narragansett Bay	No	0.69	3,120	3.72	2.58
Easton Bay	RI	EBR	Narragansett Bay	No	2.00	8,130	15.6	3.68
Kickamuit River	RI	KRR	Narragansett Bay	No	2.24	4,380	20.1	3.82

WWTF = wastewater treatment facility

[1] Determined using GIS analysis: Total Embayment Area: Existing coastline data were used as a starting point to delineate the embayments. For each embayment, a bounding line was chosen to define the terminal extent of the embayment. For some embayments, a bounding line was also chosen for the upstream extent of the embayment. Seaward boundary was defined using shoreline features and best professional judgment. The area of water was summed within each embayment to arrive at the total embayment area. Additional metadata are available from first author.

[2] Estimated using GIS analysis and bathymetry data (USGS Mylar 7.5 minute quad maps). Additional metadata are available from first author.

[3] Determined using GIS analysis: Watershed area without embayment: Existing watershed data for each embayment were augmented by delineations performed on-screen using USGS topographic sheets as backdrops. Once the data layers were completed, the ArcInfo software generated area values. The area of the embayment itself was not included in this value. Additional metadata are available from first author.

[4] See text and appendix 1 for methodology.

Estimation of Nitrogen Loading Rate

Watershed and atmospheric derived nitrogen loading rates were estimated for each of the study embayments using a modification of a published nitrogen loading model (NLM) (Valiela *et al.* 1997) (Figure 3). As originally constructed, the model predicts total dissolved nitrogen loads to shallow embayments for rural and suburban watersheds where the watersheds are underlain by unconsolidated sands and groundwater flow is the dominant method of transport. The original NLM includes nitrogen inputs from wastewater (via septic systems, using values for per capita contributions of nitrogen), fertilizer use on turf and agriculture, and atmospheric deposition (estimated from regional data). We have modified the model to include inputs from wastewater treatment facilities (WWTFs) for those embayments when present. Equations in the NLM depict attenuation of nonpoint source nitrogen during passage through the different landcover types on a watershed (*i.e.*, natural vegetation, turf, agriculture, and impervious surfaces) and losses during travel through the soils, the vadose zone, and the aquifer (Figure 3). WWTF inputs to the embayments are not attenuated and are computed from point source effluent monitoring data. The attenuation rates used in each component of the watersheds were from published empirical measurements (Valiela *et al.* 1997). The estimates derived from the NLM reflect the sum of the attenuated nitrogen loading from each source (wastewater, fertilizer, and atmospheric deposition) to produce an estimate of the total dissolved nitrogen entering the receiving embayment.

Figure 4 provides a listing of the assumptions, calculations, and variables used in the model and Table 3 provides the estimated nitrogen loading rate to the study embayments used for this report. Ocean-derived nitrogen loading rates were not included in the estimates. Loads ranged from 1,050 – 42,000 kg N/yr. Normalizing the loading to embayment area yields a calculated loading range of 23.5 – 330 kg N/ha/yr; this is similar to other shallow embayments in the region (5.3 – 407 kg N/ha/yr (Hauxwell *et al.* 2003)) and comparable to, although on the high side of, estuaries nationwide (1.0 – 49.0 kg N/ha/yr (Castro *et al.* 2003)).

The nitrogen loading estimates from the NLM were compared to those derived from the USGS Spatially Referenced Regression on Watersheds (SPARROW) model (Moore *et al.* 2004) to assess correspondence. The loading output from the NLM compared favorably with output from the SPARROW model when applied to a set of common study systems (Figure 5A; $r^2 = 0.97$, n = 17). Even when the highest loaded system is excluded, the fit is quite good (Figure 5B, $r^2 = 0.73$, n = 16). The SPARROW model calculates nitrogen and phosphorus delivery to the coastal environment, in a very different way from the NLM, by estimating the delivery of total nitrogen and accounting for in-stream losses based on the travel time and stream flow for each stream reach within the continental United States (Smith *et al.* 1997, Moore *et al.* 2004) So the fact that the two models, each with very different conceptual frameworks, provide comparable estimates suggests that they may reflect something of the actual nitrogen inputs to the embayments.

The land use model has been evaluated for the uncertainty of estimated loading values (Collins *et al.* 2000). Using parametric (propagation of error estimates) and nonparametric (bootstrap and enumeration of combinations) statistical techniques on 8 of the 16 input variables, and testing them on one estuarine system, it was estimated that the standard error of the single value estimate (*i.e.*, N load kg N/yr) was ± 14% and the 95% confidence interval (CI) was 73-127%. The authors recommend, however, that a better assessment of the uncertainty of the loading estimate is accomplished by estimating the central tendency in an inferred population. This will give the worst case estimate of uncertainty and, according to the authors, better reflects the heterogeneity of watershed characteristics. This estimate was the standard deviation of the population distribution. Its value was ± 38% with the 95% CI of 25-175%. These estimates of uncertainty in loading should be considered by Regional, State and Tribal authorities in the sphere of planning, management, and risk assessment.

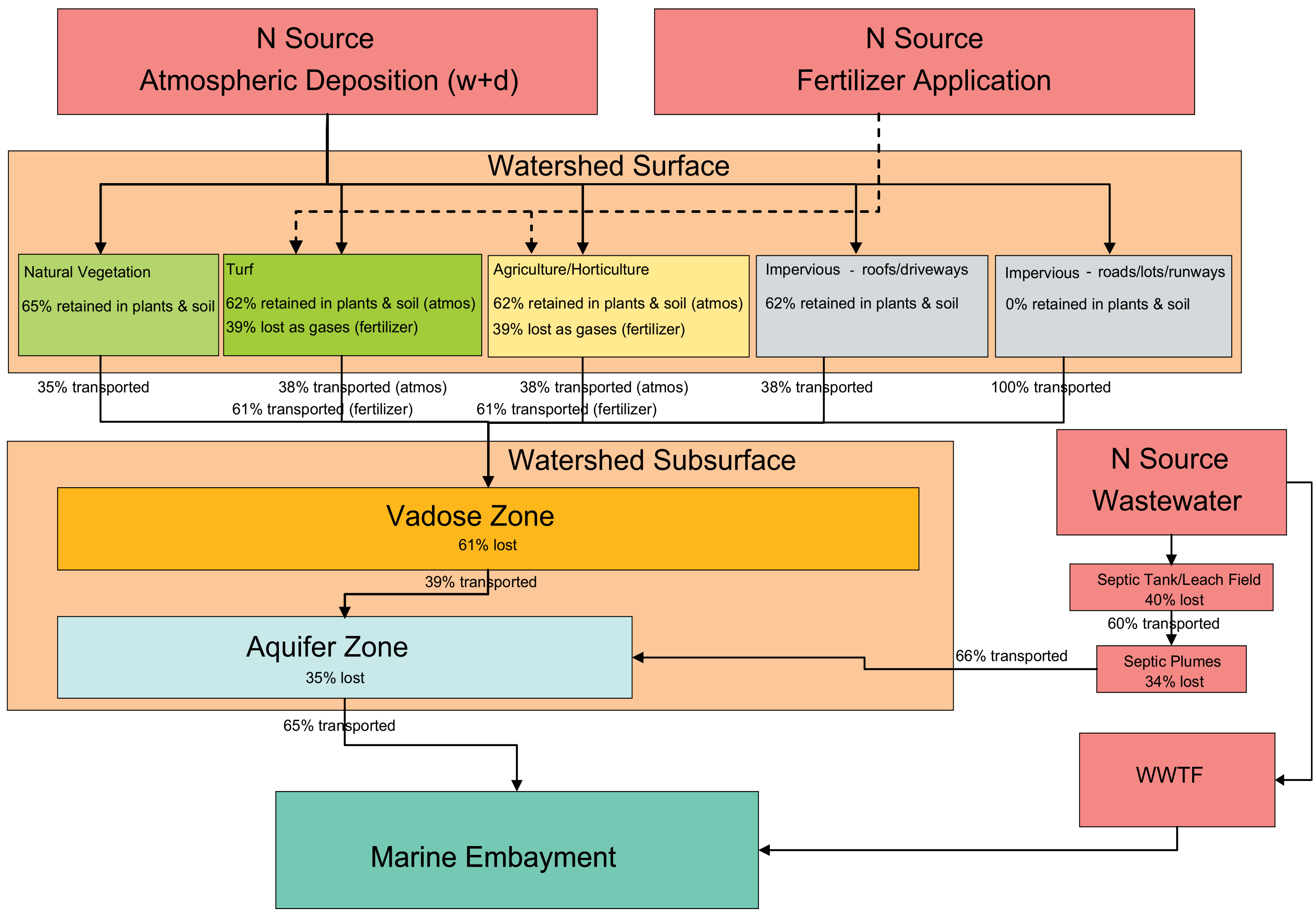

Figure 3. Schematic diagram of the nitrogen loading model (NLM) with WWTFs (wastewater treatment facility) included; modified from (Valiela *et al.* 1997); W+D = wet and dry atmospheric deposition; WWTF data were derived from reported monitoring data (sources: CT, Paul Stacey (CT DEP); RI, Scott Duerr (City of Westerly), Scott Nixon (Univ. RI); MA, Brian Friedman, Russell Isaac (MA DEP)).

Input Category	Included Landuse Types	Nitrogen Load
***Atmospheric Deposition*:**		
Natural Vegetation	forests, wetlands, natural lands	ATMOS. DEP.[1, 2] x AREA (kg N ha/ yr) (ha)
Turf	lawns, golf courses	ATMOS. DEP. x AREA (kg N ha/ yr) (ha)
Agricultural Land	crop land	ATMOS. DEP. x AREA (kg N ha/ yr) (ha)
Impervious Surfaces: roofs, driveways	roofs, driveways	ATMOS. DEP. x AREA (kg N ha/yr) (ha)
Impervious Surfaces: roads, runways, parking lots	roads, runways, parking lots	ATMOS. DEP. x AREA (kg N ha/yr) (ha)
***Fertilizer Application*:**		
Turf	lawns, golf courses	APPL. RATE.[3] x AREA x F_N (kg N ha/ yr) (ha) (fract)
Agricultural Land	crop land	APPL. RATE. x AREA - N rem[4] (kg N ha/yr) (ha)
***Human Septic Wastewater*[5]:**	residential land	(kg N person/ yr) x (persons per house) x (# of houses)

[Rainfall nitrate]:	**270 ug N/L**
[Rainfall ammonia]:	**920 ug N/L**
[Rainfall dissolved organic N]:	**180 ug N/L**
[TDN]:	**1370 ug N/L**
Ave Annual Rainfall:	**123.4 cm**
Wet to Total Deposition Factor:	**1.25**
% atmos N transported from Nat'l Veg Soils:	**35%**
% atmos N transported from Turf Soils:	**38%**
% atmos N transported from Agr. Soils:	**38%**
Median Home Size:	**179 sq m**
No of stories/home:	**2**
House footprint area:	**89 sq m**
Average area of roof:	**0.00996 ha**
Average area of driveway:	**0.01254 ha**
% atmos N transported from Impervious Soils (roof/driveway):	**38%**
Fertilizer N applied to lawns:	**104 kg N/ha**
Fertilizer N applied to agriculture:	**136 kg N/ha**
Fertilizer N applied to rec/golf courses:	**115 kg N/ha**
Average lawn area:	**0.05 ha**
% of homes that use fertilizer:	**34%**
% of fertilizer N transported from Turf Soils:	**61%**
% of fertilizer N transported from Agri Soils:	**61%**
% of fertilizer N transported from Rec. Soils:	**61%**
Per capita human N excretion rate:	**4.8 kg N/pp/yr**
People per house:	**2.4**
% waste transported from septic tank/leach fields:	**60%**
% waste transported from septic plumes:	**66%**
% watershed N transported from vadose zone:	**39%**
% N transported from aquifer:	**65%**
# of houses in high density residential areas:	**8**
# of houses in medium-high density residential areas:	**6**
# of houses in medium density residential areas:	**1.33**
# of houses in medium-low density residential areas:	**0.667**
# of houses in low density residential areas:	**0.5**

Figure 4. Listing of equations (top section), variables, and magnitudes (bottom section) used in the NLM. Data sources: (Valiela *et al.* 1997, Luo *et al.* 2002). Example landuse categories are for RI. References/Notes: [1]Uses the concentration of NO_3^-, NH_4^+, and DON in local precipitation and yearly rainfall totals to generate the atmospheric deposition term. [2]Model also includes dry deposition (*i.e.*, from NO_x, particles adhering to leaves and impervious surfaces), which is adjustable as a proportion of wet deposition. [3]Uses average fertilizer addition rates for Cape Cod of 105 kg N ha/yr for lawns and 115 kg N ha/ yr for golf courses. F_N = "0.34" refers to the fraction of homeowners applying fertilizer (applies to lawns only). [4]The term "N rem" allows for nitrogen removed from the watershed as crops (*i.e.*, consumed outside of the watershed); assumed zero in current estimates. [5]As published, the NLM applies to nonpoint sources; however, some of the estuaries contain municipal point source inputs. The magnitude of these inputs was estimated from effluent monitoring data. In these cases, if point source inputs were significantly greater than estimated nonpoint source inputs, then nonpoint sources were not included.

Table 3. Nitrogen loading rate (kg N/yr) from watershed sources to each of the study embayments (after attenuation through watershed).

Embayment	ID	Watershed Wastewater	Atmospheric	Fertilizer	Point Sources (WWTF)	Atmospheric Deposition to Water Surface	Total Nitrogen Load to Embayment[1]
Branford Harbor	BHC	12,300	11,800	8,860	17,600	2,980	**41,000**
Clarks Cove	CCM	19,700	1,570	2,640	0	5,910	29,800
Cuttyhunk Pond	CPM	179	9	16	0	843	1,050
Falmouth Inner Harbor	FHM	3,060	418	388	0	264	4,130
Katama Bay	KBM	3,170	951	1,470	0	12,400	18,030
Lagoon Pond	LPM	3,460	1,430	1,520	0	4,490	10,900
Little Bay	LTM	2,230	1,460	1,280	0	1,850	6,820
Mattapoisett Harbor upper	MHM	6,740	10,700	5,680	0	5,930	29,050
Menemsha Pond	MPM	536	233	61	0	5,840	6,670
Onset Bay	OBM	9,100	1,620	1,050	0	5,480	17,200
Tarpaulin Cove	TCM	5	181	0	0	1,650	1,830
Vineyard Haven-Inner	VHM	573	120	81	0	389	1,160
Allen Harbor	AHR	1,440	529	1,570	0	664	4,210
Bonnet Shores	BSR	4,830	493	576	0	1,460	7,350
Easton Bay	EBR	29,130	3,460	5,340	0	4,230	42,200
Kickamuit River	KRR	12,870	3,220	7,720	0	4,730	28,500

WWTF = wastewater treatment facility (direct discharge to embayment)

[1]Total is the sum of all land-derived nitrogen sources. In the cases where WWTF inputs are significant (in bold), the total wastewater input is considered solely from WWTFs; otherwise watershed wastewater inputs are used.

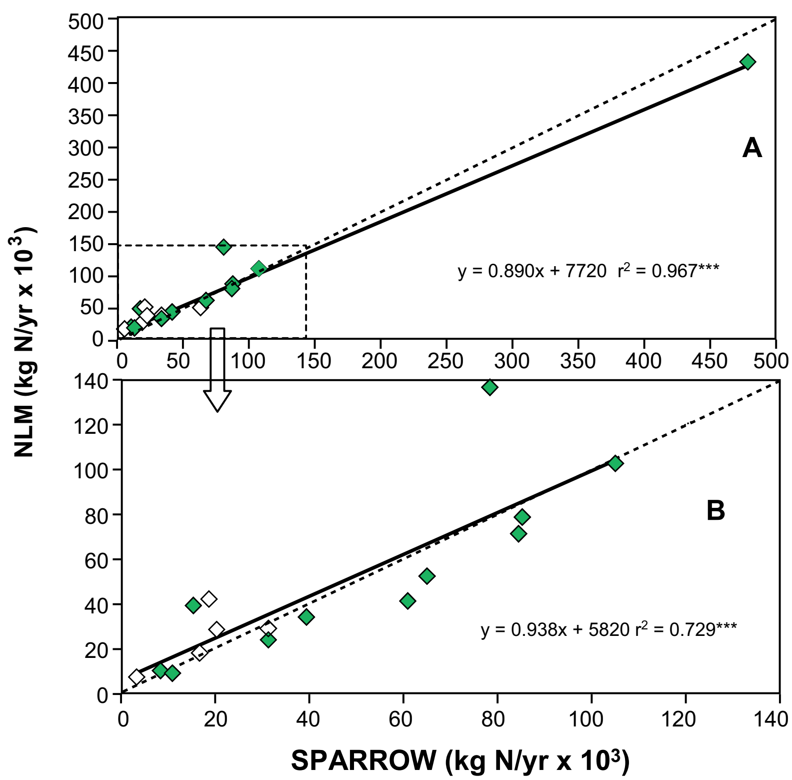

Figure 5 A. Comparison of nitrogen loading rate calculated using the NLM and SPARROW models for embayments in which both models provide estimates (includes additional systems from a larger study, open symbols represent systems in this report). B. Is the comparison with the highest loaded system removed. Dashed line represents 1:1 correspondence between the two models. ***p<0.01.

Flushing Time

Flushing time is the time required to exchange water out of an entire defined water body (Monsen *et al.* 2002). We computed the flushing time for each of the study systems using an empirical model that used literature-derived flushing time data and morphological characteristics for estuarine embayments that are similar to those in this study (see Appendix 1 for a detailed description of the methodology). The morphological variables evaluated include area, length, depth, and volume. Both linear and nonlinear (power law) regressions were used to assess which morphological characteristic best fit the flushing time data. The correlation of embayment area to flushing time was better than for the other morphological characteristics (see Appendix 1).

Flushing time is expected to be important in increasing the variance explained by the derived load-response models (Dettmann 2001, Kelly 2001, Latimer & Kelly 2003). The calculated flushing times for this study, derived by using the results of the power law equations, ranged over a factor of 3.5, from 1.5 – 5.3 days (Table 2). Finally, it should be noted that the flushing times are approximations of the average conditions - the actual flushing times are time variant and are a function of a number of factors (*e.g.*, tidal forcing, wind, etc.) that are not explicitly included. In addition, the approximations can only be applied for systems that are similar to those used to develop the algorithm.

Determination of Eelgrass Extent

The primary purpose of this study was to develop an empirical relationship between the extent of eelgrass (*Zostera marina*) and the amount of nitrogen input to shallow estuarine embayments in southern New England. During fall 2002, aerial imagery was collected in 16 study embayments from the Connecticut, Rhode Island, and Massachusetts shoreline.

The intent of characterizing ecosystem response using an empirical approach was to assign a single number describing eelgrass extent for each of these embayments. This differs from the usual goals of seagrass habitat assessment, which are generally focused on the scale of individual eelgrass beds rather than on whole-system responses among many embayments. Traditional habitat assessments look for subtle changes in the health of seagrass beds from one year to another, focusing on changes in bed density or bed area. For this study, comparability between whole embayments (over three states) is more important than detailed assessments of any one, or complex of beds, in a particular embayment. In summary, rapid methods of data acquisition and analysis are preferred because the purpose is to characterize eelgrass extent for entire estuarine embayments over a large geographic area. Thus our methods are based on 1) rapid aerial data acquisition; 2) minimal image processing; 3) a metric of extent based on linear adjacent-to-shoreline segment length, rather than bed area; and 4) the reference of image data to existing larger-scale georectifications (as opposed to georectification of each individual image). See Appendix 2 for additional details of the methodology.

The original sample design included imagery collection from 38 embayments; however some systems could not be flown due to FAA restrictions, and data from other systems were unattainable due to technical shortcomings. Additional systems will be included in future surveys.

A camera system was mounted downward-looking through a port in a Cessna Skymaster fixed wing aircraft. Target flight altitude was 1000 feet, and speed was generally maintained at 100 knots. Flights covered as much of the coastline as possible in straight-line transects, since excessive maneuvering of the aircraft resulted in added effort in post flight image analysis. Aerial imagery was collected from Oct 10th through Oct 31st 2002. As weather and surface chop dictated flight times, some concessions were made in desired conditions. Sampling and analysis protocols followed the procedures outlined by NOAA's Coastal Ocean Services Coastal Change Analysis Program (Dobson *et al.* 1995, Finkbeiner *et al.* 2001).

Images were acquired and analyzed using off-the-shelf image software, reviewed for representativeness, tiled into mosaics, and analyzed using GIS software. A set of existing digital orthophoto images was obtained as an aid in reference scaling the flight-derived images (USGS NAPP 2003, MassGIS 2005, RIGIS 2006). Eelgrass beds were identified in images based on vegetative structure and growth morphologies reported in the literature (Costa 1988).

Eelgrass extent was delineated using a simplified linear measure of the shoreline that had eelgrass present, rather than the more traditional areal extent. This method is useful when efficient assessment of the total amount of eelgrass in a large number of embayments is required. Under conditions of excess nitrogen individual eelgrass beds may lose area from the deepwater edge (Orth & Moore 1983) which may not be visible in aerial images. In addition, embayments may first lose eelgrass at the head of the estuary where nitrogen loading is greatest (Costa 1988). Since the need is to quantify vegetation extent at the scale of the entire embayment, the shoreline segment measurement is an appropriate ecosystem response measure for eelgrass.

In order to validate our linear-eelgrass-extent indicator, an existing digital map of eelgrass beds in Massachusetts (MassGIS 2005) was used to examine the relationship of the linear measurement (length of shoreline adjacent to eelgrass beds, subsequently called shoreline segment) to the more traditional bed area measure in 23 Massachusetts embayments (12 of the embayments were identical to the current study systems, 11 additional embayments were selected to increase sample size) (Pesch *et al.* Submitted). Shoreline segment was measured and summed by embayment. A significant relationship between shoreline segment and area of eelgrass beds was observed (r^2 = 0.96). This relationship followed a strong power law functionality (Figure 6). Therefore, we concluded that shoreline segment provides a good first-order assessment of extent of eelgrass for an entire embayment (Pesch *et al.* Submitted).

Results and Discussion

Eelgrass extent results are reported in Table 4. In these systems, eelgrass extent (eelgrass shoreline segment) ranged from 55 m – 4500 m (excluding non-detectable eelgrass). Expressed as a percentage of shoreline, eelgrass extent ranged from < 0.5% to 95%.

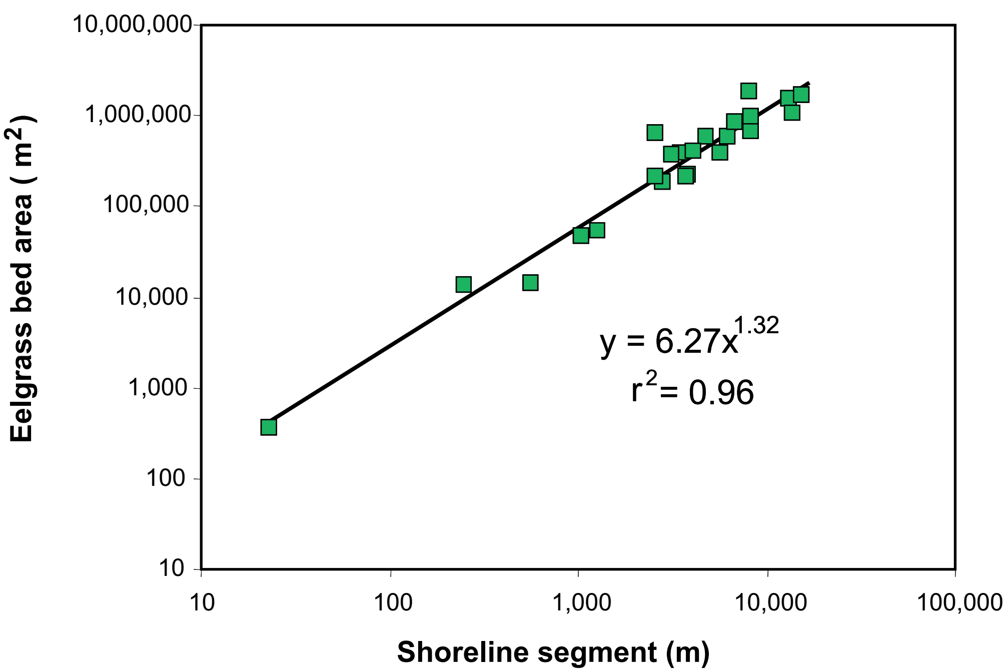

Figure 6. Relationship of eelgrass shoreline segment to bed area, for whole embayments, for 23 Massachusetts embayments (Pesch *et al.* Submitted).

Table 4. Extent indicators of eelgrass for study systems.

System Name	Eelgrass Shoreline Segment Length, m	% of Shoreline	Historical Presence[1]
Branford Harbor	55	0.3%	Yes
Clarks Cove	2,570	43%	Yes
Cuttyhunk Pond	1,880	55%	Yes
Falmouth Inner Harbor	228	9%	No data
Katama Bay	2,150	10%	Yes
Lagoon Pond	3,370	30%	Yes
Little Bay	327	4%	Yes
Mattapoisett Harbor	4,470	36%	Yes
Menemsha Pond	3,420	26%	Yes
Onset Bay	1,100	5%	Yes
Tarpaulin Cove	2,460	95%	Yes
Vineyard Haven - Inner	687	49%	Yes
Allen Harbor	0	0%	Yes
Bonnet Shores	0	0%	No
Easton Bay	0	0%	No
Kickamuit River	421	3%	Yes

[1]Based on assessment of historical maps and extant records C. Pesch EPA, personal communication.

Model Development and Refinement using Physical Characteristics of Estuaries

Empirical relationships (models) were developed between eelgrass extent and derived nitrogen loading rates using data from all embayments that had observable eelgrass. The calculated annual nitrogen loading rate was normalized using a suite of physical variables. These variables (embayment area, volume, and flushing time) were expected to be important factors contributing to the sensitivity of estuaries to anthropogenic nitrogen (Biggs *et al.* 1989, NRC 2000). We determined the extent to which these variables could be used to explain and reduce the variance in eelgrass response to nitrogen. As expected, one result of this analysis shows that there is little or no significant relationships between eelgrass extent and simple annual nitrogen loading rate (kg N/yr) to the embayments (Table 5, Figures 7A & 8A). However, eelgrass extent is significantly related to nitrogen loading rate when total embayment volumes or flushing times are taken into consideration (Figures 7C, D and 8C, D and Table 5). The extent of eelgrass, expressed as a percentage of shoreline (Figure 8), was most highly related (r^2 = 0.82 $p<0.0001$) when both flushing time and volume were used to normalize nitrogen loading rate. (Figure 8 D). These results are reasonable since the nitrogen load is likely to be processed according to the volume and flushing time of the embayments. Thus, those systems that have greater volumes and/or shorter flushing times will be able to either dilute, or export, the nitrogen and have less ability to stimulate phytoplankton or epiphytes which ultimately shade eelgrass.

Model Functionality

After exploring linear and non-linear functions, the best-fit of the data was observed using the power law functional relationship in Figure 8 D (nonlinear curve fitting SAS PROC NLIN). This implies a threshold-type behavior of eelgrass to loading rate. The observed data reveal a precipitous drop in eelgrass extent over a narrow loading window, above which eelgrass drops off more slowly. Hauxwell and others observed similar behavior for eelgrass in the Waquoit Bay embayment system (Hauxwell *et al.* 2003). In that study, eelgrass bed area was observed to drop off at a loading of 30 kg N/ha/yr, with complete disappearance at 60 kg N/ha/yr. These thresholds fall into a similar range as those for our models (*i.e.*, translated into volume and flushing time loading rates, the equivalent is estimated to be between 13 and 24 mgN/m^3). Thus, our observed load-response models yield similar responses to nitrogen, using more systems in the same class of estuaries, but along a larger geographic area.

Uncertainty Explained by the Models

If eelgrass extent were solely, or mainly, related to nitrogen loading rate from the watershed and atmosphere, then all of the variance would be explained by the constructed models. While the variance explained by the model, in which nitrogen is normalized to volume or flushing time, is striking, there is still approximately 20% of the variance not explained. This may be due to other factors that affect eelgrass ecology that are unrelated to nitrogen inputs, *e.g.*, suitable substrate, lack of seed stock due to historical wasting disease, wave action and current conditions, depth, etc. In addition, the unexplained variance can be caused by the variability inherent in the estimation of the variables that comprise the constructed models, *i.e.*, eelgrass extent, nitrogen loading rate, flushing time, and volume. For example, the static nature of the nitrogen loading rate estimations, which are based on landuse data from the mid-to-late 1990s, may not be appropriately compared to the direct measures of eelgrass extent in 2002. Furthermore, oceanic inputs of nitrogen were not considered at all. Explicit consideration of light limitation may improve the variance explained by the model. Further work needs to be done to address these issues. Nevertheless, the relationship is significant even with the assumptions and limitations of the study.

While the best-fit power law model (Figure 8D) explains approximately 80% of the variance in the data, this statistic alone does not mean that all of the uncertainty associated with the construction of the model has been quantified. Appendix 3 catalogs the assumptions and limitations of the model components. It is important that anyone using the preliminary model consider exactly how the model was derived and the assumptions and limitations upon which it was based. There are two aspects of the model that need to be highlighted here: revision and validation. The model will be revised based on at least two more years worth of eelgrass response data for the embayments noted in this report as well as additional embayments in the same class. In addition, the model will be validated by using eelgrass data from embayments not used for the construction of the empirical model. In this way, the model may be assessed for its ability to predict eelgrass extent for other southern New England embayments in the same class.

There were three embayments - Allen Harbor, Bonnet Shores, and Easton Bay - that had no detectable eelgrass. We suspect that Allen Harbor is a system that receives more nitrogen than the NLM depicts, and thus is more degraded than would be expected from the model predictions. The harbor has a large number of floating docks and moorings that host a great number of boats during the boating season. These boats are potential sources of additional nitrogen as well as direct carbon inputs from sewage. In many parts of the harbor the sediment is highly organic and suffers from severe periodic anoxia in the summer (Cicchetti *et al.* 2006). These factors suggest that Allen Harbor is too heavily loaded with nitrogen to support eelgrass – however, this is not known with certainty. Bonnet Shores and Easton Bay are two other systems that had no detectable eelgrass. Moreover, there is no evidence that these systems ever had eelgrass present (Table 4). They have the following characteristics: high wave action due to their geographic orientation (SSE and S), high mouth openness, and a hard-sand/rocky substrate characterizes much of the benthic environment (G. Cicchetti EPA, personal communication). For these reasons it is understandable that eelgrass would not be able to flourish. Therefore, Allen Harbor, Bonnet Shores, and Easton Bay were not used in the development of the load-response models.

Table 5. Statistical summary values (r^2, a, and b) from a non-linear fit of a power law function of the load-response data when loading is normalized using different physical variables.

Variable used to Normalize Load	Loading Rate Units	r^2	a	b	Figure
Eelgrass Shoreline Segment length (m)					
No normalization	kg N/yr	0.008	1070	-0.056	7A
Embayment area	kg N/ha/yr	0.230	10900	-0.440	7B
Embayment volume	g N/m^3/yr	0.346	2650	-0.389	7C
Embayment flushing time/volume	mg N/m^3	0.176	4120	-0.260	7D
Eelgrass as % of shoreline					
No normalization	kg N/yr	0.415	1690	-0.473	8A
Embayment area	kg N/ha/yr	0.331	552	-0.742	8B
Embayment volume	g N/m^3/yr	0.688*	52.5	-0.817	8C
Embayment flushing time/volume	mg N/m^3	0.819*	280	-0.793	8D

*significant at p<0.001

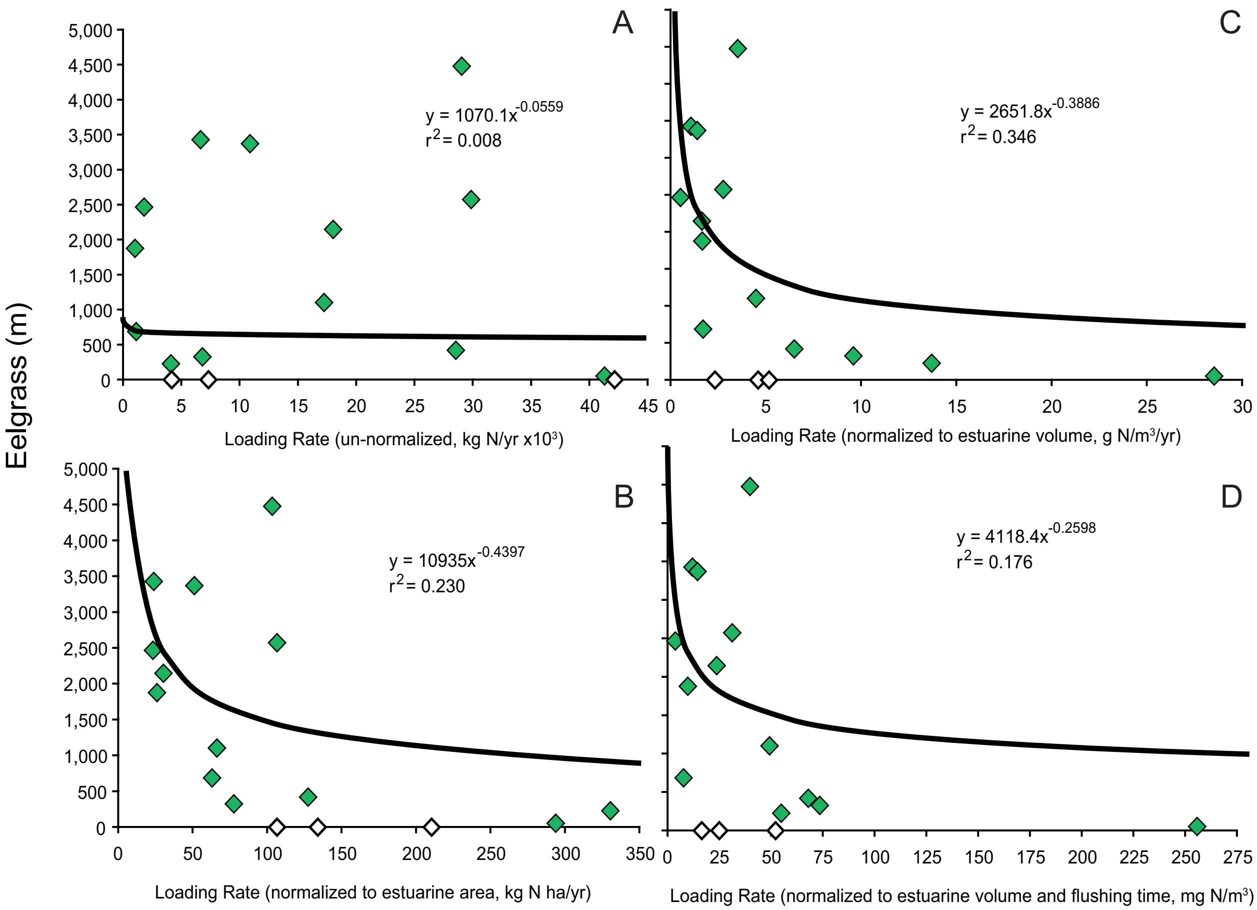

Figure 7. Graphical representations of the relationships between nitrogen load, normalized with important physical variables, and eelgrass extent (sum of shoreline segments, m): A un-normalized load, B normalized to embayment area, C normalized to embayment volume, and D normalized to flushing time/volume. Open symbols indicate embayments with no detectable eelgrass (these were not included to derive equations).

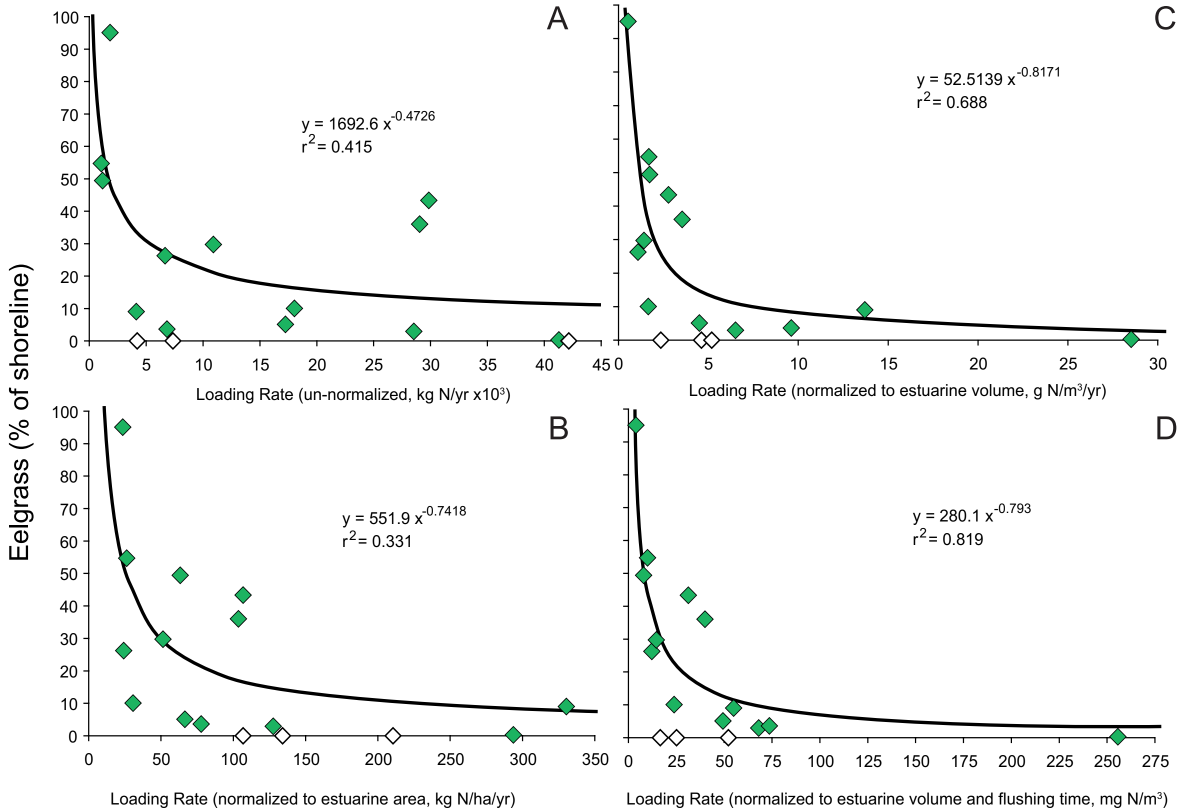

Figure 8. Graphical representations of the relationships between nitrogen load, normalized with important physical variables, and eelgrass extent (as % of total embayment perimeter): A un-normalized load, B normalized to embayment area, C normalized to embayment volume, and D normalized to flushing time/volume. Open symbols indicate embayments with no detectable eelgrass (these were not included to derive equations) only C and D were significant, $p<0.001$.

Conclusions

The derived relationships, or models, presented in this report provide evidence to environmental managers that eelgrass extent is quantitatively related to nitrogen loading inputs (properly normalized), and can aid in the development of critical nitrogen load rate limits protective of eelgrass habitat. The nitrogen load-eelgrass response model needs to be considered preliminary, because it is based on data obtained in only one year. We plan to collect data over multiple years to improve the robustness of the model and to assess inter-annual and inter-system variability. In addition, we are planning validation steps to determine the ability of the model to predict eelgrass extent for embayments not used for model development. Nevertheless, the model is compelling. Moreover, it is similar to that found for embayments of Waquoit Bay on Cape Cod, Massachusetts (Hauxwell *et al.* 2003) but greatly extends the geographic area of applicability.

Our results provide the following insights relevant to state and national efforts to derive biocriteria and nitrogen load limits for coastal embayments:

- Data for system-level ecological response variables (*e.g.*, eelgrass extent), can yield useful nitrogen load-response models through an empirical multiple-system comparative approach when important physical variables are incorporated as normalizing factors (*e.g.*, volume, flushing time).
- The load-response model derived in this study corroborates eelgrass response behavior reported elsewhere in the literature.
- Eelgrass extent, measured as length along the shoreline and expressed as percent of shoreline, is a useful system-wide metric for derivation of nitrogen loading rate limits across multiple embayments within the same estuarine class.

- Eelgrass extent for shallow embayments along the southern New England coast is highly sensitive to land-derived inputs of nitrogen (including atmospheric inputs).
- Load-response models can be used in a management context to derive critical loading limits protective of estuarine water quality.

Implementation Issues

The preliminary nitrogen load-eelgrass extent model derived in this study should be understood as a model that estimates what nitrogen levels are necessary to produce water quality conditions favorable for eelgrass. The model is based on empirical data (measured eelgrass extent) and associated estuarine nitrogen inputs and normalization factors (volume and flushing time). Further, the model is based on a system-level metric of existing eelgrass extent in relation to nitrogen inputs; it does not provide information about the trajectory of loss, or about eelgrass recovery. Eelgrass recovery may follow a different trajectory than loss, because eelgrass creates an environment favoring its growth, while the lack of eelgrass may create an environment that is less hospitable. Eelgrass extent may not increase even with the estimated nitrogen load reductions due to water quality factors other than nitrogen loading (*e.g.*, temperature effects), sediment quality, lack of seed availability, or rhizome expansion, and more.

We emphasize that the eelgrass segment indicator is method that is only useful at the whole embayment scale (*i.e.*, a system-level metric); assessment of individual beds using this indicator is inappropriate. The assessment of eelgrass extent using this metric is not meant to be used for overall state eelgrass habitat assessment, rather only in the context of predictions based on the empirical load-response model. Management officials should be aware that eelgrass segment based biocriteria may be less sensitive to changes in nitrogen inputs than other more traditional assessment methods (bed area) or newer assessment methods (bed density). The eelgrass segment method is suitable, at the system level, to determine improvement or degradation as predicted from the empirical model.

Activities / Future Research

The following are a list of additional activities that will refine and improve the current model:

- Continued ground-truthing activities for eelgrass and other submerged aquatic vegetation;
- Additional compilation of data on eelgrass extent to evaluate inter-annual and inter- system variation;
- Evaluation of other classification factors for use in minimizing variance in the load-response (*e.g.*, light limitation as expressed as area or volume of embayment above a critical depth (2-3 m));
- Evaluation of other variables that affect vegetation distributions such as habitat characteristics;
- Evaluation of other system level eelgrass indicators (*e.g.*, aerially derived eelgrass density);
- Evaluation of the magnitude and effect of nitrogen loading from oceanic sources;
- Validation of the load-response model using embayments not used to develop the preliminary model;
- Development of similar models for other classes of estuary.

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Appendix 1
Estimation of Flushing Time

Empirical Estimation of Water Residence Times
of Non-Riverine Embayments

Edward H. Dettmann

December, 2002
Revised July 2006

U.S. Environmental Protection Agency
Office of Research and Development
National Health and Environmental Effects Research Laboratory
Atlantic Ecology Division
27 Tarzwell Drive
Narragansett, Rhode Island 02882

Introduction

This study is a component of the Aquatic Stressors Nutrient Program at the Atlantic Ecology Division of the National Health and Environmental Effects Research Laboratory of the U.S. Environmental Protection Agency's Office of Research and Development. This program is developing response relationships between nutrient loading to estuaries, and ecological responses such as reductions in dissolved oxygen concentrations, loss of seagrasses, and shifts in the composition or functioning of food webs. Preliminary consideration of factors that affect estuarine response to nutrient loading has determined that water residence time in estuaries plays a crucial role in this regard.

The purpose of this study is to develop a method for empirically estimating water residence times for small to medium-sized estuaries in southern New England, and to use this method to estimate the residence time of the estuaries being studied by the Nutrient Project.

The approach taken in this study was suggested by an analysis of data for a group of eleven North American and European estuaries. Regression of mean annual freshwater residence times versus estuary area for these systems showed that there appears to be a power-law relationship between these two variables (Figure A1-1). Symbols that identify the estuaries in Figure A1-1, and data sources, are given in Table A1-1. While there is a strong relationship between estuary area and water residence time, note that the residence times for six of the eleven estuaries differ by more than a factor of two from that given by the regression line. Note that the residence times of many of these estuaries vary with freshwater inflows; the residence times given in Table A1-1, and used in Figure A1-1, are mean annual values.

One might expect that freshwater residence time would increase with system size since the amount of time that it takes fresh water to travel through an estuary depends on the distance that it must travel. Similarly, flushing of the system by tidal action might also be expected to be less efficient for larger estuaries with linear dimensions that are large compared with the mean tidal excursion than for small ones. Perhaps the most surprising aspect of this relationship is that it is as good as it is, given the number of factors that can affect flushing. The sample of estuaries used in the above analysis had a wide range of sizes, tidal ranges, and depths.

Somewhat analogous approaches have been used elsewhere for other system types. System morphology has been used to estimate water residence times in microtidal embayments bordering

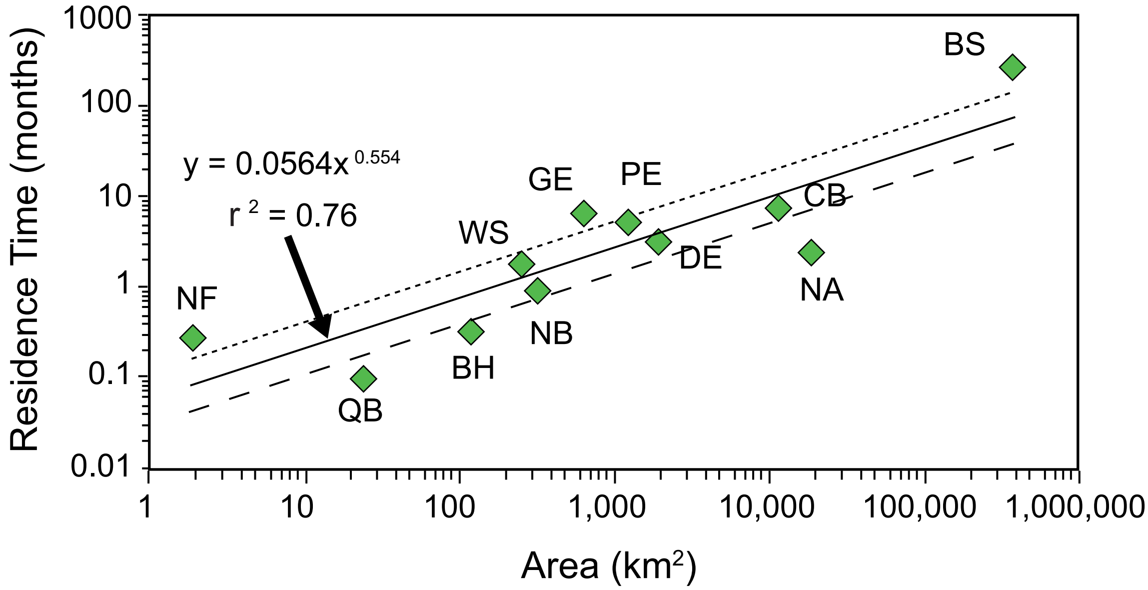

Figure A1-1. Water residence time vs. surface area for 11 North American and European Estuaries. Symbols identifying estuaries are defined in Table A1-1. The solid line is the regression line; the dashed lines give 0.5 and 2 times the value given by the regression line.

Table A1-1. Estuaries appearing in Figure A1-1, with identifying symbols and data sources.

Estuary	Symbol	Data Source
Norsminde Fjord (Denmark)	NF	Area: Nielsen *et al.* (1995); Residence time: Nielsen *et al.* (1995) and K. Nielsen (personal communication)
Ochlockonee Bay	OB	Seitzinger (1987)
Boston Harbor	BH	Signell (1992), Signell and Butman (1992)
Narragansett Bay	NB	Pilson (1985)
Guadalupe Estuary	GE	D. Brock (personal communication), Residence time is average of values for 1984 and 1987.
Westerschelde (Belgium, Netherlands)	WS	Soetaert and Herman (1995)
North Adriatic Sea	NA	Degobbis *et al.* (1986)
Delaware Estuary	DE	Area: NOAA (1985), Residence time: Polis and Kupferman (1973)
Potomac Estuary	PE	Area: Boynton *et al.* (1995), Residence time: estimate by W. Boicourt, (personal communication)
Chesapeake Bay	CB	Area:Lippson *et al.* (1973), Residence time: Nixon *et al.* (1996)
Baltic Sea	BS	Wulff and Stigebrandt (1989)

the Baltic Sea. For instance, Håkanson *et al.* (1986) developed empirical models for systems with areas 1–100 km^2 based on embayment morphometry. Their best model depended on the mean width of the embayment (W) and the topographic openness (or exposure) of the embayment (E) to calculate the residence time of the surface water. They calculated the topographical openness from the relationship between the width of the seaward boundary of the embayment at five water depths and the length of the contour line at each of these depths.

Persson *et al.* (1994) developed models for embayments with areas 1–150 km^2 that had good predictive capabilities, and that depend only on the topographic openness, defined in this case as the ratio of the cross-sectional area at the seaward boundary and the bottom area.

Data Sources and Methods

It was decided to continue this analysis with data for systems that are more similar in size and other characteristics to those for which we wish to estimate residence times. The method employed is similar to that described above. A sample of New England estuaries for which there have been determinations of residence time was assembled, and regression relationships obtained between residence time and estuary area.

The embayments used to develop relationships between area and water residence time are listed in order of increasing surface area in Table A1-2. These data were obtained from the papers and reports cited in the last column of the table. The symbols defined in the second column of the table are used to identify the embayments associated with data points in some figures. Surface areas of these embayments are in the range 0.23–328 km^2. Residence times are given in units of both days and months, and are between 2 and 26 days.

The water residence times listed in Table A1-2 were determined by various methods. The freshwater replacement method was used to determine long-term average residence times in Greenwich Cove and Greenwich Bay (Granger *et al.*, 2000). The freshwater replacement method was also used to determine freshwater residence time in Narragansett Bay as a function of freshwater inflow rate (Pilson, 1985). The freshwater residence time in Narragansett Bay varies between approximately 10 days at high inflow rates to approximately 40 days at low inflow rates. The value of 26 days given in Table A1-2 is for the long-term mean inflow flow rate (105 $m^3\ s^{-1}$). The residence time for Boston Harbor was determined by a model analysis (Signell, 1992; Signell and Butman, 1992), and represents the e-folding time, the time required for the mass of a uniformly-distributed conservative tracer in the estuary to decrease to 37% (e^{-1}) of its original amount. The residence time was determined to be variable, depending on tidal and wind influences, with 10 days being the midpoint of the range 8-12 days determined to be the best estimate. The residence time for Broad Cove, Sunset Cove, Allen's Pond, and Little Bay were determined using dye studies and the freshwater replacement method (Geyer *et al.* 1997). Determinations using dye studies approximate the estuary residence time, while the calculations using the freshwater replacement method give the freshwater residence time. The results of the two methods agreed to within 0–13 percent for Broad Cove, Sunset Cove, Onset Bay, and Allen's Pond; for Little Bay, the dye study gave the larger of the two values, which exceeded the average of the two values by 48 percent (Geyer *et al.* 1997). The average of the results of the two methods is given in Table A1-2.

Residence times were available for a few more local estuaries, but were not included in this analysis, either because they are microtidal or very shallow, and therefore judged to be sufficiently different from the estuaries in our study that flushing may well be governed by different processes.

The embayments for which residence times are to be determined are listed in Table A1-3 with values for morphological parameters: areas, maximum lengths, mean depths, and volumes. The values of the morphological parameters were determined by Michael Charpentier (Computer Sciences Corporation) using GIS techniques, based on system boundaries determined by members of the Nitrogen Project Team (Mohamed Abdelrhman, Giancarlo Cicchetti, Edward Dettmann, and Jim Latimer). The surface areas of these estuaries vary between 0.2 km^2 (Fort Wetherill Coves) to 605 km^2 (Buzzards Bay).

Table A1-2. Embayments used to develop area-residence time relationships.

Embayment	Symbol	Surface Area (km^2)	Residence Time (days)	(months)	Reference for Residence Time
Broad Cove (MA)	BC	0.23	2.0	0.0657	Geyer *et al.* (1997)
Sunset Cove (MA)	SC	0.34	1.8	0.0591	Geyer *et al.* (1997)
Allen's Pond (MA)	AP	0.64	2.95	0.0969	Geyer *et al.* (1997)
Little Bay (MA)	LB	0.74	1.45	0.0476	Geyer *et al.* (1997)
Mumford Cove (CT)	MC	1.0	3.5	0.115	French *et al.* (1989)
Greenwich Cove (RI)	GC	1.0	3.9	0.128	Granger *et al.* (2000)
Onset Bay (MA)	OB	2.1	3.85	0.126	Geyer *et al.* (1997)
Greenwich Bay (RI)	GB	11.8	7	0.23	Granger *et al.* (2000)
Boston Harbor (MA)	BH	125	10	0.33	Signell (1992), Signell and Butman (1992)
Narragansett Bay (RI)	NB	328*	26	0.854	Pilson (1985)

* The area determined by Pilson (1985) does not include the Sakonnet River, and the Sakonnet River was not included in his calculations of residence time.

Measurements have been made of residence times in a number of these embayments: Acushnet River (New Bedford Inner Harbor, Apponaug Cove, Greenwich Bay, Greenwich Cove, Narragansett Bay, Providence-Seekonk River, and Warwick Cove. The Providence River is a riverine estuary for which the mechanisms governing flushing are probably significantly different from those for most of the other systems. The same is probably true for the Pawcatuck River.

Results

Both linear and nonlinear regression was used with the data for local embayments listed in Table A1-2. The results of nonlinear regression with a power law function are shown in Figure A1-2. The symbols identifying embayments are defined in Table A1-2. Overall most points are close to the regression line, even the data point for Little Bay, the one farthest from the line, is less that a factor of two distant. The line in Figure A1-3 shows the linear regression to the data. The curvature of the line is attributable to the fact that the abscissa is logarithmic, in order to show all data points legibly in the plot. All data points except that for Little Bay are within a factor of two of the linear regression line. Overall, both regressions seem to represent the data well. However, the linear regression has a minimum value (y intercept) of 3.02 days. This seems unrealistic, since some small systems would be expected to have shorter residence times (Broad Cove, Sunset Cove, and Allen's Pond, and Little Bay do).

Both regressions were used to calculate residence times for the Nitrogen Project's study systems. Residence times calculated with the power law and linear equations are listed in Table A1-4, as well as the average of these two values, and the difference between the two values. Also listed in Table A1-4 are measured values for the few systems for which they exist.

Table A1-3. Estuaries for which residence times are to be determined.

Embayment	State	Total Estuarine Area (km^2)	Maximum Length (m)	Depth (m) (Area Weighted Mean)	Estuarine Volume (m^3)
Black Rock Harbor	CT	1.10	3,330	2.4	2,592,842
Branford Harbor	CT	1.41	4,475	1.0	1,451,460
Greenwich Cove	CT	2.21	3,304	1.5	3,365,874
Hammonasset R/Clinton Harbor	CT	2.73	5,457	0.6	1,635,647
Niantic River	CT	3.29	5,712	1.8	5,865,096
Pawcatuck River Estuary	CT	2.64	7,901	4.5	11,921,409
Acushnet River (NB inner harbor)	MA	4.06	7,141	3.4	13,647,557
Buzzards Bay	MA	605	50,195	10.4	6,267,250,920
Clarks Cove	MA	2.79	2,352	3.8	10,730,342
Cuttyhunk Pond	MA	0.40	949	1.6	621,857
Falmouth Inner Harbor	MA	0.12	1,113	2.4	301,163
Hadley Inner Harbor	MA	0.08	471	2.3	188,494
Katama Bay	MA	5.88	6,393	1.8	10,886,181
Lagoon Pond	MA	2.12	3,694	3.6	7,687,689
Lewis Bay	MA	4.77	3,217	1.4	6,481,274
Little Bay	MA	0.88	2,068	0.8	710,154
Mattapoisett Harbor upper	MA	2.81	2,710	2.9	8,192,927
Megansett Harbor	MA	2.26	3,170	2.1	4,675,521
Menemsha Pond	MA	2.76	3,144	3.9	10,656,051
Onset Bay	MA	2.59	4,091	1.5	3,838,143
Phinneys Harbor	MA	1.87	2,178	2.6	4,813,594
Sippican Harbor u&l	MA	7.28	6,139	3.0	21,481,455
Tarpaulin Cove	MA	0.78	822	4.4	3,422,848
Vineyard Haven-Inner	MA	0.18	343	3.7	673,019
Allen Harbor	RI	0.31	819	2.9	913,344
Apponaug Cove	RI	0.43	1,627	1.2	533,421
Bonnet Shores	RI	0.69	854	4.5	3,123,927
Bristol Harbor	RI	2.06	2,399	4.6	9,446,200
Coggeshall Point Harbor	RI	0.05	281	3.5	183,693
Easton Bay	RI	2.00	1,538	4.1	8,131,343
Fort Wetherill Cove - West	RI	0.02	232	7.8	138,452
Fort Wetherill Cove-Unnamed	RI	0.02	285	7.0	171,992
Great Salt Pond	RI	2.52	3,469	3.6	9,109,860
Greenwich Bay*	RI	12.0	5,174	2.5	29,747,801
Greenwich Cove	RI	0.75	2,577	1.8	1,360,725
Kickamuit River	RI	2.24	4,248	2.0	4,384,402
Mackerel Cove	RI	0.86	1,605	5.2	4,509,450
Mt. Hope Bay	RI	51.1	32,975	4.6	236,347,166
Narragansett Bay	RI	411	54,066	8.3	3,423,467,008
Old Harbor	RI	0.08	256	1.5	122,260
Potter Cove	RI	0.40	876	2.3	917,918
Providence-Seekonk River	RI	23.8	19,733	4.2	98,926,231
Sakonnet Harbor	RI	0.10	357	2.1	202,803
Warwick Cove	RI	0.56	2,203	1.5	824,373

* Area Includes Apponaug, Greenwich and Warwick Coves.

Discussion

Overall, both linear and nonlinear regression gave fits to the data for 10 systems for which there were measured residence times. The data used to derive these regressions included measurements of both freshwater residence time and estuary residence time, and the results should not be interpreted as distinguishing between these two measures. It should be noted, however, that measurements for five of these systems (Broad Cove, Sunset Cove, Onset Bay, Little Bay, and Allen's Pond) were measured with two techniques that would be expected to yield the freshwater residence time, and that these two values were usually in close agreement.

The linear regression equation gives a minimum residence time of approximately 3 days, and is therefore probably not reliable for very small systems. When the linear and power-law regressions were used to calculate residence times for the Nitrogen Project's study systems, the results given by the two methods were generally in good agreement. There are two notable exceptions: Buzzards Bay (25 vs. 45 days), and Narragansett Bay (22 vs. 32 days). Interestingly, the average of the two values for Narragansett Bay was close to the measured value. The area of Buzzards Bay is outside the range of values used in the regression.

The Acushnet River (New Bedford Inner Harbor) was not used to develop the regressions. This is because one determination of this value (3.2 days) by Abdelrhman (2002) was for river inflow rates that are known to exceed the mean river flow rates. The other reason is that a second study conducted under contract to USEPA Region 1 has produced some values that exceed this by a factor of about five. These results are still under review, and have not been made available to us yet for detailed study. In view of this confusing picture, New Bedford Harbor was not included in the base data set used to develop the regressions. The regression results (4.65 and 3.30 days) are close to the value of 3.2 days found by Abdelrhman (2002).

There are three approaches that we can take in applying these results. We could use either of the methods, or we could use the average of the two. Given the fact that both equations give good fits overall, and the fact that the linear equation gives a minimum result of about 3 days, I think it advisable to use the use the power law approach, at least for smaller systems.

Table A1-4. Study systems with calculated residence times, using both linear and nonlinear formulas. Differences between selected total estuarine areas given in this table and in Table 2 are attributable to differences in choice of estuary boundaries.

Embayment	State	Total Estuarine Area (km^2)	Residence Time Power Law (days)	Residence Time Linear (days)	Residence Time Average (days)	Difference (days)	Measured Residence Time (days)
Black Rock Harbor	CT	1.10	3.02	3.10	3.06	-0.08	n.a.
Branford Harbor	CT	1.41	3.27	3.12	3.20	0.15	n.a.
Greenwich Cove - CT	CT	2.21	3.80	3.18	3.49	0.63	n.a.
Hammonasset R./Clinton Hbr.	CT	2.73	4.07	3.21	3.64	0.86	n.a.
Niantic River	CT	3.29	4.34	3.25	3.79	1.09	n.a.
Pawcatuck River Estuary	CT	2.64	4.03	3.21	3.62	0.83	n.a.
Acushnet River (NB inner hbr)[a]	MA	4.06	4.65	3.30	3.98	1.35	3.20
Buzzards Bay	MA	605	24.50	45.37	34.93	-20.86	n.a.
Clarks Cove	MA	2.79	4.11	3.22	3.66	0.89	n.a.
Cuttyhunk Pond	MA	0.40	2.15	3.05	2.60	-0.90	n.a.
Falmouth Inner Harbor	MA	0.12	1.46	3.03	2.25	-1.57	n.a.
Hadley Inner Harbor	MA	0.08	1.27	3.03	2.15	-1.75	n.a.
Katama Bay	MA	5.88	5.26	3.43	4.35	1.83	n.a.
Lagoon Pond	MA	2.12	3.75	3.17	3.46	0.58	n.a.
Lewis Bay	MA	4.77	4.90	3.35	4.13	1.55	n.a.
Little Bay[b]	MA	0.88	2.80	3.08	2.94	-0.29	1.45
Mattapoisett Harbor upper	MA	2.81	4.11	3.22	3.66	0.90	n.a.
Megansett Harbor	MA	2.26	3.83	3.18	3.50	0.65	n.a.
Menemsha Pond	MA	2.76	4.09	3.21	3.65	0.88	n.a.
Onset Bay[b]	MA	2.59	4.01	3.20	3.60	0.81	3.85
Phinneys Harbor	MA	1.87	3.59	3.15	3.37	0.44	n.a.
Sippican Harbor u&l	MA	7.28	5.65	3.53	4.59	2.12	n.a.
Tarpaulin Cove	MA	0.78	2.69	3.07	2.88	-0.39	n.a.
Vineyard Haven - Inner	MA	0.18	1.66	3.03	2.35	-1.37	n.a.
Allen Harbor	RI	0.31	1.99	3.04	2.51	-1.05	n.a.
Apponaug Cove[c]	RI	0.43	2.20	3.05	2.63	-0.85	0.7
Bonnet Shores	RI	0.69	2.58	3.07	2.82	-0.49	n.a.
Bristol Harbor	RI	2.06	3.71	3.16	3.44	0.55	n.a.
Coggeshall Point Harbor	RI	0.05	1.10	3.02	2.06	-1.93	n.a.
Easton Bay	RI	2.00	3.68	3.16	3.42	0.52	n.a.
Fort Wetherill Cove - West	RI	0.02	0.77	3.02	1.89	-2.25	n.a.
Fort Wetherill Cove - Unnamed	RI	0.02	0.85	3.02	1.94	-2.17	n.a.
Great Salt Pond	RI	2.52	3.97	3.20	3.58	0.77	n.a.
Greenwich Bay[c,d]	RI	12.0	6.67	3.86	5.27	2.81	7
Greenwich Cove-RI	RI	0.75	2.66	3.07	2.86	-0.42	3.9
Kickamuit River	RI	2.24	3.82	3.18	3.50	0.64	n.a.
Mackerel Cove	RI	0.86	2.78	3.08	2.93	-0.30	n.a.
Mt. Hope Bay	RI	51.1	10.78	6.59	8.69	4.19	n.a.
Narragansett Bay	RI	411	21.56	31.82	26.69	-10.27	26
Old Harbor	RI	0.08	1.27	3.03	2.15	-1.76	n.a.
Potter Cove	RI	0.40	2.15	3.05	2.60	-0.90	n.a.
Providence-Seekonk River[e]	RI	23.8	8.37	4.69	6.53	3.68	3.6
Sakonnet Harbor	RI	0.10	1.35	3.03	2.19	-1.67	n.a.
Warwick Cove[c]	RI	0.56	2.42	3.06	2.74	-0.64	4.2

[a] Source: (Abdelrhman, 2002)
[b] Source: (Geyer *et al.*, 1997)
[c] Measured freshwater residence time from Granger *et al.* (2000).
[d] Area includes Apponaug, Greenwich, and Warwick Coves.
[e] The Providence-Seekonk River estuary is a river-dominated system; this calculation method is not appropriate for this estuary. Measured residence time from Asselin and Spaulding (1993).

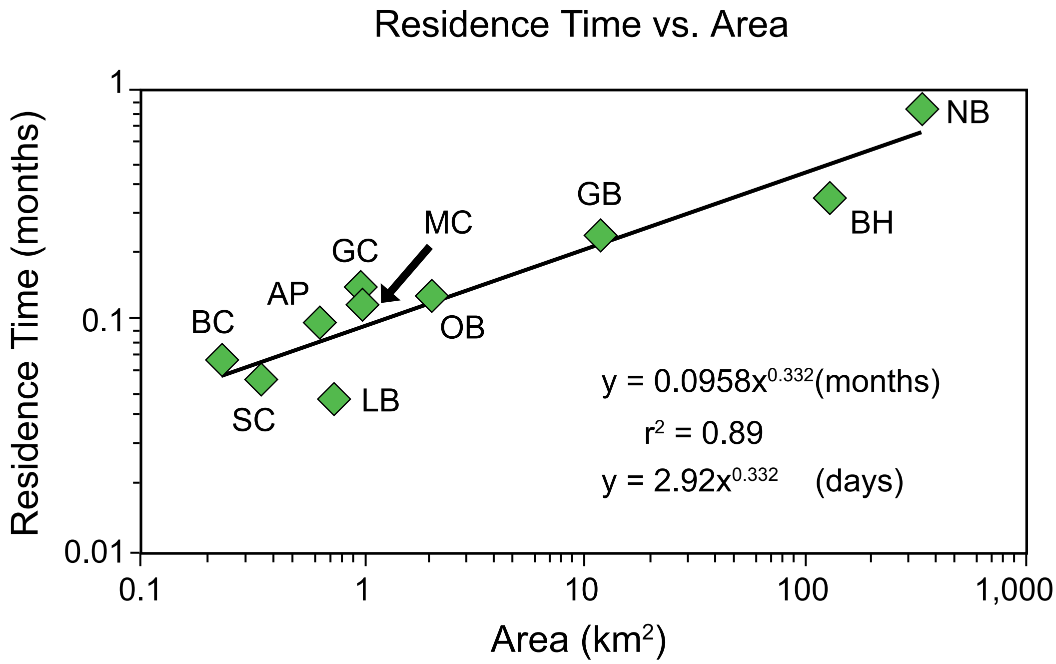

Figure A1-2. Water residence time vs. surface area for the 10 local estuaries listed in Table A1-2. Symbols are defined in Table A1-2. The line is the power law fit to the data using nonlinear regression. The regression equation is expressed in both monthly and daily time units.

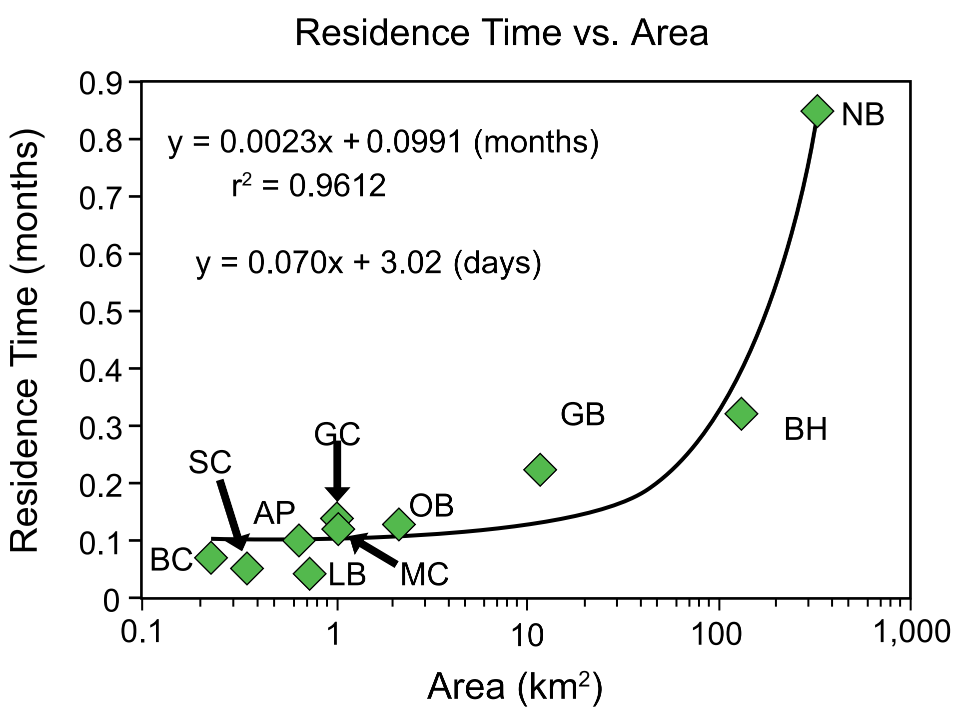

Figure A1-3. Water residence time *vs.* surface area for the 10 local estuaries listed in Table A1-2. Symbols are defined in Table A1-2. The line is the linear regression to the data. The regression equation is expressed in both monthly and daily time units.

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Boicourt, W., Horn Point Laboratory, University of Maryland, P.O. Box 775, Cambridge, MD 21613.

Brock, D. A., Environmental Section, Texas Water Development Board, P.O. Box 13231, Austin, TX 78711-3231, USA.

Nielsen, K., Ministry of Environment and Energy, National Environmental Research Institute, Silkeborg, Denmark.

Appendix 2
Method for Determination of the Spatial Extent of Submerged Vegetation

Steven Rego

Our primary goal in characterizing the extent of submerged vegetation is to assign a single number describing seagrass abundance in shallow well-flushed embayments in southern New England. Our seagrass goal differs from the usual goals of seagrass assessment, which are often focused on single beds rather than on entire systems, and which often look for subtle changes in seagrass health from one year to another, focusing on changes in bed density or bed area. For our purposes, comparability among systems across states is more important than is detail of assessment in any one bed or system. In fact, a method insensitive to subtle year-to-year variability would be preferable for our purposes. Since our needs are to characterize seagrass in entire systems over a large geographic area, rapid methods of data acquisition and analysis are greatly preferable. For these reasons, our methods are based on 1) rapid aerial data acquisition; 2) minimal image processing; 3) linear adjacent-to-shoreline (segment length) habitat analysis rather than bed areal analysis; and 4) reference of image data to existing larger-scale georectifications (as opposed to georectification of each individual image.) We feel that this approach will deliver excellent data that will meet our needs, will be appropriately rapid to use, and will be easily exported to other researchers with comparable goals.

Camera system:

The camera system consisted of a 1.4 megapixel progressive scan (Sony® DFW-SX900), color camera mounted downward through a port on the aircraft. The camera was outfitted with a rectilinear wide-angle (90 degree) Schneider® lens. Imagery was collected with approximately 30-60% image overlap using a 0.5 FPS collection rate. Data was streamed through a firewire port to an integrated laptop (Sony® Vaio GR270) and stored on two external 150GB hard drives connected via IEEE-1394 firewire hub. Imagery was collected using camera control and image acquisition software (Streampix® SpectraServices Inc.). Approximate file storage sizes ranged from 75-350MB / system and total images per system ranged 345-1193. Imagery collected using streampix was stored in a proprietary format (SEQ files). Individual frames were then viewed and exported individually or played in sequence during image processing. Imagery pixel scale was approximately 0.15 meters (6 inches). Figure A2-1 shows an example image showing submerged vegetation.

Aerial survey:

Each flight was characterized by a ‘pre-flight’ review in which sampling systems were reviewed by the pilot for “best flight capability”. Our camera system was mounted on Cessna Skymaster® aircraft. Optimal flight altitude was 1000 ft., as flying any lower raises concerns security concerns in our region and permitting issues with the FAA. Speed was generally maintained at 100 knots. Flights were scheduled during the biomass season (early fall) but did not occur until October 10th through 31st during a 5 day sampling window. As weather and surface chop dictate flight times some concessions must be made to desired conditions. Our flight / system conditions are listed below (Finkbeiner, 2001):

- Desired 30% min. image end lap
- Desired sun angle between 30 and 45 degrees
- Desired wind less than 10 knots
- Desired tide stage within approximately 2 hours of low tide.
- Desired cloud cover less than 5%.

Image processing and analysis:

Images acquired using the Streampix® software system was stored in a native file format (SEQ). This format stores individual flight frames (0.5 / sec) for viewing later either in video or frame by frame format. Images were reviewed directly on the Sony® Vaio GR270 laptop using the Streampix® software. This initial review was performed to identify only images taken of coastal areas of the systems in question, other frames, inland and over-water imagery taken during aircraft maneuvering were ignored (See Figure A2-1). Images of interest were identified by frame number during review and later retrieved for export to a lossless generic image format (TIFF – tagged image file format) (Figure A2-2). These images, due to the great deal of overlap, were easily tiled into mosaics to speed analysis. For measurement purposes only, a second set of imagery (1995 orthophotography) was obtained for each area where available from state agencies. Image analysis and measurement was performed using ArcView® ESRI version 3.2 and Photoshop V10. Scaling coefficients were established for imagery using the following process:

(1) a clearly identified feature occurring in both images was identified and measured on the orthophoto (m) and in the collected imagery (pu – page units).

(2) a scale ratio (total length in meters of target / total length in pu of target) was calculated to develop a coefficient for length between the registered and un-registered imagery.

Submerged aquatic vegetation (SAV) and other vegetative features were identified in the image mosaics. SAV was delineated along the coastline and measured in page units in Photoshop®. Lengths were calculated in tables using the scale coefficients. This technique was checked for accuracy using identifiable ground control features in each image. Imagery suffering from distortion due to severe axial tilt was not used in the final analysis.

Vegetation Identification:

A brief literature review of seagrass identification techniques showed that visual identification from aerial imagery was vague. Image interpretation depended on the interpreter's experience and image quality. Our guidance document provided the basis for our interpretative analysis and was drawn from Eelgrass in Buzzards Bay: Distribution, Production and Historical Changes in Abundance (Costa 1988). Although a guide, the report does not provide hard factual features to assist with SAV or botanic identification; rather, they state the photographic images of habitat features vary in ways that cannot readily be modeled, described or communicated. This is in part due to the temporal and geographic variations seen in vegetative beds.

Tiered System for Identification of Submerged Aquatic Vegetation:

A tiered approach was used for the identification of submerged aquatic vegetation (Table A-1). The first tier represents the initial assessment of the digital image, if the factors were not definitive, factors noted in the second tier were evaluated to obtain vegetative identification (see decision rules noted below). Figure A2-3 shows an example data sheet for a single embayment.

The following decision rules were applied to the tiers:

- In the absence of any contradictory features, any Tier 1 identifier alone constitutes a habitat identification.
- In the absence of any contradictory features, any two Tier 2 identifiers constitute a habitat identification.
- If contradictory features are found, this may possibly be due to 1) several visual signals coming from two or more mixed habitat types in the same area; 2) an unusual arrangement of a single habitat; or 3) artifacts causing problems of interpretation.
- In some cases, further clues will lead to resolution of these contradictory features - - for instance, heavily epiphitized seagrass can have visual features in common with both seagrass and macroalgae. Where seagrass is mixed with another vegetated habitat such as epiphytic algae, macroalgae, or rockweed, the amalgam is categorized as a "mixed" bed descriptor.
- Where conflicting evidence exists, contradictory features in Tier 2 can be cancelled against each other and Decision Rules 1 or 2 above applied to the balance of features to classify habitat types. If contradictory features or lack of evidence result in classification information such that none of the Decision Rules are met, and this appears not to be due to habitat mixing, then the habitat is classified as "Unidentifiable Vegetation" or "Unidentifiable Habitat".

Target Species	Tier	Descriptor Abbreviation	Identifying Remark	Description
Eelgrass	The following image features were used to identify dark subtidal photographic features as seagrass.			
	I	PPS	Propeller scars	long thin light colored scar lines on a dark background
		MCS	Mooring chain scars	circular light colored scar areas on a dark background around moorings, where swing of the mooring chain has scoured circular area
	II	CGF	Circular Growth Form	patches circular in shape
		GTX	Grainy texture	large grains seen in above
		SEG	Sharp edges	beds without the blurry smeared edges not uncommon in macroalgal habitat
		STG	Subtidal growth pattern	very little growth in the intertidal or very shallow subtidal and clear “band” of no vegetation in the intertidal area
		BGC	Blue-green color	typical coloration of Zostera in digital imagery
		ADJ	Adjacent image	conclusive identification of seagrass habitat in the adjacent image
Drifting Macroalgae	The following image features were used to identify subtitdal photographic features as macroalgae.			
	I	IWU	Intertidal wash-up	seen at the water’s edge and wrack line - drift algae pushed up onto shore or very shallow areas
	II	WFP	Wave formed patterns	clearly visible as parallel wavy bands in near shore areas
		RVP	Rivulet patterns formed	where flowing water has shaped algal patterns
		UNI	Uniform texture	reflecting a dense bed that completely covers substrate
		BSE	Blurry smeared edges	patterns of drift algae grading into unvegetated habitat
		MBC	Color	muddy brown color
		MGC	Color	muddy green color
		DGC	Color	dark green color
Rockweeds	The following image features were used to identify dark tidal/subtidal photographic features as rockweed.			
	I	RKH	Rock halos	dark rings around light intertidal rocks
		IDB	Intertidal growth	growth in intertidal rocky areas with dark brownish color typical of Ascophyllum and Fucus
	II	MGP	Mottled growth patterns	reflecting variation of the substrate elevation and zonation patterns
		RCK	Bedrock	ridge patterns seen as parallel bands where algae grows on exposed bedrock|
		COB	Cobble	patterns seen as very high-contrast grainy texture where algae grows on cobble substrate
		ITG	Intertidal growth	predominantly intertidal growth pattern
		RBC	Color	reddish brown color
		DBC	Color	very dark brown color
		RDC	Color	reddish color
		ADJ	Adjacent Image	conclusive identification of rockweed habitat in an adjacent image

Table A2-1. Listing of the descriptions of factors used to identify three types of submerged aquatic vegetation.

Figure A2-1. Example of 2002 aerial image depicting vegetation targets.

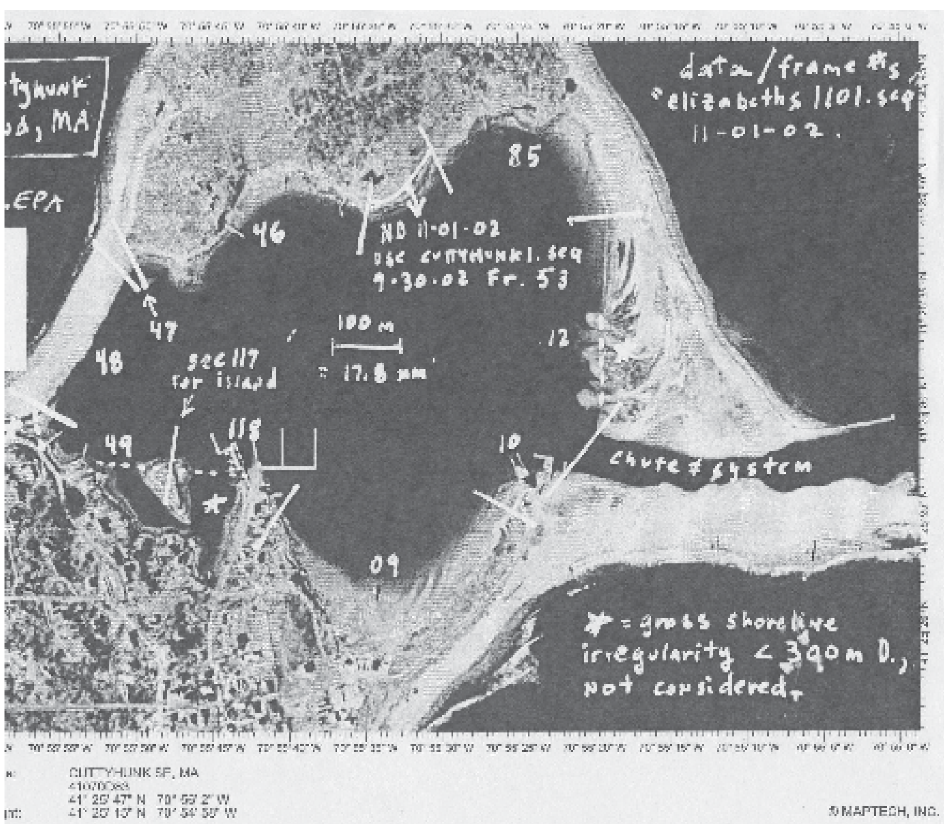

Figure A2-2: Aerial image of an embayment divided into segments for analysis.

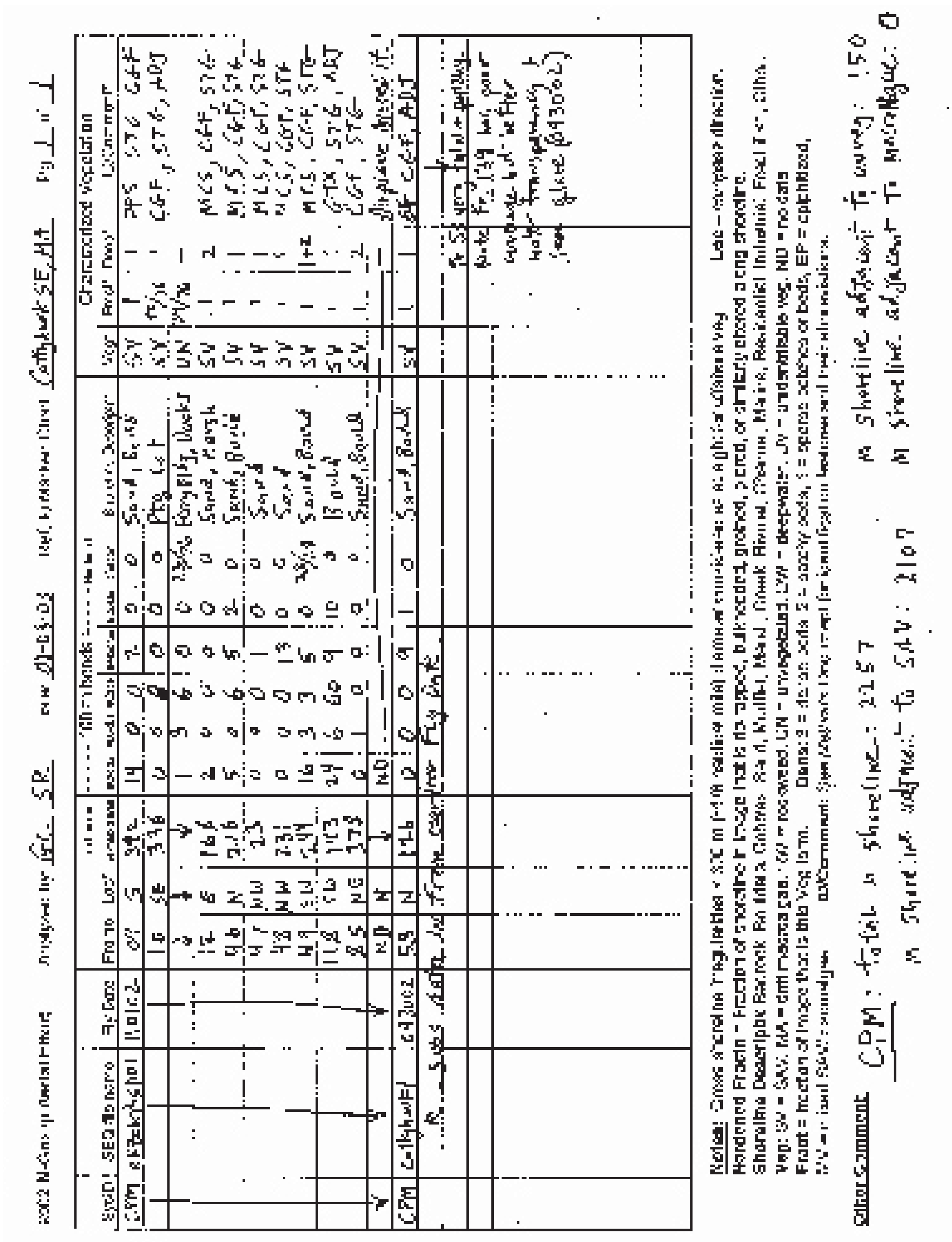

Figure A2-3: Example aerial imagery datasheet.

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Appendix 3. Nitrogen Load-Eelgrass Extent Model: Assumptions and Limitations

Four (4) sub-components make up the construction of the empirically derived multiple-system nitrogen load - eelgrass extent model detailed in this research summary:

(1) Nitrogen loading model
(2) Water flushing time model
(3) Eelgrass extent sampling and analysis
(4) Estuarine morphological characteristics

Each of these components is subject to a unique set of assumptions and limitations. The combination of all of the limitations will apply to the overall empirical model. The major assumptions and limitations of these components are listed below, as well as the overarching set of assumptions and limitations.

Loading Sub-model

Nitrogen loading rate values calculated for the study embayments are based on application of the published Nitrogen Loading Model (NLM) (Valiela *et al.* 1997). The reader should refer to this publication for details; however, the major assumptions of this model are noted below.

The NLM estimates nitrogen loading to watersheds and receiving waters (see Figure 3, in the main body of this report, for additional details). It considers diffuse, non-point source inputs and attempts to estimate losses in various compartments of the watershed. The model was developed for Waquoit Bay, MA, but "...with inputs for local conditions it is applicable to other rural to suburban watersheds underlain by unconsolidated sandy sediments." The authors include two major categories of nitrogen inputs to the watershed: i) nitrogen to watershed surfaces, which includes atmospheric deposition to 4 landuse types (natural vegetation, turf, agricultural land, and impervious surfaces) and fertilizer application to 2 landuse types (turf and agricultural land); and ii) septic wastewater nitrogen.

Assumptions:

1. Table A3-1 contains the input categories for the NLM and represents the total watershed derived and atmospherically derived nitrogen to the estuary. In addition, the algebraic expression used to calculate the nitrogen inputs to the watersheds are listed.

2. Table A3-2 contains the loss and transport coefficients applied to each type of land use category. The nitrogen that comes from the three sources (Table A3-1) and is deposited on the watershed is lost, or attenuated, according to the coefficients in the table.

3. Table A3-3 lists the data required in order to use the NLM to compute nitrogen inputs to the watersheds and surfaces of the study estuaries. It should be noted that because estimates of loading are largely based on landuse data the values represent long-term annual averages.

Limitations:

1. The model is applicable to estuaries dominated by nonpoint sources of nitrogen underlain by unconsolidated sandy soils. However, it is here applied to other estuaries in so far as the variance explained by the nitrogen load-eutrophication response model is statistically significant.

2. Does not include other direct nitrogen inputs (*e.g.*, from boats)

3. Does not include nitrogen loading from oceanic sources

4. Does not include nitrogen loading from regeneration in the estuary

5. The model cannot be used to assess temporal trends and variation in loading

Table A3- 1. Input categories for the Nitrogen Loading Model.

Input Category	Included Landuse Types	Nitrogen Load
Atmospheric Deposition:		
Natural Vegetation	forests, wetlands, natural lands	ATMOS. DEP.[1, 2] x AREA (kg N ha/ yr) (ha)
Turf	lawns, golf courses	ATMOS. DEP. x AREA (kg N ha/ yr) (ha)
Agricultural Land	crop land	ATMOS. DEP. x AREA (kg N ha/ yr) (ha)
Impervious Surfaces: roofs, driveways	roofs, driveways	ATMOS. DEP. x AREA (kg N ha/yr) (ha)
Impervious Surfaces: roads, roads, runways, parking lots	roads, runways, parking lots	ATMOS. DEP. x AREA (kg N ha/yr) (ha)
Fertilizer Application:		
Turf	lawns, golf courses	APPL. RATE.[3] x AREA x F_n (kg N ha/ yr) (ha) (fract)
Agricultural Land	crop land	APPL. RATE. x AREA - N rem[4] (kg N ha/yr) (ha)
Human Septic Wastewater[5]:	residential land	(kg N person/ yr) x (persons per house) x (# of houses)

[1] Uses the concentration of NO3-, NH4+, and DON in local precipitation and yearly rainfall totals to enerate the atmospheric deposition term. [2] Model also includes dry deposition (*i.e.*, from NO x. particles adhering to leaves and impervious surfaces), which is adjustable as a proportion of wet deposition. [3] Uses average fertilizer addition rates for Cape Cod of 105 kg N ha/yr for lawns and 115 kg N ha/ yr for golf courses. F_N = "0.34" refers to the fraction of homeowners applying fertilizer (applies to lawns only). [4] The term "N rem" allows for nitrogen removed from the watershed as crops (*i.e.*, consumed outside of the watershed); assumed zero in current estimates. [5] As originally published, the loading model applies to nonpoint sources; however, some of the estuaries contained municipal point source inputs. The magnitude of these inputs was estimated from effluent monitoring data. In these cases, if point source inputs were significantly greater than estimated nonpoint source inputs, then nonpoint sources were not included.

Table A3- 2. Loss and transport percentages used in the Nitrogen Loading Model.

Landuse Type	***In Situ* Loss**	**Transport**
Atmospheric Deposition:		
Natural Vegetation	65%, through retention in plants and soil	61% in vadose zone 35% in aquifer
Turf	62%, through retention in plants and soil	61% in vadose zone 35% in aquifer
Agricultural Land	62%, through retention in plants and soil	61% in vadose zone 35% in aquifer
Impervious Surfaces: roofs, driveways[1]	62%, through retention in plants and soil	61% in vadose zone 35% in aquifer
Impervious Surfaces: roads, runways, parking lots[2]	0 %	61% in vadose zone 35% in aquifer
Fertilizer Application:		
Turf	39% lost as gases	61% in vadose zone 35% in aquifer
Agricultural Land	39% lost as gases	61% in vadose zone 35% in aquifer
Human Septic Wastewater:	40% in septic tanks and leach fields 34% in plumes	35% in aquifer

[1]Assumes that precipitation falling on roofs and driveways subsequently runs off to lawns and natural lands where losses may occur. [2]Assumes that precipitation falling on roads, runways, parking lots is collected in catchment basins and delivered directly to the vadose zone.

Table A3- 3. Other data used to compute watershed loading by the Nitrogen Loading Model.

Data Required	Source/Reference
Concentration of NO3, NH4+, DON in regional precipitation	(Luo *et al.* 2002)
Yearly rainfall totals	NCDC (http://www.ncdc.noaa.gov)
Fertilizer application rates for lawns and golf courses	(Valiela *et al.* 1997)
Average lawn size	(Valiela *et al.* 1997)
Fertilizer application rates for farms	(Valiela *et al.* 1997)
Average roof and driveway areas	(Valiela *et al.* 1997)
Annual N release per person	(Valiela *et al.* 1997)
Number of houses	residential landuse (see landuse data below)
Occupancy rate for houses	residential landuse (see landuse data below)
Direct municipal wastewater treatment facility inputs	CT: Paul Stacey (CT DEP)* RI: Scott Duerr (Westerly) Scott Nixon (URI) * MA: Brien Friedman, Russell Isaac (MA DEP) *
Landuse areas for: natural vegetation, golf courses, crop land, commercial and industrial development, and residential land	State of Connecticut[1] State of Rhode Island[2] State of Massachusetts[3]

[1]Developed from 30-meter Landsat thematic mapper (TM) data acquired by the Multi-resolution Land Characterization (MRLC) Consortium. The base data set was leaves-off Landsat TM data, nominal-1992 acquisitions. Collection date: 1992. Collection platform: Landsat satellite.[2]Originally interpreted from 1988 aerial photography and updated from 1992 - 1995 orthos. Intended scale 1:24,000. Collection date: Originally in 1988, updated with 1992 - 1995 data. Collection platform: Aerial photography. [3]Interpreted from 1:25,000 aerial photography. Collection date: 1999. Collection platform: Aerial photography.*personal communication.

Flushing Time Sub-model

The flushing time (the time for fresh or estuarine water to travel through an estuary) for each of the study embayments is based on an empirical relationship between published flushing times and estuarine area (see Appendix 1). An average of values generated by a power law and linear equation were used to calculate flushing times for this study.

Assumptions (see Appendix 1)

1. Flushing time for study estuaries can be based on empirical data from 10 estuaries of similar size and in a similar biogeographic region.
2. Fresh water and estuarine flushing times are not distinguished and are grouped as simply "water" flushing time. This is because of the similarity of results from literature values.
3. Morphological factors other than length and width (*e.g.*, depth, circulation restrictions, tides, and shape) are not primary factors in predicting flushing times within similar estuarine classes.

Limitations (see Appendix 1)

The model is applicable to embayment type estuaries with the characteristics (Madden *et al.* 2005) contained in Table A3-4.

Eelgrass Extent Sampling and Analysis

The eelgrass extent (length along the shoreline) for each of the study estuaries was derived from airplane digital photography, image processing, image analysis, and vegetation identification. The details of each of these components can be found in Appendix 2.

Assumptions (see Appendix 2):

1. That airplane derived images, processed and analyzed for submerged vegetation, represent an indicator of extent of submerged aquatic vegetation during the time of flights.

Table A3- 4. List of CMECS descriptors to define the class of estuarine embayments for the study systems

Temperature Class:	Cold (0-10°C) to temperate (10-20°C)
Salinity Class:	Mesohaline (5-18 psu) to euhaline (30-40 psu)
Oxygen Class:	Variable (anoxic, hypoxic, oxic, saturated, supersaturated)
Turbidity Class:	Moderately turbid (2-4 m)
Turbidity Type:	Mixed (chlorophyll, mineral, colloidal, dissolved color, detrital)
Turbidity Provenance:	Mixed (allochthonous, autochthonous, resuspended, terrigenous, marine)
Energy Type:	Wind/tide/current
Energy Intensity:	Moderate (moderate currents and wave action, 2-4 kn)
Energy Direction:	Mixed
Depth Class:	Very shallow (0-5 m)
Tide Class:	Small (0.1-1 m) to moderate (1-5 m) tidal range
Primary Water Source:	Watershed, local estuary, local marine (non-river dominated)
Enclosure Status:	Partially-enclosed (50-75% area encircled by land)
Trophic Status:	Oligotrophic (<5 ug Chl-a/L) to eutrophic (>50 ug Chl-a/L)
Region:	Eight (8); Virginian Atlantic Region. The region extends along the eastern North American continent from Cape Hattaras northward to Cape Cod. The region lies within the temperate climatological zone, and is interposed between the east coast and the Northern Gulf Stream Transition Region offshore (Region 9).
Embayment Size:	Small (0.1 km^2) to medium (6 km^2)
Watershed Size:	Small (0.5 km^2) to medium (73 km^2)
Ecoregions:	Northeastern Coastal Zone and Atlantic Coast Pine Barrens (Shirazi *et al.* 2003)
Geographic:	Southern New England Region (CT, RI and southeastern MA coastal)

2. That the segment indicator is a reasonably accurate index of eelgrass extent for study estuaries and compares favorably with areal estimates (Pesch *et al.* Submitted).

3. That bottom substrate was likely only of major importance for those systems in which no eelgrass was detected.

Limitations (see Appendix 2):

1. Derived eelgrass extent indices represent the SAV extent for 2002 only; additional data are needed to assess temporal variability.

2. No point-specific groundtruthing during this period took place; however, comparisons with other studies for some of the study systems were favorable.

3. Track-lines were not optimal for efficient analysis.

Estuarine Morphological Characteristics

Assumptions:

That data derived from GIS coverages give a reasonably accurate depiction of major system morphological characteristics (*e.g.*, depth, volume, area, etc.)

Limitations:

Subject to limitations of data availability, accuracy, precision of the underlying data.

Overall Assumptions and Limitations of the Nitrogen load – Eelgrass Extent Model

Assumptions:

1. The model is subject to the assumptions of the constituent components of the empirical model (*i.e.*, load model, residence time model, eelgrass sampling, analysis, and morphological characteristics). See above.

2. Nitrogen loading from sources outside of the loading model are not primary drivers (*e.g.*, oceanic inputs, direct carbon inputs, direct nitrogen inputs).

3. Annual average nitrogen inputs (long-term) provided by the loading model can be successfully related to annual seagrass extent assessments.

Limitations:

The model is subject to the limitations of the constituent components that make up the empirical model (*i.e.*, load model, residence time model, eelgrass sampling, analysis, and morphological characteristics). See above. Therefore the nitrogen load-eelgrass response model can be applied to the estuarine embayments with the characteristics contained in Table A3-5.

Table A3- 5. Characteristics of estuarine embayments for which the nitrogen load-eelgrass extent model can apply.

Temperature Class:	Cold (0-10°C) to temperate (10-20°C)
Salinity Class:	Mesohaline (5-18 psu) to euhaline (30-40 psu)
Oxygen Class:	Variable (anoxic, hypoxic, oxic, saturated, supersaturated)
Turbidity Class:	Moderately turbid (2-4 m)
Turbidity Type:	Mixed (chlorophyll, mineral, colloidal, dissolved color, detrital)
Turbidity Provenance:	Mixed (allochthonous, autochthonous, resuspended, terrigenous, marine)
Energy Type:	Wind/tide/current
Energy Intensity:	Moderate (moderate currents and wave action, 2-4 kn)
Energy Direction:	Mixed
Depth Class:	Very shallow (0-5 m)
Tide Class:	Small (0.1-1 m) to moderate (1-5 m) tidal range
Primary Water Source:	Watershed, local estuary, local marine (non-river dominated)
Enclosure Status:	Partially-enclosed (50-75% area encircled by land)
Trophic Status:	Oligotrophic (<5 ug Chl-a/L) to eutrophic (>50 ug Chl-a/L)
Region:	Eight (8); Virginian Atlantic Region. The region extends along the eastern North American continent from Cape Hattaras northward to Cape Cod. The region lies within the temperate climatological zone, and is interposed between the east coast and the Northern Gulf Stream Transition Region offshore (Region 9).
Embayment Size:	Small (0.1 km^2) to medium (6 km^2)
Watershed Size:	Small (0.5 km^2) to medium (73 km^2)
Ecoregions:	Northeastern Coastal Zone and Atlantic Coast Pine Barrens (Shirazi *et al.* 2003)
Geographic:	Southern New England Region (CT, RI and southeastern MA coastal)
Habitat Suitability:	Estuaries that have suitable habitat characteristics that support eelgrass (in this report, historical presence is used as a proxy)

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