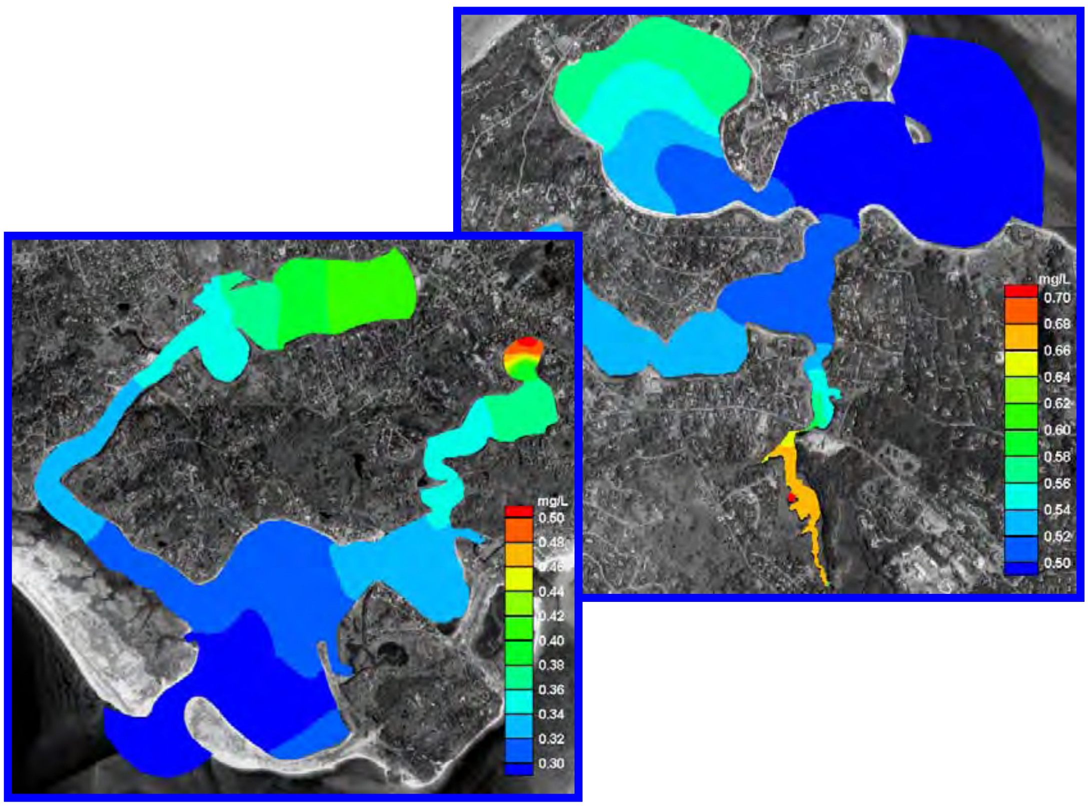
Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Re-evaluate Critical Nitrogen Loading Thresholds for Stage Harbor/Oyster Pond, Sulphur Springs/Bucks Creek and Taylors Pond/Mill Creek Chatham, Massachusetts



FINAL REPORT – FEBRUARY 2007

Massachusetts Department of Environmental Protection

University of Massachusetts Dartmouth School of Marine Science and Technology

Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Re-evaluate Critical Nitrogen Loading Thresholds for Stage Harbor/Oyster Pond, Sulphur Springs/Bucks Creek and Taylors Pond/Mill Creek Chatham, Massachusetts

FINAL REPORT – FEBRUARY 2007

Brian Howes
Roland Samimy
David Schlezinger

Sean Kelley
John Ramsey

Ed Eichner

Contributors:
US Geological Survey
Don Walters and John Masterson

Applied Coastal Research and Engineering, Inc.
Elizabeth Hunt

SMAST Coastal Systems Program
Jennifer Benson, Michael Bartlett, Sara Sampieri

Cape Cod Commission
Kiaotong Wu

Preface and Acknowledgements

This current re-evaluation of the nitrogen thresholds for the three south side Chatham embayments represents the fifth product in a series of deliverables developed under the overall Massachusetts Estuaries Project nutrient evaluation for the Town of Chatham that got underway in 2002 (building upon an earlier effort by ACRE/ASA/SMAST 2000). The present effort was undertaken by the MEP Technical Team to assist the Town of Chatham in its CWMP for the restoration of its estuaries. The MEP Technical Team received no support from MassDEP for updating the water-use/land-use database and the re-calibration of the MEP Linked Models presented herein. However, as the updating of these models is critical to making management decisions relative to these estuaries, the MEP Technical Team and the Town of Chatham performed this update. Critical MEP documents that preceded this specific evaluation include:

1) MEP Linked Watershed-Embayment Model to Determine Critical Nitrogen Thresholds for Stage Harbor, Sulphur Springs, Taylors Pond, Bassing Harbor and Muddy Creek, Chatham, Massachusetts (FINAL - December 2003)
2) MEP linked Watershed-Embayment Model to Determine Critical NitrogenThresholds for the Pleasant Bay System, Orleans, Chatham, Brewster, and Harwich, Massachusetts (FINAL – May 2006)
3) MEP Technical Memorandum: Muddy Creek Scenario Analyses (FINAL – August 2006
4) MEP Technical Memorandum: Cockle Cove Salt Marsh Nitrogen Threshold (DRAFT – May 2006, FINAL – November 2006)

The Massachusetts Estuaries Project Technical Team would like to acknowledge the contributions of the many individuals who have worked tirelessly for the restoration and protection of the critical coastal resources within the Town of Chatham. Without these stewards and their efforts, this project would not have been possible.

First and foremost is the significant time and effort in data collection and discussion spent by members of the Chatham Technical and Citizens Advisory Committees, and particularly the contributions of Dr. Bob Duncanson, Bill Redfield, Margaret Swanson, Ted Keon, Fred Jensen, and John Payson. Similarly, critical data to support this assessment and synthesis was provided by the citizens who have generously given of their time to collect the needed nutrient samples over the past 7 summers through the Chatham WaterWatchers and the Town of Chatham Water Quality Monitoring Program. In addition, the Chatham Water Quality Laboratory has collaborated in the sample collection for a variety of sub-projects, needed to address site-specific nitrogen issues..

In addition to local contributions, we are also thankful for the long hours in the field and laboratory spent by SMAST staff (Dr. David White, Elizabeth White), Nat Donkin, interns and students within the Coastal Systems Program at SMAST-UMD.

SUGGESTED CITATION

Howes B.L., E.M. Eichner, S. W. Kelley, J. S. Ramsey, R.I. Samimy (2007). Water-Use Update to the Linked Watershed-Embayment Model to Evaluate Critical Nitrogen Loading Thresholds for Stage Harbor/Oyster Pond, Sulphur Springs/Bucks Creek, Taylors Pond/Mill Creek, Chatham, Massachusetts. Massachusetts Estuaries Project Technical Team Update Report.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

III. DELINEATION OF WATERSHEDS

III.1 BACKGROUND

The Massachusetts Estuaries Project Technical Team includes staff from the United States Geological Survey (USGS). The USGS groundwater modelers were central to the development of the groundwater modeling approach used by the Estuaries Project. The USGS has a long history of developing regional models for the six-groundwater flow cells on Cape Cod. Through the years, advances in computing, lithologic information from well installations, water level monitoring, stream flow measurements, and reconstruction of glacial history have allowed the USGS to update and refine the groundwater models. The MODFLOW and MODPATH models utilized by the USGS to organize and analyze the available data use up-to-date mathematical codes and create better tools to answer the wide variety of questions related to watershed delineation, surface water/groundwater interaction, groundwater travel time, and drinking water well impacts that have arisen during the MEP analysis of southeastern Massachusetts estuaries, including the three southern Chatham estuaries: Taylors Pond/Mill Creek, Sulfur Springs/Bucks Creek, and Stage Harbor/Oyster Pond. The southern Chatham estuaries and their watersheds are located within the Towns of Chatham and Harwich. The estuaries are situated along the southeastern edge of Cape Cod and are bounded by the Nantucket Sound.

In the present (MEP) investigation, the USGS was responsible for the application of its groundwater modeling approach to define the watershed or contributing area to the three southern Chatham estuarine systems under evaluation by the Project Team. Further modeling of the estuary watersheds was undertaken to sub-divide the overall watersheds into functional sub-units based upon: (a) defining inputs from contributing areas to each major portion within the embayment system, (b) defining contributing areas to major freshwater aquatic systems which generally attenuate nitrogen passing through them on the way to the estuary (lakes, streams, wetlands), and (c) defining 10 year time-of-travel distributions within each sub-watershed as a procedural check to gauge the potential mass of nitrogen from “new” development, which has not yet reached the receiving estuarine waters. The three-dimensional numerical model employed is also being used to evaluate the contributing areas to public water supply wells in the overall Monomoy groundwater flow cell. Model assumptions for calibration were matched to surface water inputs and flows from the most current (2002 to 2003) stream gage information.

The relatively transmissive sand and gravel deposits that comprise most of Cape Cod create a hydrologic environment where watershed boundaries are usually better defined by elevation of the groundwater and its direction of flow, rather than by land surface topography (Cambareri and Eichner 1998, Millham and Howes 1994a,b). Freshwater discharge to estuaries is usually composed of surface water inflow from streams, which receive much of their water from groundwater base flow, and direct groundwater discharge. For a given estuary, differentiating between these two water input pathways and tracking the sources of nitrogen that they carry requires determination of the portion of the watershed that contributes directly to the stream and the portion of the groundwater system that discharges directly into the estuary as groundwater seepage.

III.2 MODEL DESCRIPTION

Contributing areas to the three southern Chatham estuaries and their associated upland freshwater bodies were delineated using a regional model of the Monomoy Lens flow cell (Walter and Whealan, 2005). The USGS three-dimensional, finite-difference groundwater model MODFLOW-2000 (Harbaugh, *et al.*, 2000) was used to simulate groundwater flow in the aquifer. The USGS particle-tracking program MODPATH4 (Pollock, 2000), which uses output files from MODFLOW-2000 to track the simulated movement of water in the aquifer, was used to delineate the area at the water table that contributes water to wells, streams, ponds, and coastal water bodies. This approach was used to determine the contributing areas to the Pleasant Bay system and also to determine portions of recharged water that may flow through ponds and streams prior to discharging into coastal water bodies.

The Monomoy Flow Model grid consists of 164 rows, 220 columns and 20 layers. The horizontal model discretization, or grid spacing, is 400 by 400 feet. The top 17 layers of the model extend to a depth of 100 feet below NGVD 29 and have a uniform thickness of 10 ft. The top of layer 8 resides at NGVD 29 with layers 1-7 stacked above and layers 8-20 below. Layer 18 has a thickness of 40 feet and extends to 140 feet below NGVD 29, while layer 19 extends to 240 feet below NGVD 29. The bottom layer, layer 20, extends to the bedrock surface and has a variable thickness depending upon site characteristics (up to 525 feet below NGVD 29). The rewetting capabilities of MODFLOW-2000, which allows drying and rewetting of model cells, was used to simulate the top of the water table, which varies in elevation depending on the location in the Lens. Since water elevations are less than +40 ft in the portion of the Monomoy Lens in which the three southern Chatham estuaries reside, the three uppermost layers of the model are inactive.

The glacial sediments that comprise the aquifer of the Monomoy Lens consist of gravel, sand, silt, and clay that were deposited in a variety of depositional environments. The sediments generally show a fining downward sequence with sand and gravel deposits deposited in glaciofluvial (river) and near-shore glaciolacustrine (lake) environments underlain by fine sand, silt and clay deposited in deeper, lower-energy glaciolacustrine environments. Most groundwater flow in the aquifer occurs in shallower portions of the aquifer dominated by coarser-grained sand and gravel deposits. The three southern Chatham estuaries watersheds are generally located in the Harwich Outwash Plain, which was deposited as glacial ice lobes were retreating to positions near the current Cape Cod Bay shoreline and the barrier beach along the eastern edge of Pleasant Bay (Walter and Whealan, 2005). The Chatham area is also somewhat more geologically complicated because of older kame deposits in the middle of town that extend upward through the outwash plain materials (Oldale and Koteff, 1970). Lithologic data used to determine hydraulic conductivities used in the model were obtained from a variety of sources including well logs from USGS, local Town records and data from previous investigations. Final aquifer parameters were determined through calibration to observed water levels and stream flows. Hydrologic data used for model calibration included historic water-level data obtained from USGS records and local Town water-level data.

The model simulates steady state, or long-term average, hydrologic conditions including a long-term average recharge rate of 27.25 inches/year and the pumping of public-supply wells at average annual withdrawal rates for the period 1995-2000 with a 15% consumptive loss. This recharge rate is based on the most recent USGS information. Large withdrawals of groundwater from pumping wells may have a significant influence on water tables and watershed boundaries and therefore the flow and distribution of nitrogen within the aquifer. After accounting for the consumptive loss and measured discharge at municipal wastewater treatment facilities, water withdrawn from the modeled aquifer by public drinking water supply wells is evenly returned within designated residential areas utilizing on-site septic systems. The model incorporates return flow from the Chatham municipal wastewater treatment facility based on effluent discharge the facility site.

III.3 TAYLORS POND/MILL CREEK, SULFUR SPRINGS/BUCKS CREEK, AND STAGE HARBOR/OYSTER POND CONTRIBUTORY AREAS

Newly revised watershed and sub-watershed boundaries were determined by the United States Geological Survey (USGS) for the three southern Chatham estuary systems (Figure III-1). Model outputs of MEP watershed boundaries were "smoothed" to (a) correct for the grid spacing, (b) to enhance the accuracy of the characterization of the pond and coastal shorelines, and (c) to more closely match the sub-embayment segmentation of the tidal hydrodynamic model. The smoothing refinement was a collaborative effort between the USGS and the rest of the MEP Technical Team. The MEP sub-watershed delineations also include 10 yr time of travel boundaries. Overall, 30 sub-watershed areas, including six freshwater ponds, were delineated within the watersheds to the three southern Chatham estuary systems.

Table III-1 provides the daily freshwater discharge volumes for each of the sub-watersheds as calculated by the groundwater model. These volumes were used to assist in the salinity calibration of the tidal hydrodynamic models and to determine hydrologic turnover in the lakes/ponds, as well as for comparison to measured surface water discharges. The overall estimated freshwater inflow into the Taylors Pond/Mill Creek, Sulfur Springs/Bucks Creek, and Stage Harbor/Oyster Pond estuaries from their respective MEP watersheds are 6,343 m3/d, 10,512 m3/d, and 25,031 m3/d.

The delineations completed by this revised MEP analysis are the third watershed delineation completed in recent years for these estuaries. Figure III-2 compares the delineation completed under the initial MEP analysis with the delineation utilized in previous Chatham assessments (e.g., Stearns and Wheler, 1999). Figure III-3 compares the delineation completed under the current MEP analysis with the delineation completed in the previous MEP analysis (Howes, et al., 2003). The pre-MEP delineation was defined based on regional water table measurements collected from available wells over a number of years and normalized to average conditions; delineations based on this previous effort were incorporated into the Commission's regulations through the Regional Policy Plan (CCC, 1996 & 2001).

Overall, the MEP contributing areas to the three southern Chatham estuaries based upon the groundwater modeling effort are very similar in area to the previous delineation based upon available well data. However, a small number of the interior sub-watersheds areas are different; for example the sub-watershed area to Little Mill Pond is reduced by 36%. The only difference between the current and previous MEP delineations is the inclusion of Hawknest Pond, which was added during the MEP analysis of Pleasant Bay (Howes, et al., 2005), but has a negligible effect on the watershed area.

The evolution of the watershed delineations for the three southern Chatham estuaries has allowed increasing accuracy as each new version adds new hydrologic data to that previously collected; the model allows all this data to be organized and to be brought into congruence with data from adjacent watersheds. The evaluation of older data and incorporation of new data during the development of the model is important as it decreases the level of uncertainty in the final calibrated and validated linked watershed-embayment model used for the evaluation of nitrogen management alternatives. Errors in watershed delineations do not necessarily result in proportional errors in nitrogen loading as errors in loading depend upon the land-uses that are included/excluded within the contributing areas. Small errors in watershed area can result in large errors in loading if a large source is counted in or out. Conversely, large errors in watershed area that involve only natural woodlands have little effect on nitrogen inputs to the down gradient estuary.

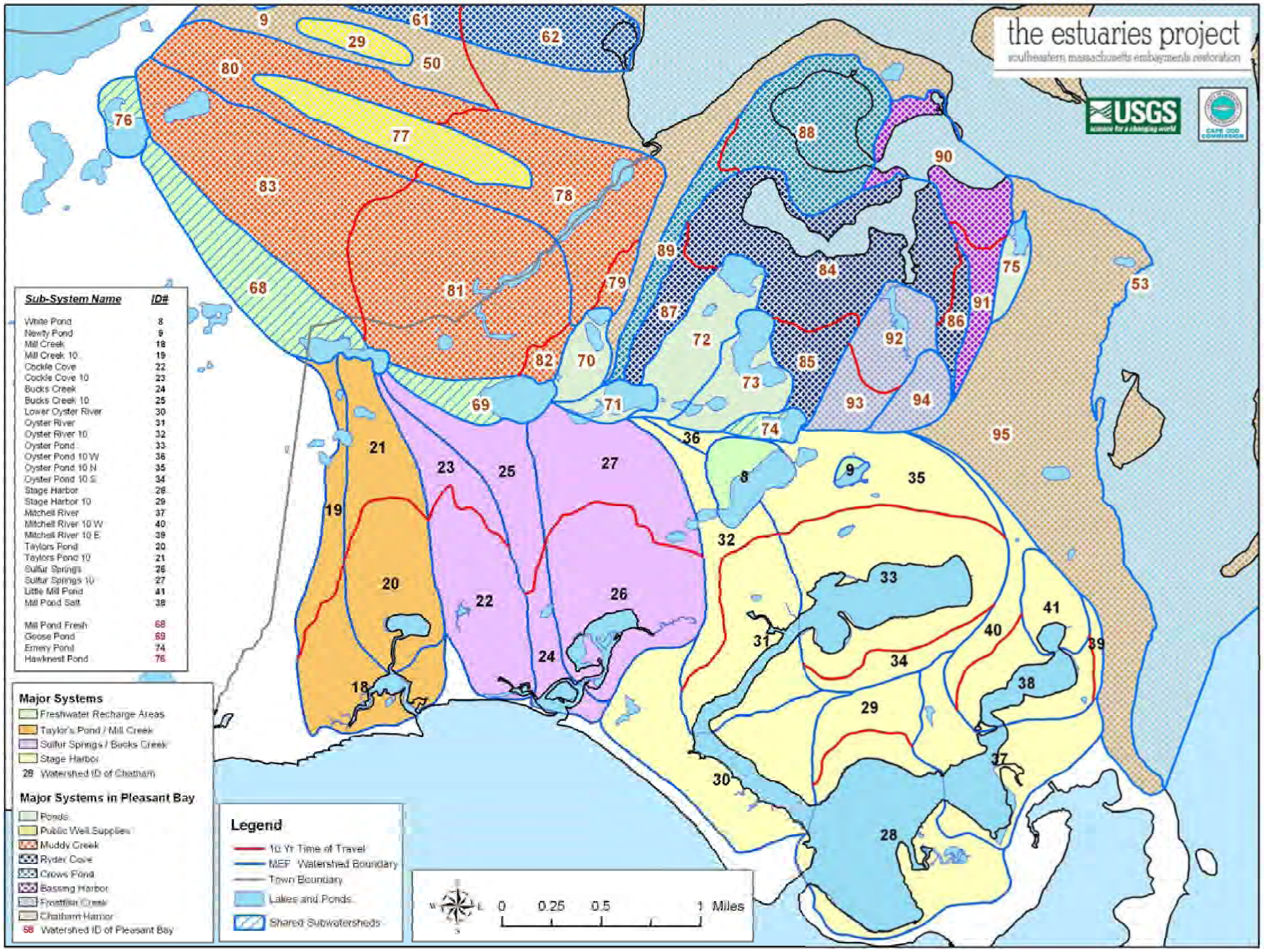


Figure III-1. Watershed delineation for Taylors Pond/Mill Creek, Sulfur Springs/Bucks Creek, and Stage Harbor/Oyster Pond estuaries. Approximate ten year time-of-travel delineations were produced for quality assurance purposes and are designated with a "10" in the watershed names (above). Sub-watersheds to embayments were selected based upon the functional estuarine sub-units in the water quality model (see section VI). Assigned watershed numbers are same as Howes et al (2003); watersheds shared with Pleasant Bay have watershed numbers assigned in Howes et al (2006).

Table III-1. Daily groundwater discharge to each of the sub-watersheds in the watersheds to Taylors Pond/Mill Creek, Sulfur Springs/Bucks Creek, and Stage Harbor/Oyster Pond estuaries, as determined from the USGS groundwater model.

Watershed	Watershed #	Discharge	
		m^3/day	ft^3/day
Hawknest Pond	75	375	13,243
Mill Pond Fresh	68	1,807	63,828
Goose Pond	69	703	24,820
Emery Pond	74	270	9,533
White Pond	8	731	25,815
Newty/Perch Pond	9	86	3,052
Mill Creek	18	1,595	56,331
Mill Creek 10	19	1,081	38,173
Taylors Pond	20	1,636	57,772
Taylors Pond 10	21	1,136	40,116
Cockle Cove	22	1,791	63,249
Cockle Cove 10	23	717	25,307
Bucks Creek	24	725	25,616
Bucks Creek 10	25	1,261	44,535
Sulfur Springs	26	2,700	95,345
Sulfur Springs 10	27	2,221	78,430
Stage Harbor	28	3,907	137,986
Stage Harbor 10	29	704	24,873
Lower Oyster River	30	4,736	167,237
Oyster River	31	6,670	235,549
Oyster River 10	32	1,692	59,760
Oyster Pond	33	3,460	122,206
Oyster Pond 10 S	34	465	16,439
Oyster Pond 10 N	35	1,994	70,425
Oyster Pond 10 W	36	127	4,474
Mitchell River	37	2,044	72,179
Mill Pond Salt	38	1,689	59,645
Mill Pond Salt 10 E	39	103	3,625
Mill Pond Salt 10 W	40	652	23,034
Little Mill Pond	41	688	24,308

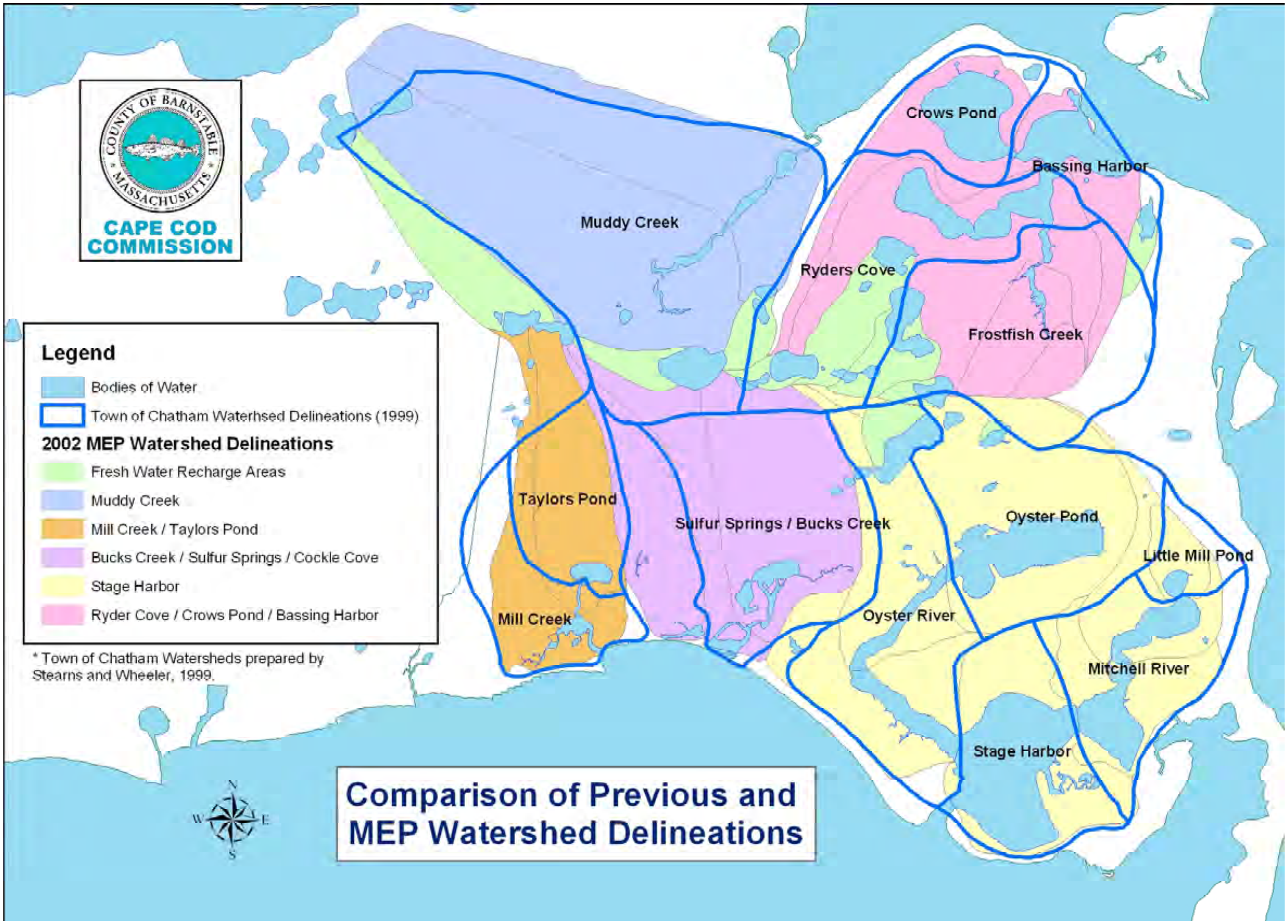


Figure III-2. Comparison of watershed and subwatershed delineations used prior to the MEP and the delineations used for the 2003 MEP analysis.

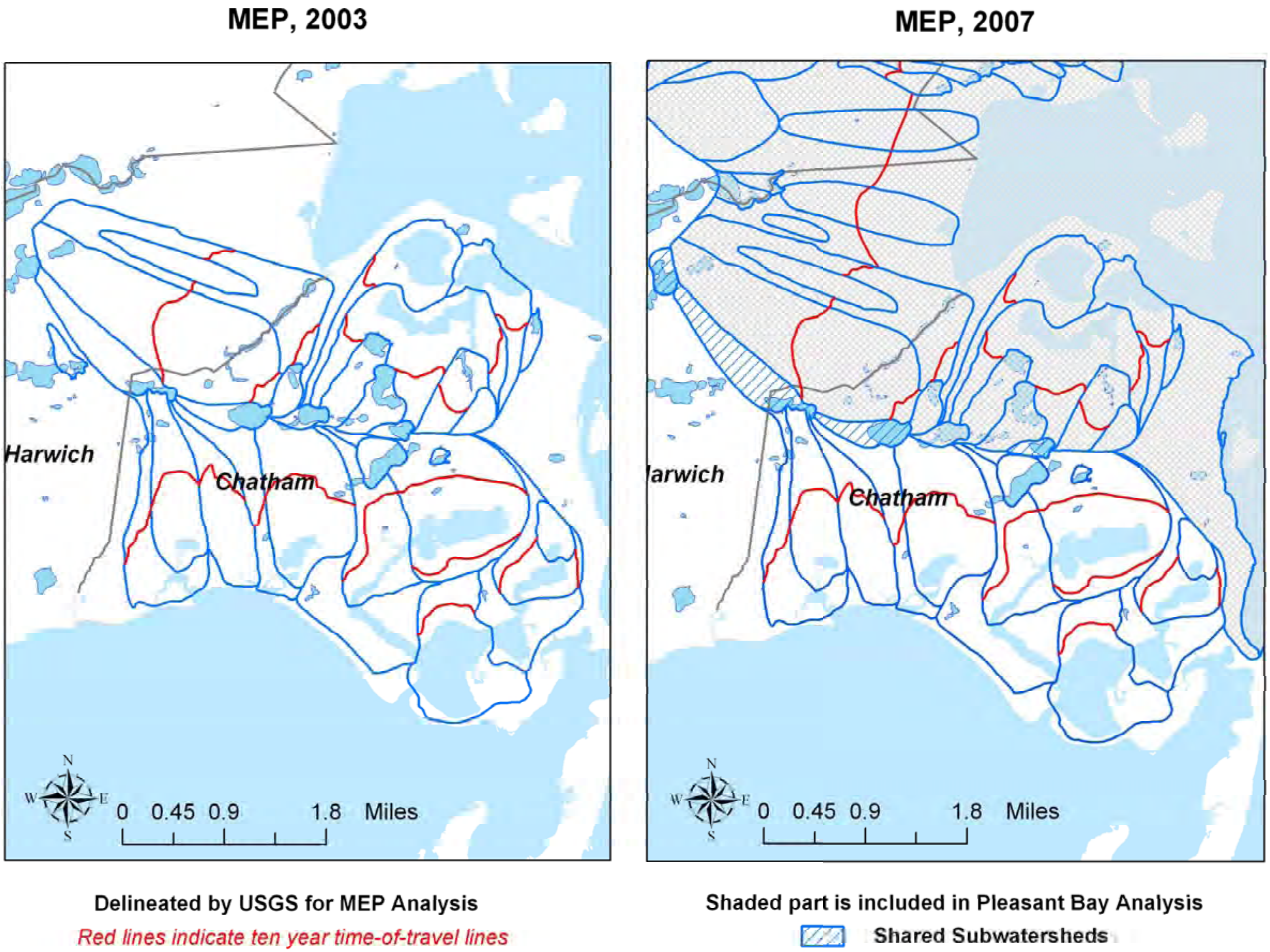


Figure III-3. Comparison of watershed and sub-watershed delineations used in the 2003 MEP analysis and the current analysis. Watersheds to the southern Chatham estuaries are the same except for the addition of the watershed to Hawknest Pond in Harwich (WS#76 in Figure III-1), which was delineated during the MEP analysis of Pleasant Bay (Howes, et al., 2005).

IV. WATERSHED NITROGEN LOADING TO EMBAYMENT: LAND USE, STREAM INPUTS, AND SEDIMENT NITROGEN RECYCLING

IV.1 WATERSHED LAND USE BASED NITROGEN LOADING ANALYSIS

Management of nutrient related water quality and habitat health in coastal waters requires determination of the amount of nitrogen transported by freshwaters (surface water flow, groundwater flow) from the surrounding watershed to the receiving embayment of interest. In southeastern Massachusetts, the nutrient of management concern for estuarine systems is nitrogen and this is true for the three southern Chatham systems (Taylors Pond/Mill Creek, Sulfur Springs/Bucks Creek, and Stage Harbor/Oyster Pond). Determination of watershed nitrogen inputs to these embayment systems requires the (a) identification and quantification of the nutrient sources and their loading rates to the land or aquifer, (b) confirmation that a groundwater transported load has reached the embayment at the time of analysis, and (c) quantification of nitrogen attenuation that can occur during travel through lakes, ponds, streams and marshes. This latter natural attenuation process results from biological processes that naturally occur within ecosystems. Failure to account for attenuation of nitrogen during transport results in an over-estimate of nitrogen inputs to an estuary and an underestimate of the sensitivity of a system to new inputs (or removals). In addition to the nitrogen transport from land to sea, the amount of direct atmospheric deposition on each embayment surface must be determined as well as the amount of nitrogen recycling within the embayment, specifically nitrogen regeneration from sediments. Sediment nitrogen recycling results primarily from the settling and decay of phytoplankton and macroalgae (and eelgrass when present). During decay, organic nitrogen is transformed to inorganic forms, which may be released to the overlying waters or lost to denitrification within the sediments. Burial of nitrogen is generally small relative to the amount cycled. Sediment nitrogen regeneration can be a seasonally important source of nitrogen to embayment waters and leads to errors in predicting water quality if it is not included in determination of summertime nitrogen load.

The MEP Technical Team includes technical staff from the Cape Cod Commission (CCC). In coordination with other MEP technical team staff, CCC staff developed nitrogen-loading rates (Section IV.1) to the three southern Chatham embayment systems whose watersheds delineations are discussed in Section III. The southern Chatham watersheds were sub-divided to define contributing areas to each of the major inland freshwater ponds and further sub-divided into regions greater and less than 10 year groundwater travel time from the receiving estuary, a total of 30 sub-watersheds in all. The nitrogen loading effort also involved further refinement of watershed delineations to accurately reflect shoreline areas to freshwater ponds and each embayment (see Section III).

The initial task in the MEP land use analysis is to gauge whether or not nitrogen discharges to the watershed have reached the embayment. This involves a temporal review of land use changes and the time of groundwater travel provided by the USGS watershed model. After reviewing the percentage of nitrogen loading in the less than 10 year time of travel (LT10) and greater than 10 year time of travel (GT10) watersheds (Table IV-1), land use development records, and water quality modeling, it was determined that the southern Chatham estuaries are currently in balance with their watershed loads. The bulk of the watershed nitrogen load is within 10 years flow to each of the estuaries: Taylors Pond/Mill Creek (61% within 10 years travel time), Sulfur Springs/Bucks Creek (66%), and Stage Harbor/Oyster Pond (70%). Therefore, the distinction of less than 10 year and greater than 10 year time of travel regions within a sub-watershed (Figure III-1) was eliminated and the number of sub-watersheds was reduced to 19 (Figure IV-1). The overall result of the timing of development relative to groundwater travel times is that the present watershed nitrogen load appears to accurately reflect the present nitrogen sources to the estuaries (after accounting for natural attenuation, see below).

Table IV-1. Percentage of unattenuated nitrogen loads in less than 10 time of travel subwatersheds to Southern Chatham Estuaries

WATERSHED	LT10	GT10	TOTAL	%LT10
Name	kg/yr	kg/yr	kg/yr	
Taylors Pond/Mill Creek	2,622	1,678	4,300	61%
Sulfur Springs/Bucks Creek	5,258	2,714	7,972	66%
Stage Harbor	9,514	4,075	13,590	70%

In order to determine nitrogen loads from the watersheds, detailed individual lot-by-lot data is used for some portion of the loads, while information developed from other detailed studies is applied to other portions. The Linked Watershed-Embayment Management Model (Howes & Ramsey 2001) uses a land-use Nitrogen Loading Sub-Model based upon sub-watershed specific land uses and pre-determined nitrogen loading rates. For the southern Chatham embayment systems, the model used Chatham specific land use data transformed to nitrogen loads using both regional nitrogen load factors and local watershed specific data (such as parcel by parcel water use). Determination of the nitrogen loads required obtaining watershed specific information regarding wastewater, fertilizers, runoff from impervious surfaces and atmospheric deposition. The primary regional factors were derived for southeastern Massachusetts from direct measurements. The resulting nitrogen loads represent the “potential” or unattenuated nitrogen load to each receiving embayment, since attenuation during transport has not yet been included.

Natural attenuation of nitrogen during transport from land-to-sea (Section IV.2) within the three southern Chatham watersheds was determined based upon a site-specific study within the freshwater portions of Cockle Cove Creek and through the six freshwater ponds within the watershed. Attenuation during transport through each of the major fresh ponds was determined through (a) comparison with other Cape Cod lake studies and (b) data collected on each pond. Attenuation during transport through these fresh ponds was conservatively assumed to equal 50% based on available monitoring of selected Cape Cod lakes. Available historic data collected from individual fresh ponds in the southern Chatham estuary watersheds confirmed the appropriateness of this conservative estimate.

Natural attenuation during stream transport or in passage through fresh ponds of sufficient size to effect groundwater flow patterns (area and depth) is a standard part of the data collection effort of the MEP. However, if additional attenuation of nitrogen were occurring during transport, given the distribution of the nitrogen sources, nitrogen loading to the estuary would only be slightly (<10%) overestimated. Based upon these considerations, the MEP Technical Team used the conservative estimate of nitrogen loading based upon direct groundwater discharge. Internal nitrogen recycling was also determined throughout the tidal reaches of the southern Chatham estuaries; measurements were made to capture the spatial distribution of sediment nitrogen regeneration from the sediments to the overlying water-column. Nitrogen regeneration focused on summer months, the critical nitrogen management interval and the focal season of the MEP approach and application of the Linked Watershed-Embayment Management Model (Section IV.3).

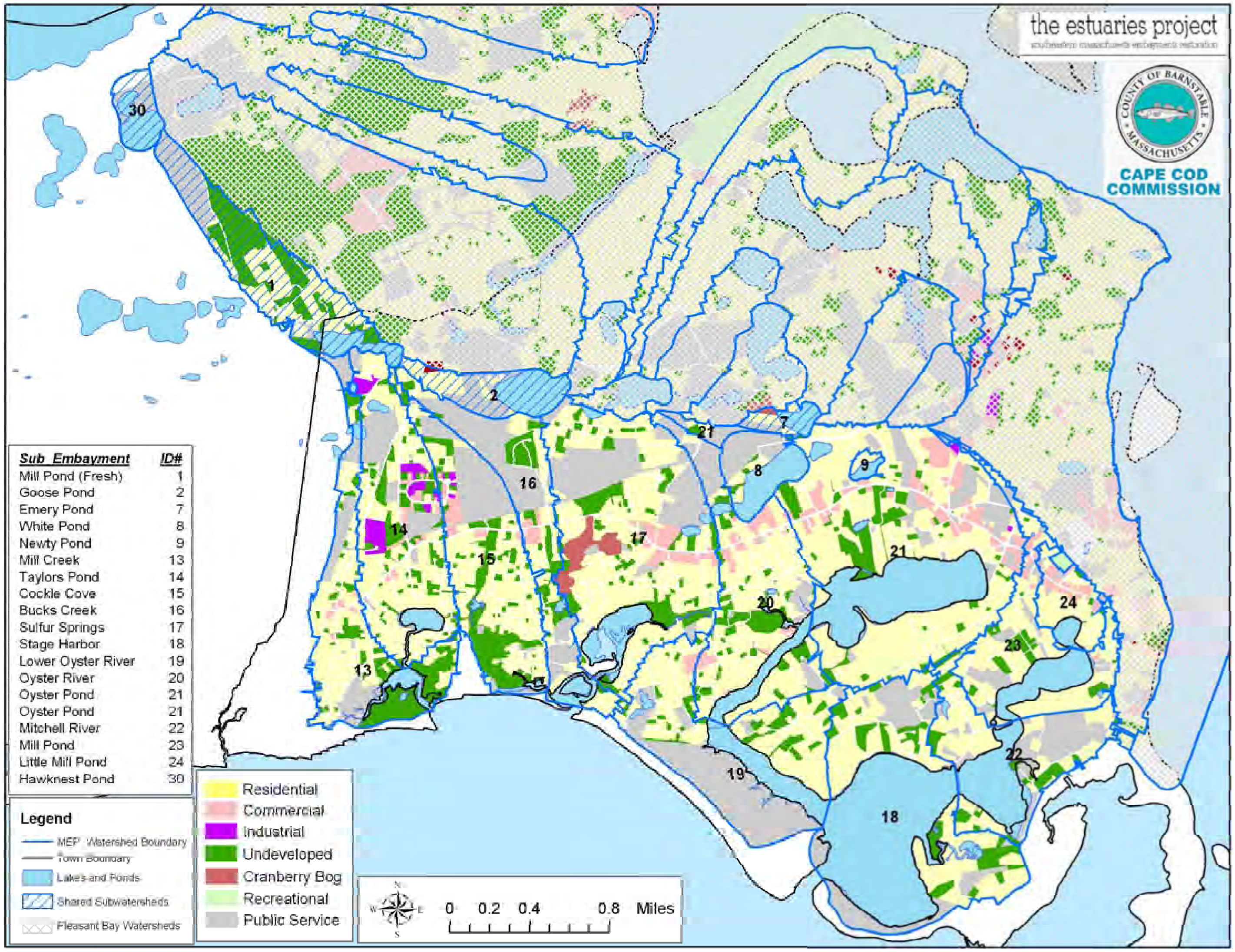


Figure IV-1. Land-use in the Taylors Pond/Mill Creek, Sulfur Springs/Bucks Creek, and Stage Harbor/Oyster Pond estuary watersheds. Land use classifications are based on assessors' records provided by the town. Assigned watershed numbers are same as Howes et al (2003); watersheds shared with Pleasant Bay have watershed numbers assigned in Howes et al (2006).

IV.1.1 Land Use and Water Use Database Preparation

Estuaries Project staff obtained digital parcel and tax assessors data from the Town of Chatham. Digital parcel and land use data are from 2004 and were obtained from the Chatham Department of Health & Environment. The land use database contains traditional information regarding land use classifications (MADOR, 2002) plus additional information developed by the Town. The parcel data and assessors' databases were combined by the Town and were brought into the Cape Cod Commission Geographic Information System (GIS) for the MEP analysis.

Figure IV-1 shows the land uses within the southern Chatham estuaries study area. Land use in the study area is one of eight land use categories: 1) residential, 2) commercial, 3) industrial, 4) undeveloped, 5) agricultural, 6) mixed use, 7) public service/government, including road rights-of-way, and 8) freshwater ponds. These land use categories, except the ponds, are aggregations derived from the major categories in the Massachusetts Assessors land uses classifications (MADOR, 2002). These categories are common to each town in the watershed. “Public service” in the MADOR system includes tax exempt properties, including lands owned by government (e.g., wellfields, schools, open space, roads) and private groups like churches and colleges.

In each of the watersheds to the three southern Chatham estuaries, the predominant land use based on area is residential, which accounts for 38% of the watershed area to Taylors Pond/Mill Creek, 39% to Sulfur Springs/Bucks Creek, and 55% to Stage Harbor/Oyster Pond. Public service (government owned lands, roads, and rights-of-way) is the second highest percentage in all the watersheds, 36%, 38%, and 34%, respectively (Figure IV-2). Residential parcels are also the highest percentage among the parcel count with 75% of the parcels in the Taylors Pond/Mill Creek watershed classified as residential, 75% within the Sulfur Springs/Bucks Creek watershed, and 72% in the Stage Harbor/Oyster Pond watershed. Single-family residences (MADOR land use code 101) are 88 to 96% of the residential parcels and single-family residences are 42 to 90% of the residential land area. Undeveloped land uses are the third highest percentage land use in all watersheds, ranging from 5% in the Stage Harbor watershed to 16% in the other two watersheds. Commercial properties account for only 1% to 3% of the area of each watershed.

In order to estimate wastewater flows within the southern Chatham estuaries study area, MEP staff also obtained parcel-by-parcel water use information from the Chatham Department of Health and Environment. Chatham data is annualized water consumption between 2002 and 2003. Water use information was linked to the parcel and assessors data using GIS techniques. In addition, information on flow, effluent quality, and the service area delineation for the Chatham Wastewater Treatment Facility (WWTF) were also obtained. Wastewater-based nitrogen loading from the individual parcels using on-site septic systems is based upon the measured water-use, nitrogen concentration, and an assumed consumptive loss of water before the remainder is treated in a septic system.

IV.1.2 Nitrogen Loading Input Factors

Wastewater/Water Use

The Massachusetts Estuaries Project septic system nitrogen loading rate is fundamentally based upon a per Capita Nitrogen Load to the receiving aquatic system. Specifically, the MEP septic system wastewater nitrogen loading is based upon a number of studies and additional information that directly measured septic system and per capita loads on Cape Cod or in similar geologic settings (Nelson et al. 1990, Weiskel & Howes 1991, 1992, Koppelman 1978, Frimpter et al. 1990, Brawley et al. 2000, Howes and Ramsey 2000, Costa et al. 2001). Variation in per capita nitrogen load has been found to be relatively small, with average annual per capita nitrogen loads generally between 1.9 to 2.3 kg person-yr^{-1}.

However, given the seasonal shifts in occupancy and rapid population growth throughout southeastern Massachusetts, decennial census data yields accurate estimates of total population only in selected watersheds. To correct for this uncertainty and more accurately assess current nitrogen loads, the MEP employs a water-use approach. The water-use approach is applied on a parcel-by-parcel basis within a watershed, where annual water meter data is linked to assessors parcel information using GIS techniques. The parcel specific water use data is converted to septic system nitrogen discharges (to the receiving aquatic systems) by adjusting for consumptive use (e.g. irrigation) and applying a wastewater nitrogen concentration. The water use approach focuses on the nitrogen load, which reaches the aquatic receptors down-gradient in the aquifer.

All nitrogen losses within the septic system are incorporated. For example, information developed at the DEP Alternative Septic System Test Center at the Massachusetts Military Reservation on Title 5 septic systems have shown nitrogen removals between 21% and 25%. Multi-year monitoring from the Test Center has revealed that nitrogen removal within the septic tank was small (1% to 3%), with most (20 to 22%) of the removal occurring within five feet of the soil adsorption system (Costa et al. 2001). Down-gradient studies of septic system plumes indicate that further nitrogen loss during aquifer transport is negligible (Robertson et al. 1991, DeSimone and Howes 1996).

In its application of the water-use approach to septic system nitrogen loads, the MEP has ascertained for the Estuaries Project region that while the per capita septic load is well constrained by direct studies, the consumptive use and nitrogen concentration data are less certain. As a result, the MEP has derived a combined term the effective N Loading Coefficient (consumptive use times N concentration) of 23.63, to convert water (per cubic meter) to nitrogen load (N grams). This coefficient uses a per capita nitrogen load of 2.1 kg N person-yr^{-1} and is based upon direct measurements and corrects for changes in concentration that result from per capita shifts in water-use (e.g. due to installing low plumbing fixtures or high versus low irrigation usage, etc.).

The nitrogen loads developed using this approach have been validated in a number of long and short term field studies where integrated measurements of nitrogen discharge from watersheds could be directly measured. Weiskel and Howes (1991, 1992) conducted a detailed watershed/stream tube study that monitored septic systems, leaching fields and the transport of the nitrogen in groundwater to adjacent Buttermilk Bay. This monitoring resulted in estimated annual per capita nitrogen loads of 2.17 kg (as published) to 2.04 kg (if new attenuation information is included). Modeled and measured nitrogen loads were determined for a small sub-watershed to Mashapaquit Creek in West Falmouth Harbor (Smith and Howes 2006) where measured nitrogen discharge from the aquifer was within 5% of the modeled N load. Another evaluation was conducted by surveying nitrogen discharge to the Mashpee River in reaches with swept sand channels and in winter when nitrogen attenuation is minimal. The modeled and observed loads showed a difference of less than 8%, easily attributable to the low rate of attenuation expected at that time of year and under the ecological situation (Samimy and Howes, unpublished data).

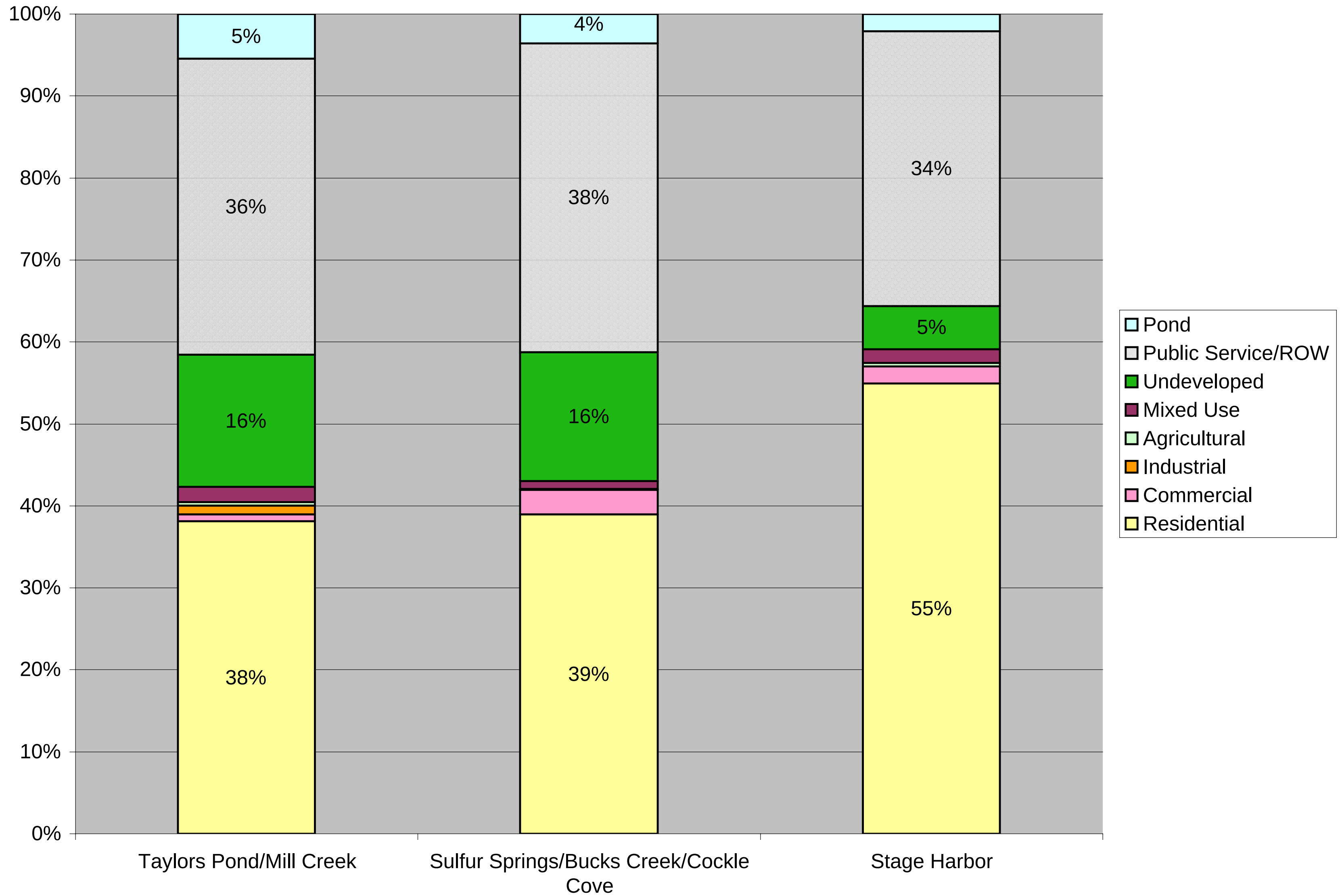


Figure IV-2. Distribution of land-uses within the watersheds to the southern Chatham estuaries.

While census based population data has limitations in the highly seasonal MEP region, part of the regular MEP analysis is to compare expected water used based on average residential occupancy to measured average water uses. This is performed as a quality assurance check to increase certainty in the final results. This comparison has shown that the larger the watershed, the better the match between average water use and occupancy. For example, in the cases of the combined Great Pond, Green Pond and Bournes Pond watershed in the Town of Falmouth and the Popponesset Bay/Eastern Waquoit Bay watershed, which cover large areas and have significant year-round populations, the septic nitrogen loading based upon the census data is within 5% of that from the water use approach. This comparison matches some of the variability seen in census data itself, census blocks, which are generally smaller areas of a given town, have shown up to a 13% difference in average occupancy form town-wide occupancy rates. These analyses provide additional support for the use of the water use approach in the MEP study region.

Overall, the MEP water use approach for determining septic system nitrogen loads has been both calibrated and validated in a variety of watershed settings. The approach: (a) is consistent with a suite of studies on per capita nitrogen loads from septic systems in sandy outwash aquifers; (b) has been validated in studies of the MEP Watershed "Module", where there has been excellent agreement between the nitrogen load predicted and that observed in direct field measurements corrected to other MEP Nitrogen Loading Coefficients (e.g, stormwater, lawn fertilization); (c) the MEP septic nitrogen loading coefficient agrees in specific studies of consumptive water use and nitrogen attenuation between the septic tank and the discharge site; and (d) the watershed module provides estimates of nitrogen attenuation by freshwater systems that are consistent with a variety of ecological studies. It should be noted that while these points support the use of the MEP Septic N Coefficient, they were not used in its development. The MEP Technical Team has developed the septic system nitrogen load over many years, and the general agreement among the number of supporting studies has greatly enhanced the certainty of this critical watershed nitrogen loading term.

The independent validation of the water quality model (Section VI) adds additional weight to the nitrogen loading coefficients used in the MEP analyses and a variety of other MEP embayments. While the MEP septic system nitrogen load is the best estimate possible, to the extent that it may underestimate the nitrogen load from this source reaching receiving waters provides a safety factor relative to other higher wastewater loading coefficients that are generally used in regulatory situations. The MEP coefficient results in slightly higher amounts of nitrogen mitigation (estimated at 1% to 5%) needed to lower embayment nitrogen levels to a nitrogen target (e.g. nitrogen threshold, cf. Section VIII). The additional nitrogen removal is not proportional to the septic system nitrogen level, but is related to the how the septic system nitrogen mass compares to the nitrogen loads from all other sources that reach the estuary (i.e. attenuated loads).

During the development of the previous MEP technical report on the southern Chatham estuaries (Howes et al, 2003), the original watershed nitrogen loadings were developed using three quarters of water use adjusted to an annual rate. Although this was a mutual decision among town staff and the project team, subsequent review of water use showed that inclusion of the fourth quarter of water use reduced the overall water use rate by 15%. This difference led to this revised analysis.

In order to maintain a balance between wastewater nitrogen loads based on water use and the measured nitrogen in the estuaries, the MEP Technical Team reviewed a number of characteristics of the water use dataset that are generally not reviewed during MEP analysis.

This review was coordinated through extensive interaction with town staff and included a review of billing abatements and water meter performance as well as comparisons between the completed four-quarter dataset from the original analysis and the 2002/03 dataset and between the north and south sides of town. These analyses found that significantly more abatements were granted in southern Chatham in the new dataset and that town water meters (n=477) overall record 4% less water than actually flows through them. Water uses on individual parcels and collectively in the individual estuary watersheds were adjusted to account for these findings.

In order to provide an additional independent validation of the residential water use average within the southern Chatham estuaries study area, MEP staff reviewed US Census population values in Chatham. The state on-site wastewater regulations (*i.e.*, 310 CMR 15, Title 5) assume that two people occupy each bedroom and each bedroom has a wastewater flow of 110 gallons per day (gpd), so each person generates 55 gpd of wastewater. Based on data collected during the 2000 US Census, average occupancy within Chatham is 2.1 people per household, while year-round occupancy of available housing units is 47%. Average water use for single-family residences with municipal water accounts in the three watersheds is 128 gpd. If this flow is multiplied by 0.9 to account for consumptive use, the watershed average is 117 gpd. If this flow is then divided by 55 gpd, the average estimated occupancy in the watershed is 2.1 people per household. This simple comparison between population and water use shows a good match and provides further validation for the use of water use data for calculating wastewater nitrogen loads.

Although water use information exists for 88% of the over 3,600 developed parcels in the watersheds to the southern Chatham estuaries, there are 424 parcels (12%) that are assumed to utilize private wells for drinking water. These are properties that were classified with land use codes that should be developed (e.g., 101 or 325), have been confirmed as having buildings on them through a review of aerial photographs, and do not have a listed account in the water use databases. Of the 424 parcels, 78% of them (331) are classified as single-family residences (land use code 101) and another 10% are classified as other types of residential development (e.g. 109 - multiple houses on a single property). The remaining 12% are commercial (6% of the total), industrial, or tax-exempt (e.g., 900's in state class code). MEP staff used current water use to develop a watershed-specific water use estimate for land uses assumed to utilize private wells (Table IV-2).

Table IV-2. Average Water Use in three southern Chatham estuary watersheds

Land Use	State Class Codes	# of Parcels with Water Use in Watershed	Water Use per parcel (gallons per day)	
			Watershed Average	Subwatershed Average Range
Residential	101	2,535	128	92 to 220
Commercial	300 to 389	60	430	6 to 3,015
Industrial	400 to 439	14	253	12 to 1,537

Note: All data for analysis supplied by Town of Chatham.

Town of Chatham Wastewater Treatment Facility

The Town of Chatham maintains a municipal wastewater treatment facility (WWTF) with discharge basins within the Cockle Cove sub-watershed of the Sulfur Springs/Bucks Creek estuarine system. The WWTF imports wastewater from a sewer collection system generally concentrated in the main town center and the Stage Harbor/Oyster Pond watershed, treats it, and discharges the treated effluent within the sub-watershed to the Cockle Cove Creek salt marsh. MEP staff obtained seven years (1999-2005) worth of influent and effluent flow information, including effluent total nitrogen concentrations, from the Town (B. Duncanson, personal communication, 4/27/06) to review nitrogen loading from the WWTF.

Review of the provided data showed that nitrogen loading from the current WWTF has generally been declining between 1999 and 2005, except for a large increase (35% above average) in 2003. Groundwater modeling in the area has shown that flow from the plant to down gradient Cockle Cove is fairly rapid (0-5 years) (D. Walter, USGS, personal communication) and that existing groundwater monitoring wells are insufficient to adequately characterize a plume from the WWTF. Since the water use dataset is from the period of the large nitrogen loading increase and since the groundwater travel times suggest that loads may reach the Cove within the same season of discharge, project staff focused on an average of the 2002 and 2003 loads as the basis for the WWTF nitrogen loading. The average annual nitrogen load from the WWTF during this period was 1,170 kg.

Nitrogen Loading Input Factors: Fertilized Areas

The second largest source of estuary watershed nitrogen loading is usually fertilized lawns and golf courses, with lawns being the predominant source within this category. In order to add this source to the nitrogen loading model for the three southern Chatham estuary systems, MEP staff reviewed available information about residential lawn fertilizing practices and incorporated site-specific fertilizer application rates for large tracts of turf, such as golf courses, by contacting turf managers.

Residential lawn fertilizer use has rarely been directly measured in watershed-based nitrogen loading investigations. Instead, lawn fertilizer nitrogen loads have been estimated based upon a number of assumptions: a) each household applies fertilizer, b) cumulative annual applications are 3 pounds per 1,000 sq. ft., c) each lawn is 5000 sq. ft., and d) only 25% of the nitrogen applied reaches the groundwater (leaching rate). Because many of these assumptions had not been rigorously reviewed in over a decade, the MEP Technical Staff undertook an assessment of lawn fertilizer application rates and a review of leaching rates for inclusion in the Watershed Nitrogen Loading Sub-Model.

The initial effort in this assessment was to determine nitrogen fertilization rates for residential lawns in the Towns of Falmouth, Mashpee and Barnstable. The assessment accounted for proximity to fresh ponds and embayments. Based upon ~300 interviews and over 2,000 site surveys, a number of findings emerged: 1) average residential lawn area is ~5000 sq. ft., 2) half of the residences did not apply lawn fertilizer, and 3) the weighted average application rate was 1.44 applications per year, rather than the 4 applications per year recommended on the fertilizer bags. Integrating the average residential fertilizer application rate with a leaching rate of 20% results in a fertilizer contribution of N to groundwater of 1.08 lb N per residential lawn; these factors are used in the MEP nitrogen loading calculations. It is likely that this still represents a conservative estimate of nitrogen load from residential lawns. It should be noted that professionally maintained lawns were found to have the higher rate of fertilization application and hence higher estimated loss to groundwater of 3 lb/lawn/yr.

There is only one golf course within the watersheds to the three southern Chatham estuaries and it is Chatham Seaside Links. A portion of this golf course (~1 acre) is located within the subwatershed to Little Mill Pond. Table IV-3 summarizes the fertilizer application rates used for this golf course in the watershed nitrogen-loading model.

Nitrogen Loading Input Factors: Other

The nitrogen loading factors for atmospheric deposition, impervious surfaces and natural areas are from the MEP Embayment Modeling Evaluation and Sensitivity Report (Howes and Ramsey 2001). The factors are similar to those utilized by the Cape Cod Commission's Nitrogen Loading Technical Bulletin (Eichner and Cambareri, 1992) and Massachusetts DEP's Nitrogen Loading Computer Model Guidance (1999). The recharge rate for natural areas and lawn areas is the same as utilized in the MEP-USGS groundwater modeling effort (Section III). Cranberry bog fertilizer application rate and percent nitrogen attenuation in the bogs is based on the only annual study of nutrient cycling and loss from cranberry agriculture (Howes and Teal, 1995). Only the bog loses measurable nitrogen, the forested upland release only very low amounts. For the watershed nitrogen loading analysis, the areas of active bog surface are based on 85% of the total area for properties classified as cranberry bogs in the town-supplied land use classifications. Factors used in the MEP nitrogen loading analysis for the three southern Chatham estuary watersheds are summarized in Table IV-3.

Table IV-3. Primary Nitrogen Loading Factors used in the MEP analyses for the three southern Chatham estuaries. General factors are from MEP modeling evaluation (Howes & Ramsey 2001). Site-specific factors are derived from Chatham data. *Data from MEP lawn study in Falmouth, Mashpee & Barnstable 2001.

Nitrogen Concentrations:	mg/l	Recharge Rates:	in/yr
Road Run-off	1.5	Impervious Surfaces	40
Roof Run-off	0.75	Natural and Lawn Areas	27.25
Direct Precipitation on Embayments and Ponds	1.09	Water Use/Wastewater:	
Natural Area Recharge	0.072	Existing developed parcels wo/water accounts:	128 gpd
Wastewater Coefficient	23.63		
Fertilizers:			
Average Residential Lawn Size (ft^2)*	5,000		
Residential Watershed Nitrogen Rate (lbs/lawn)*	1.08	Existing developed parcels w/water accounts:	Measured annual water use
Golf course nitrogen application (lbs/ac)	3	Buildout Parcels Assumptions:	
Fertilizer nitrogen leaching rate	20%	Residential parcels:	128 gpd
Cranberry Bogs nitrogen application (lbs/ac)	31	Commercial parcels:	667 gpd
Cranberry Bogs nitrogen attenuation	34%	Industrial parcels:	447 gpd

IV.1.3 Calculating Nitrogen Loads

Once all the land and water use information was linked to the parcel coverages, parcels were assigned to various watersheds based initially on whether at least 50% or more of the land area of each parcel was located within a respective watershed. Following the assigning of boundary parcels, all large parcels were examined individually and were split (as appropriate) in order to obtain less than a 2% difference between the total land area of each sub-watershed and the sum of the area of the parcels within each sub-watershed. The resulting “parcelized” watersheds to three southern Chatham estuaries are shown in Figure IV-3.

The review of individual parcels straddling watershed boundaries included corresponding reviews and individualized assignment of nitrogen loads associated with lawn areas, septic systems, and impervious surfaces. Individualized information for parcels with atypical nitrogen loading (condominiums, golf courses, etc.) were also assigned at this stage. It should be noted that small shifts in nitrogen loading due to the above assignment procedure generally have a negligible effect on the total nitrogen loading to the three southern Chatham estuaries. The assignment effort was undertaken to better define the sub-embayment loads and enhance the use of the Linked Watershed-Embayment Model for the analysis of management alternatives.

Following the assignment of all parcels, sub-watershed modules were generated for each of the 30 sub-watersheds summarizing water use, parcel area, frequency, sewer connections, private wells, and road area. As mentioned above, these results were then condensed to 19 sub-watersheds based upon the time of travel analysis (less than 10 years vs. greater than 10 years) discussed above. The individual sub-watershed modules were then integrated to create a Nitrogen Loading module with summaries for each of the individual estuaries. The sub-embayments within the larger estuary systems represent the functional embayment units for the Linked Watershed-Embayment Model’s water quality component.

For management purposes, the aggregated embayment watershed nitrogen loads are partitioned by the major types of nitrogen sources in order to focus development of nitrogen management alternatives. Within the three southern Chatham estuaries, the major types of nitrogen loads are: wastewater (e.g., septic systems and the municipal WWTF), fertilizer, impervious surfaces, direct atmospheric deposition to water surfaces, and recharge within natural areas (Table IV-4). The output of the watershed nitrogen-loading model is the annual mass (kilograms) of nitrogen added to the contributing area of component sub-embayments, by each source category (Figure IV-4 a-c). In general, the annual watershed nitrogen input to the watershed of an estuary is then adjusted for natural nitrogen attenuation through ponds and/or streams during transport to the estuarine system before use in the embayment water quality sub-model.

Pond-attenuated nitrogen loads are divided among to respective down-gradient watersheds using the percentage of the length of shoreline on the down-gradient side that borders each down-gradient sub-watershed. So for example, White Pond has a down-gradient shoreline of 1,098 meters; 47% of that shoreline discharges to Oyster River (watershed 20 in Figure IV-1) and 53% goes to the Oyster Pond (watershed 21 in Figure IV-1). The attenuated nitrogen load discharging from White Pond is divided among these sub-watersheds based on these percentages of the down-gradient shoreline.

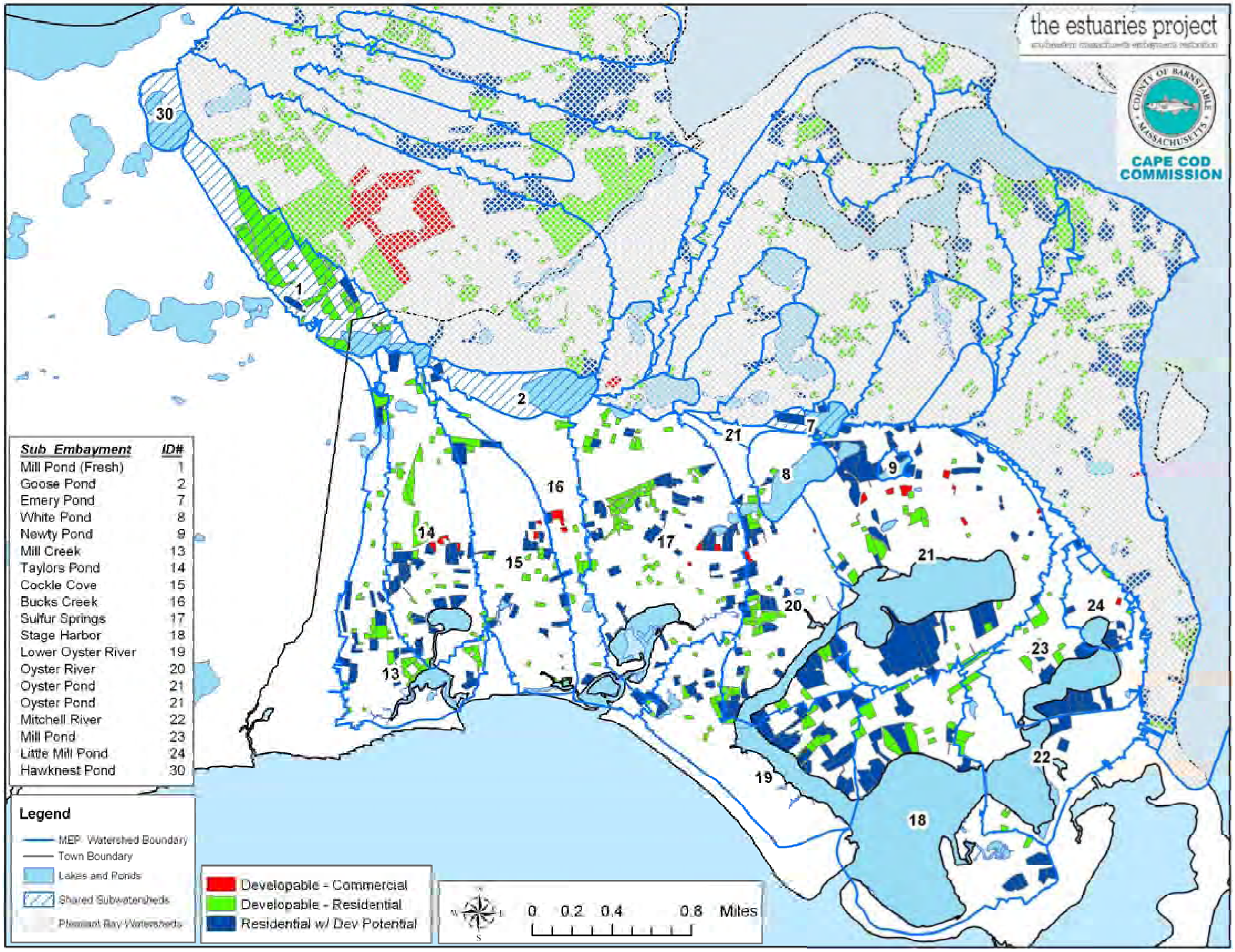
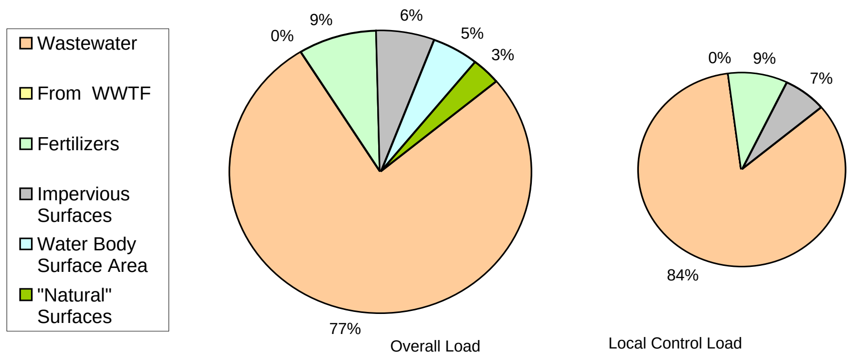


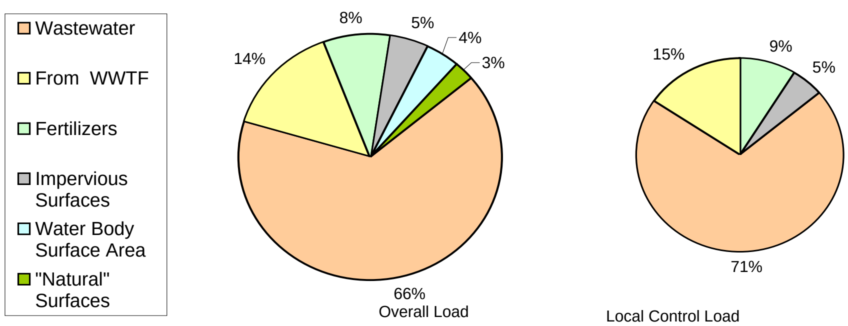
Figure IV-3. Potentially Developable Parcels in the Taylors Pond/Mill Creek, Sulfur Springs/Bucks Creek, and Stage Harbor/Oyster Pond estuary watersheds. Development potential was determined by the Town of Chatham, using assessor land use classifications and current zoning. Assigned watershed numbers are same as Howes et al (2003); watersheds shared with Pleasant Bay have watershed numbers assigned in Howes et al (2006).

Table IV-4. Nitrogen Loads to Taylors Pond/Mill Creek, Sulfur Springs/Bucks Creek, and Stage Harbor/Oyster Pond estuary watersheds. Attenuation nitrogen loads occurs as nitrogen moves through upgradient ponds during transport to the estuaries. The Buildout nitrogen loads are discussed in the section below.

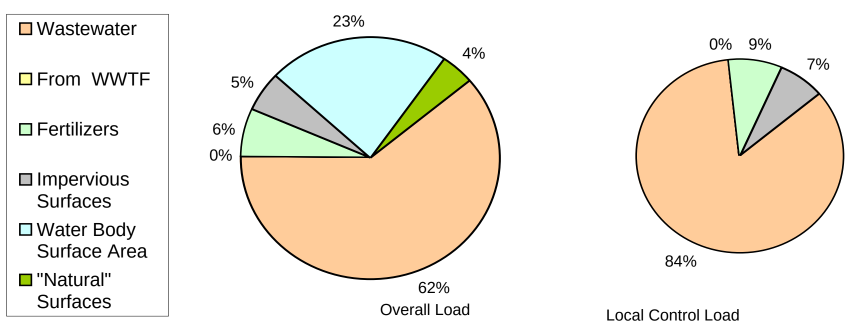
		Chatham Estuary N Loads by Input (kg/y):								***Present N Loads***			***Buildout N Loads***		
Name	**Watershed ID#**	**Wastewater**	**From WWTF**	**Fertilizers**	**Impervious Surfaces**	**Water Body Surface Area**	**"Natural" Surfaces**	**Buildout**	**% of Pond Outflow**	**UnAtten N Load**	**Atten %**	**Atten N Load**	**UnAtten N Load**	**Atten %**	**Atten N Load**
Taylor Pond/Mill Creek System		**3263**	**0**	**362**	**264**	**208**	**134**	**1306**		**4232**		**4028**	**5538**		**5278**
Mill Creek		1353	0	163	113	94	64	438		1786		1712	2224		2129
Mill Pond Fresh Total MPF + HP		89	0	13	11	27	8	41	22%	148	50%	74	189	50%	95
Mill Creek Estuary surface deposition						61				61		61	61		61
Taylors Pond		1910	0	199	151	115	70	869		2446		2316	3314		3149
Mill Pond Fresh Total MPF + HP		156	0	22	19	47	15	72	38%	259	50%	129	330	50%	165
Taylors Pond Estuary surface deposition						68				68		68	68		68
Sulfur Springs/Cockle Cove/ Bucks Creek System		**5345**	**1170**	**670**	**387**	**340**	**203**	**1685**		**8115**		**7983**	**9800**		**9653**
Cockle Cove		1574	1170	169	118	29	54	470		3114		3096	3584		3561
Mill Pond Fresh Total MPF + HP		22	0	3	3	6	2	10	5%	36	50%	18	46	50%	23
Cockle Cove Estuary surface deposition						22				22		22	22		22
Bucks Creek		1039	0	86	63	97	49	390		1334		1271	1723		1653
Goose Pond Total GP + MPF		33	0	4	2	41	3	3	22%	83	50%	42	87	50%	43
Mill Pond Fresh Total MPF + HP		26	0	4	3	8	2	12	6%	42	50%	21	54	50%	27
Bucks Creek Estuary surface deposition						48				48		48	48		48
Sulfur Springs		2731	0	415	206	214	101	825		3667		3616	4492		4439
Goose Pond Total GP + MPF		41	0	5	3	51	3	4	27%	102	50%	51	107	50%	53
Sulfur Springs Estuary surface deposition						138				138		138	138		138
Stage Harbor System		**8346**	**0**	**863**	**712**	**3095**	**574**	**2814**		**13590**		**13357**	**16404**		**16153**
Stage Harbor		556	0	60	51	1184	63	319		1914		1914	2233		2233
Stage Harbor Estuary surface deposition						1184				1184		1184	1184		1184
Oyster River		2597	0	277	223	494	309	892		3901		3816	4793		4702
White Pond Totals	8	45	0	4	19	96	5	12	47%	169	50%	85	181	50%	90
Lower Oyster River Estuary surface deposition						230				230		230	230		230
Oyster River Estuary surface deposition						155				155		155	155		155
Oyster Pond		3011	0	271	248	820	113	1261		4465		4316	5726		5566
White Pond Totals	8	52	0	4	19	96	5	13	53%	176	50%	88	189	50%	95
Newty Pond	9	45	0	3	1	28	1	5	100%	78	50%	39	83	50%	42
Emery Pond	7	15	0	2	1	24	1	6	38%	43	50%	21	48	50%	24
Oyster Pond Estuary surface deposition						651				651		651	651		651
Mitchell River		791	0	63	55	322	34	95		1265		1265	1360		1360
Mitchell River Estuary surface deposition						322				322		322	322		322
Mill Pond Salt		1390	0	191	135	274	55	247		2045		2045	2292		2292
Little Mill Pond Estuary surface deposition						44				44		44	44		44
Mill Pond Salt Estuary surface deposition						229				229		229	229		229



a. Taylors Pond/Mill Creek System



b. Sulfur Springs/Bucks Creek System



c. Stage Harbor System

Figure IV-4 (a-c). Land use-specific unattenuated nitrogen load (by percent) to the 3 estuaries, (a) Taylors Pond/Mill Creek, (b) Sulfur Springs/Bucks Creek, and (c) Stage Harbor. "Overall Load" is the total nitrogen input from the watershed plus atmospheric deposition to the estuary surface, while the "Local Control Load" represents only those nitrogen sources from the watershed, which are potentially under local regulatory control.

Freshwater Pond Nitrogen Loads

Freshwater ponds on Cape Cod are generally kettle hole depressions that penetrate the surrounding groundwater table revealing what some call “windows on the aquifer.” Groundwater typically flows into the pond along the up-gradient shoreline, then lake water flows back into the groundwater system along the down-gradient shoreline. Occasionally a Cape Cod pond will have a stream outlet or herring run too. Since the nitrogen loads flow into the pond with the groundwater, the relatively more productive ecosystems in the ponds incorporate some to the nitrogen, retain some of it in the sediments, and change it among its various oxidized and reduced forms. As result of these interactions, some of the nitrogen is removed from the watershed system, mostly through burial in the sediments and denitrification that returns it to the atmosphere. Following these reductions, the remaining, reduced or attenuated loads flow back into the groundwater system along the down-gradient side of the pond or through a stream outlet eventually discharging to the down-gradient embayment. The nitrogen load summary in Table IV-4 includes both the unattenuated (nitrogen load to each sub-watershed) and attenuated nitrogen loads.

Pond nitrogen attenuation in freshwater ponds is generally assumed to be 50% in MEP analyses; in some cases, if sufficient monitoring information is available, an alternative attenuation rate is incorporated into the watershed nitrogen loading modeling (Howes et al, 2005). Detailed studies of other southeastern Massachusetts freshwater systems including Ashumet Pond (AFCEE, 2000) and Agawam/Wankinco River Nitrogen Discharges (CDM, 2001) have supported a 50% attenuation factor. In order to estimate nitrogen attenuation in the ponds physical and chemical data for each pond is reviewed. Available bathymetric information is reviewed relative to measured pond temperature profiles to determine whether an epilimnion (*i.e.*, well mixed, homothermic, upper portion of the water column) exists in each pond. Bathymetric information is necessary to develop a residence or turnover time and complete an estimate of nitrogen attenuation. Of the six ponds situated within the watersheds to the three southern Chatham estuaries, bathymetric information is available for: Emory, Goose, Mill, and White. Of the ponds with bathymetric information, Goose and White are deep enough to develop strong temperature stratification and a separate epilimnion. Generally, if a stable epilimnion develops, it is the appropriate volume for gauging nitrogen attenuation in a pond, since it is separate from the lower thermal layers, which are, in turn, usually impacted by sediment regeneration of nitrogen and are in less contact with the groundwater flow system.

In MEP analyses, available nitrogen concentrations from individual ponds are reviewed to establish whether there is a significant gradient (higher at bottom) in a pond and, if not, the entire volume of the pond is used to determine a turnover time. Turnover time is how long it takes the recharge from the up-gradient watershed to completely exchange the water in the pond or, in the case of a thermally stratified pond with a significant vertical nitrogen gradient, exchange is estimated from the epilimnion. The total mass of nitrogen in the pond or epilimnion is adjusted using the pond turnover time to determine the annual nitrogen load returned to the aquifer through the down-gradient shoreline. This mass is then compared to the nitrogen load coming from the pond’s watershed to determine the nitrogen attenuation factor for the pond. Generally, there is not sufficient monitoring data to support use of a factor different than the standard 50% attenuation. In these cases the pond data is developed to "validate" the attenuation rate used in the model. Table IV-5 presents available turnover times and attenuation factors for the six ponds with sub-watersheds within the overall watersheds to the three southern Chatham estuaries.

The standard attenuation assumption for the six ponds situated within the watersheds to the three southern Chatham estuaries was checked through the use of pond water quality information collected from the annual Cape Cod Pond and Lake Stewardship (PALS) water quality snapshot. The PALS Snapshot is a collaborative between the Water Resources Office of the Cape Cod Commission and the School for Marine Science and Technology within UMASS-Dartmouth. This Program allows trained, citizen volunteers of each of the 15 Cape Cod towns to collect pond samples in August and September using a standard protocol with analysis using accepted methods and QA/QC protocols. Snapshot samples have been collected every year between 2001 and 2005. The standard protocol for the Snapshot includes field collection of dissolved oxygen and temperature profiles, Secchi disk depth readings and water samples for nutrient related water quality analyses at various depths depending on the total depth of the pond. Water samples were analyzed at the SMAST laboratory for total nitrogen, total phosphorus, chlorophyll *a*, alkalinity, and pH. PALS Snapshot data is available for each of the six ponds with delineated watersheds within the watersheds to the three southern Chatham estuaries. In addition, except for Newty/Perch Pond which has only 2 years of data, all of the ponds have water quality data from all five PALS Snapshots. Table IV-5 summarizes the cumulative number of nitrogen samples for each pond available for review from the PALS Snapshot sampling. Nitrogen attenuation estimates for the ponds reviewed in the study area vary between 21 and 86%.

The attenuated nitrogen loads in Table IV-5 include pond attenuation based on the 50% assumption. Since each pond has this assigned attenuation factor, nitrogen loads in the Watershed Nitrogen Loading model can be subject to a number of attenuation steps as loads flow into the down-gradient aquifer from one pond and then into another pond.

Table IV-5. Nitrogen attenuation by Freshwater Ponds in the southern Chatham estuary watersheds based upon 2001 through 2005 Cape Cod Pond and Lakes Stewardship (PALS) program sampling. These data were collected to provide a site-specific check on nitrogen attenuation by these systems. The MEP Linked N Model for Pleasant Bay uses a standard value of 50% for the pond systems.

Pond	PALS ID	Area acres	Maximum Depth m	Overall turnover time yrs	# of TN samples	N Load Attenuation %
Emery	CH-491	14.1	6.2	1.8	11	21%
Goose	CH-458	41.2	11.0	4.0	18	79%
Mill	CH-440	23.4	2.8	0.2	10	45%
White	CH-516	40.2	17.0	3.2	15	68%
Newty/Perch	CH-522	5.5	1.7	0.5	3	68%
Hawknest	HA-354	27.6	8.6	3.3	9	86%
				Mean		61%
				std dev		24%

Data sources: all areas from CCC GIS; Max Depth from MADFW bathymetric maps or maximum measured station depth in Cape Cod PALS monitoring; Volume for turnover time calculations for Emery, Goos, Mill and White Ponds use MADFW bathymetric maps (www.mass.gov/dfwele/dfw/dfw_pond.htm), while Newty/Perch and Hawknest Ponds were estimated based on best professional judgment from PALS field data; TN concentrations for attenuation calculation from PALS monitoring; attenuation based on unattenuated loads shown in Table IV-4

Buildout

Part of the regular MEP watershed nitrogen loading modeling is to prepare a buildout assessment of potential development within the study area watershed. For the watershed modeling for the three southern estuaries within the Town of Chatham, The MEP Technical Team staff obtained buildout estimates from the Chatham Department of Health and Environment. This dataset is the same one that was used in the Pleasant Bay MEP technical review (Howes, et al, 2005). Buildout additions will increase unattenuated nitrogen loading within the estuary watersheds to Taylors Pond/Mill Creek, Sulfur Springs/Bucks Creek, and Stage Harbor System by 31%, 21%, and 21%, respectively (see Table IV-4).

IV.2 ATTENUATION OF NITROGEN IN SURFACE WATER TRANSPORT

IV.2.1 Background and Purpose

Modeling and predicting changes in coastal embayment nitrogen related water quality is based, in part, on determination of the inputs of nitrogen from the surrounding contributing land or watershed. This watershed nitrogen input parameter is the primary term used to relate present and future loads (build-out or sewering analysis) to changes in water quality and habitat health. Therefore, nitrogen loading is the primary threshold parameter for protection and restoration of estuarine systems. Rates of nitrogen loading to the sub-watersheds of each sub-embayment of the 3 embayment systems under study was based upon the delineated watersheds (Section III) and their land-use coverages (Section IV.1). If all of the nitrogen applied or discharged within a watershed reaches an embayment the watershed land-use loading rate represents the nitrogen load to the receiving waters. This condition exists in watershed in which nitrogen transport is through groundwater in sandy outwash aquifers. The lack of nitrogen attenuation in these aquifer systems results from the lack of biogeochemical conditions needed for supporting nitrogen sorption and denitrification. However, in most watersheds in southeastern Massachusetts, nitrogen passes through a surface water ecosystem on its path to the adjacent embayment. Surface water systems, unlike sandy aquifers, do support the needed conditions for nitrogen retention and denitrification. The result is that the mass of nitrogen passing through lakes, ponds, streams and marshes (fresh and salt) is diminished by natural biological processes which represent removal (not just temporary storage). However, this natural attenuation of nitrogen load is not uniformly distributed within the watershed, but is associated with ponds, streams and marshes within the Town of Chatham.

Failure to determine the attenuation of watershed derived nitrogen overestimates the nitrogen load to receiving waters. If nitrogen attenuation is significant in one portion of a watershed and insignificant in another, the result is that nitrogen management would likely be more effective in achieving water quality improvements if focused on the watershed region having unattenuated nitrogen transport (other factors being equal). An example of the significance of nitrogen attenuation relating to embayment nitrogen management was seen in West Falmouth Harbor (Falmouth, MA), where ~40% of the nitrogen discharge to the Harbor originating from the groundwater discharge from the WWTF was attenuated by a small salt marsh prior to reaching Harbor waters. Proper development and evaluation of nitrogen management options requires determination of the nitrogen loads reaching an embayment, not just loaded to the watershed.

The input of nitrogen to Chatham's embayments from the surrounding watersheds is based upon knowing the land area contributing to a particular embayment, quantifying the land-uses, and calculating the nitrogen loading based upon regional measures of nitrogen loading for each land-use. Previous investigations by the Town of Chatham to determine the watershed nitrogen loads indicated that natural attenuation might be occurring in some sub-watersheds. This was based upon Cape Cod Commission watershed nitrogen loading for Chatham embayments presented in the Stearns & Wheler August 1999 Final Needs Assessment Report (updated for the Pleasant Bay embayments in a Memorandum of April 20, 2001). In a study by Applied Coastal Research and Engineering, Inc (2000), both direct observations and nitrogen modeling indicated that nitrogen attenuation was likely in the Cockle Cove sub-watershed to the Sulphur Springs/Bucks Creek System.

In the previous watershed loading studies the watershed delineation's were made by the Cape Cod Commission by surveying water table elevations in available wells (1999 watershed). While this is a powerful approach, it is limited by the distribution of existing wells. A review of the watershed delineation's by the MEP Technical Team, which includes Cape Cod Commission staff, indicated that a revision of watershed and sub-watershed delineations would be necessary in order to accurately quantify watershed based nitrogen load and associated attenuations. Partnership with the United States Geological Survey has allowed for a complete revision of all of the delineations for the hydrologic features contained in Town of Chatham, including all of its coastal embayments. This re-delineation was used for both the present and previous MEP nitrogen loading analysis. The USGS re-delineation effort is described in Section III, above. Based on revised delineations a comprehensive analysis was conducted for nitrogen load determination based on watershed land-use (Section IV.1).

Given the importance of determining accurate nitrogen loads to embayments for developing effective management alternatives and the potentially large errors associated with ignoring natural attenuation, the MEP conducted multiple studies on natural attenuation relating to the 3 embayment systems in the study. Natural attenuation by fresh kettle ponds was addressed above. An additional site-specific study was conducted relative to the Cockle Cove Creek salt marshes, which has been previously presented to the Town (Howes et al. 2006). In addition, a screening approach was applied within Stage Harbor and Cockle Cove Systems (Section IV.2.4.). The Cockle Cove Creek study was conducted by SMAST with significant field and analytical contributions from the Town of Chatham and MCZM (Carlisle et al. 2005).

IV.2.2 Freshwater Inflow and Nitrogen Transport within Cockle Cove Salt Marsh

Freshwater Inflow and Nitrogen Transport to Cockle Cove Creek were determined in a separate study by SMAST (Howes, White & Samimy, 2006. Cockle Cove Salt Marsh Nitrogen Threshold). The draft (March 2006) and Final (November 2006) were previously submitted to and reviewed by the Town, MassDEP and MCZM. However, a brief overview is presented here.

Freshwater inflows to the Cockle Cove Salt Marsh were evaluated using (a) water balance derived from the MEP watershed delineation and recharge from precipitation and WWTF infiltration, (b) measured freshwater inflows from the headwater stream and within the marsh creek, and (c) measured freshwater discharge through the tidal inlet over a tidal cycle. These data and modeling outputs were developed with the Town of Chatham, the Cape Cod Commission and the USGS. In parallel, SMAST, with the assistance of the Chatham Water Quality Laboratory, undertook an analysis of nitrogen levels and transport within Cockle Cove Creek (Figure IV-5). In addition to diffuse watershed nitrogen inputs, Cockle Cove Creek is the primary recipient of treated wastewater effluent from the Town of Chatham's WWTF that discharges to the aquifer near the freshwater stream which forms the headwaters of the central salt marsh creek. Data collection included measurement of nitrogen mass flux and concentration at multiple points along the tidal channel during low tide. These data were used for assessment of the total nitrogen mass flux to Bucks Creek, the determination of nitrogen concentrations available to benthic algae and the present rate of nitrogen removal from the salt marsh prior to discharge to Bucks Creek. The biweekly surveys were supplemented by a tidal study near the outlet from Cockle Cove Creek to Bucks Creek. In conducting the tidal survey the total import and export of nitrogen was determined over a complete tidal cycle.

During the summer of 2005, current velocity measurements were made and water samples were collected during the interval 1 hour before slack low tide at multiple points from the headwaters through the marsh to the outlet at Bucks Creek (Figure IV-5). Water samples were analyzed for nitrogen concentrations (DIN, DON, PON).

Freshwater Inflow: It should be noted by the reader that freshwater analysis (volumetric inflow or spatial distribution) was not part of the SMAST Cockle Cove Study. This section was added based upon concerns raised by the Draft Technical Memorandum to assist the Town of Chatham and MassDEP evaluate potential future issues related to increased freshwater inflow to this system resulting from potential increased WWTF discharges within its watershed. It is not meant as a complete analysis, but does serve as a guide for evaluating future changes in inflow. As work is continuing relative to future WWTF effluent disposal, it is certain that this analysis will need refinement in the coming years.

Total freshwater inflow to Cockle Cove Creek was previously determined by the MEP Technical Team based upon the watershed area, precipitation and recharge. This represents a long-term average freshwater inflow to the Creek of 2335 m^3 d^{-1} or 614,000 gpd. This value agrees well with the net total freshwater outflow through the tidal inlet measured during the tidal study (August 2005), 2420 m^3 d^{-1} or 637,000 gpd (Table IV-6). This latter measurement accounts for both tidal inflow and outflow of freshwater that occurs over a complete flood/ebb cycle, based upon measurements of flow and salinity at 0.5 hr intervals. However, neither of these estimates yields information on the spatial distribution of freshwater inflow to this system. To gain insight into the spatial distribution of freshwater entry to the creek system, flow and salinity measurements collected as part of the nitrogen flux study were used to determine freshwater discharges at 6 locations (CC-1, 2, 3, 4, 4a, 5) within the stream/marsh creek (Figure IV-5).

Freshwater flow during ebb tide in the main tidal creeks showed a pattern typical of tidal marshes in New England. A single freshwater stream discharges to the headwaters of the main tidal creek. Moving down the main tidal creek, additional freshwater volume is encountered due to “pick-up” from groundwater discharge. It appears that two thirds or more of the freshwater inflow occurs within the upper portion of the marsh (above CC-3). In addition, it is clear that the eastern tributary creek is not receiving significant amounts of freshwater inflow. Daily discharge was calculated from the ebb tide data based upon a 20 hr groundwater seepage duration to the tidal creeks and a 24 hr discharge from the entering surface water stream. Unfortunately, estimating the total freshwater outflow (CC-5) was difficult due to the relatively high salinities. While waters at all sites required adjustment for mixing with seawater, the high salinities at the lowest site introduce an additional source of measurement error. Examining the mid-marsh (CC-4A) and the outlet flows relative to the MEP watershed model and tidal study results shows a relatively constrained value for freshwater inflow (Table IV-6) and supports the long-term average value of long-term average freshwater inflow to the Creek of 2335 m3 d-1 or 614,000 gpd.

Table IV-6. Estimate of freshwater flows to Cockle Cove Creek.

Estimates of watershed freshwater inflow to Cockle Cove Creek. Values are m3/d and (1000gpd, 1000's of gallons per day)				
		Freshwater Inflow m3/d (1000gpd)		
Location	ID	Ebb Tide	Tidal Cycle	Watershed
Upper Stream	CC-1	480 (126)	--	--
Marsh Head	CC-2	875 (230)	--	--
Mouth Main Stem	CC-3	1900 (553)	--	--
Mouth East Tributary	CC-4	190 (50)	--	--
Mid Marsh	CC-4A	1930 (508)	--	--
Marsh Outlet	CC-5	3050 (803)	2420 (637)	2335 (614)
* Groundwater inflow based upon 20 hr per day seepage.				

Nutrient levels, flux and attenuation: Cockle Cove Creek receives nitrogen input from its watershed, including treated effluent from disposal at the Town of Chatham WWTF, as well as the atmosphere. The result is high levels of nitrogen in ebbing tidal water from Cockle Cove Creek to Bucks Creek. Levels of inorganic nitrogen, nitrate and ammonium in the fresh headwaters to the estuary averaged 1.791 mg N L^{-1} and 1.104 mg N L^{-1}, respectively and total nitrogen (TN) at 3.154 mg N L^{-1} (Stations CM-J, CC 2, Table IV-7). These values contrast strongly with the offshore inflowing waters which typically have TN values <0.3 mg N L-1 and which are dominated by organic nitrogen forms, rather than the predominance of inorganic nitrogen forms (>90%) in the fresh water inflows.

While the high level of inorganic nitrogen is anticipated in freshwater systems, due to their limitation of plant growth by available phosphorus (primarily ortho-phosphate), the high nitrogen levels in Cockle Cove Creek were also observed in the ebbing tidal creek waters. It appears from the water quality data (Table IV-7) that algal production on the tidal creek bottom is not limited by nitrogen or phosphorus, as the levels of inorganic N and inorganic P remain above 0.3 mg N L-1 and 0.03 mg P L-1 from the headwaters to the outlet to Bucks Creek. These are very high concentrations, which are well above those used to stimulate algal growth. However, macroalgal accumulation was not observed by MCZM, SMAST or Town Staff during their frequent visits to the creek sampling sites.

Nitrogen does appear to be being transformed within the creek waters and sediments as the marsh is exporting particulate organic nitrogen and removing inorganic nitrogen from the waters that pass through it. The biweekly sampling of nitrogen transport showed nitrogen export to Bucks Creek ~46% of that predicted from the MEP watershed land-use model (Table IV-8). In addition, the ebb tide measurements along the main channel of Cockle Cove Creek were indicative of sediment nitrogen uptake. It should be noted that nitrogen enters the creek from its watershed along its length and therefore declines in nitrogen mass transport between individual locations is less dramatic than if the input were solely from the headwaters.

The tidal cycle study yielded consistent results to the ebb tidal samplings. The tidal cycle study measured both the nitrogen import and export from the salt marsh system, during a neap tide, which would minimize the measured nitrogen attenuation rate. Both DIN and Total N concentrations decreased during flood tide as waters from offshore enter the marsh. During tidal ebb, N concentrations increase as creek waters flow out of the marsh. The tidal study also indicated a net export of nitrogen from Cockle Cove Creek to Bucks Creek that is less than the watershed inputs by ~50% (Table IV-9). Moreover, it appears that in addition to removing nitrogen the marsh is transforming nitrogen from inorganic forms to organic forms. This can be best seen by comparing the average dissolved inorganic nitrogen (DIN) transport through the mid-marsh site (44.21 mg N sec^{-1}, Table IV-8) with the ebb tide transport of DIN (24.63 mg N sec^{-1}, calculated from Table IV-9). The export of particulate organic matter is seen in the net export, during the tidal study. These observations are consistent with other salt marshes of similar morphology (i.e. central tidal creek, New England marshes), both in their rates of nitrogen attenuation and in their nitrogen transformations.

IV.2.3. Confirmation of Watershed Nitrogen Discharge to Stage Harbor.

The third approach employed for evaluation of watershed nitrogen attenuation was to examine the nitrogen levels in the small or intermittent surface water discharges to the Town's embayments. Freshwater samples were collected from small surface inflows to Stage Harbor by the Chatham Water Quality Laboratory at the Water Quality Monitoring sites shown in Figure IV-6. Water samples were collected primarily during the summer months from flowing surface waters, 1999-2002. Surface flows that were tidal, brackish, and exhibited dilution of nitrogen by salt water required a correction of the data. The dilution by salt water was accounted for based upon the mean concentration of salt and total nitrogen within the water column of the adjacent embayment region. The embayment data was from the water quality monitoring database. This allowed for a site-specific correction and increased the accuracy of the analysis.

The surface water flows are fed by groundwater formed within the watersheds to the embayment's, and therefore, reflect the nitrogen levels in groundwater from a portion of an embayments watershed. These measured nitrogen levels can be compared to the nitrogen levels in freshwater discharging to the Town's embayments. This analysis is a diagnostic tool only.

Nitrogen levels in discharging waters in small streams can be lower than predicted from watershed analysis due to less loading to their contributing area, as opposed to the overall embayment watershed for which land-use nitrogen loading data is provided. The larger the watershed is to the stream, the more representative the comparison and results. Nitrogen levels can also be lower due to attenuation of nitrogen during transport.

The results of this screening indicated that the predicted and observed nitrogen concentrations for various watershed regions compared well for the Stage Harbor System. The results are relatively consistent for Oyster Pond, 1.7 mg N/L (predicted) versus 1.4 - 3.2 mg N/L observed. A similar result was observed from site CM-A in Stage Harbor where the predicted and observed total nitrogen values were 0.51 and 0.82 (s.d.=0.35; N=34) mg N/L, respectively. These results are consistent with the absence of major upland ponds and lakes within the watershed to the Stage Harbor System.

Figure IV-5. Sampling locations for nitrogen concentration and mass transport (boxes) and the tidal study (red line)

Table IV-7. Water quality parameters collected along the main channel of Cockle Cove Creek, summer 2000-2005. Values are means and standard error (s.e.) and number of samples (N). Transport of nitrogen and phosphate through the Cockle Cove Creek marshes, summer 2005. Station I.D.'s are shown in Figure 5.

Marsh Site	Sta i.d.[a]	Salinity (ppt)			Bioactive N (mgN/L)			Total N (mgLN/L)			Ortho-phosphate (mgN/L)		
		mean	s.e.	N	mean	s.e.	N	mean	s.e.	N	mean	s.e.	N
Fresh Headwater	CM-G/CC 1	0.2	0.02	70	1.514	0.053	61	1.822	0.061	61	0.009	0.001	73
Fresh Tidal	CM-J/CC 2	0.3	0.03	42	2.960	0.050	33	3.154	0.060	33	0.005	0.001	42
Main Channel													
mid-Salt Marsh	CM-F/CC 3	4.4	0.7	70	1.687	0.054	64	1.921	0.058	64	0.054	0.003	75
mid-lower SM	CM-T/4A,B4b	6.7	0.7	32	1.399	0.062	23	1.658	0.073	23	0.067	0.005	32
marsh inlet	CM-12/CC 5	21.9	0.6	95	0.540	0.029	79	0.787	0.034	79	0.038	0.003	95

a - Stations sampled by the Town of Chatham Water Quality Laboratory (Dr. R. Duncanson)/SMAST designated

Marsh Site	Sta i.d.[a]	NOx (mgN/L)			Ammonium (mgN/L)			Part. Org. N (mgN/L)			Dissolved Org N (mg/L)		
		mean	s.d.	N	mean	s.d.	N	mean	s.d.	N	mean	s.d.	N
Fresh Headwater	CM-G/CC 1	0.662	0.02	75	0.732	0.053	75	0.120	0.061	61	0.308	0.027	75
Fresh Tidal	CM-J/CC 2	1.791	0.03	42	1.104	0.050	42	0.066	0.064	33	0.193	0.044	42
Main Channel													
mid-Salt Marsh	CM-F/CC 3	1.201	0.7	75	0.321	0.054	75	0.165	0.059	64	0.234	0.021	75
mid-lower SM	CM-T/4A,B4b	0.875	0.7	32	0.314	0.062	32	0.210	0.073	23	0.259	0.032	32
marsh inlet	CM-12/CC 5	0.219	0.6	95	0.136	0.029	95	0.184	0.37	79	0.247	0.020	95

a - Stations sampled by the Town of Chatham Water Quality Laboratory (Dr. R. Duncanson)/SMAST designated

Table IV-8. Transport of nitrogen and phosphate through the Cockle Cove Creek marshes through the warmer months of 2005. Values are averages of measured watershed flux through the marsh, based upon bi monthly ebb tide sampling. All values are presented as daily transport (mg/sec) to allow comparison to the MEP Watershed Model (updated April 2006). Station I.D.'s are shown in Figure 5. Data was collected by the Town of Chatham Water Quality Laboratory and SMAST staff.

Marsh Site	I.D.	NOx	NH_4	PON	DON	BioActive N	Total N	PO_4
Freshwater: Headwater Stream								
Fresh Headwater	CC 1	3.13	4.98	0.81	1.85	8.91	10.07	0.004
Fresh Tidal	CC 2	18.75	12.15	0.69	1.16	31.60	32.99	0.004
Main Channel								
mid-Salt Marsh	CC 3	35.65	8.56	2.78	3.82	46.99	50.35	0.153
side channel to CC-3	CC 4	1.04	0.69	1.74	1.39	3.47	4.40	0.028
mid-lower SM	CC 4a	29.86	8.45	4.40	4.51	42.71	46.41	0.205
marsh inlet	CC 5	17.36	9.38	14.00	13.43	**40.86**	**52.43**	0.150
Watershed Land-Use Model N Loading								
Non-WWTF N Load	--	--	--	--	--	--	59.26	--
WWTF N Load	--	--	--	--	--	--	37.15	--
Total N Load	--	--	--	--	--	--	**96.41**	--
System N Attenuation [b]							**46%**	

a - Stations sampled by Coastal Systems Program, SMAST on 11 sampling dates during warmer months
b - Attenuation calculated between Watershed N Load and Station CC-5.
Note: Nitrogen loads measured within the stream/creek reflect the balance between uptake and new inputs from the watershed.

Table IV-9. Tidal import/export of nitrogen and chlorophyll a pigments collected near Cockle Cove Creek inlet to Bucks Creek, over a complete tidal cycle on August 3, 2005. Values are total mass flux (kg/tide phase). There was a net export from the Cockle Cove Marshes and associated watershed to Bucks Creek. Sampling was from low tide to low tide (with balance of the salt mass), location is shown in Figure 5. Comparison of the measured net export of nitrogen from the marsh and the nitrogen input from the watershed, from the MEP watershed model (updated April 2006), indicates significant summer attenuation of the nitrogen, 44%.

	NOx	NH_4	PON	DON	BioActive N	Total N	Chl a
Tide Phase							
FLOOD	0.015	0.034	0.870	0.989	0.920	1.698	0.048
EBB	0.233	0.432	1.254	1.917	1.919	3.836	0.047
Ebb-Flood							
Net Export	0.218	0.398	0.384	0.928	1.001	2.138	-0.001
Watershed Land-Use Model N Loading							
Total N Load, per 2 tidal cycles	--	--	--	--	--	**8.60**[c]	--
System N Attenuation [b]						**50%**	

a - Stations sampled by Coastal Systems Program, SMAST on 11 sampling dates during warmer months
b - Attenuation calculated between Watershed N Load and Station CC-5.
c - the daily watershed N loading was adjusted to 2 tidal cycles to compare with the measured tidal flux data.
Note: Nitrogen loads measured within the stream/creek reflect the balance between uptake and new inputs from the watershed.

Figure IV-6. Map of freshwater discharge (blue squares) and estuarine (red circles) water quality monitoring stations within the Town of Chatham's southern 3 estuaries.

IV.3 BENTHIC REGENERATION OF NITROGEN IN BOTTOM SEDIMENTS

The overall objective of the Benthic Nutrient Flux Task was to quantify the summertime exchange of nitrogen, between the sediments and overlying waters within each of the 3 southern embayments in Chatham. The mass exchange of nitrogen between watercolumn and sediments is a fundamental factor in controlling nitrogen levels within coastal waters. These fluxes and their associated biogeochemical pools relate directly to carbon, nutrient and oxygen dynamics and the nutrient related ecological health of these shallow marine ecosystems. In addition, these data are required for the proper modeling of nitrogen in shallow aquatic systems, both fresh and salt water.

IV.3.1 Sediment-Watercolumn Exchange of Nitrogen

As stated in above sections, nitrogen loading and resulting levels within coastal embayments are the critical factors controlling the nutrient related ecological health and habitat quality within a system. Nitrogen enters the embayments of Chatham predominantly in highly bioavailable forms from the surrounding upland watershed and in flooding tidal waters. If all of the nitrogen remained within the watercolumn (once it entered), then predicting watercolumn nitrogen levels would be simply a matter of determining the watershed loads, dispersion, and hydrodynamic flushing. However, as nitrogen enters the embayments from the surrounding watersheds it is predominantly in the bioavailable form nitrate. This nitrate and other bioavailable forms are rapidly taken up by phytoplankton for growth, i.e. it is converted from dissolved forms into phytoplankton “particles”. Most of these “particles” remain in the watercolumn for sufficient time to be flushed out to a downgradient larger waterbody (like Pleasant Bay or Nantucket Sound). However, some of these phytoplankton particles are grazed by zooplankton or filtered from the water by shellfish and other benthic animals. Also, in longer residence time systems (greater than 8 days) these nitrogen rich particles may die and settle to the bottom. In both cases (grazing or senescence), a fraction of the phytoplankton with their associated nitrogen “load” become incorporated into the surficial sediments of the bays.

In general the fraction of the phytoplankton population which enters the surficial sediments of a shallow embayment: (1) increases with decreased hydrodynamic flushing, (2) increases in low velocity settings, (3) increases within small basins (e.g. Mill Pond, Taylors Pond, etc). To some extent, the settling characteristics can be evaluated by observation of the grain-size and organic content of sediments within an estuary.

Once organic particles become incorporated into surface sediments they are decomposed by the natural animal and microbial community. This process can take place both in oxic (oxygenated) or anoxic (no oxygen present) conditions. It is through the decay of the organic matter with its nitrogen content that bioavailable nitrogen is returned to the embayment watercolumn for another round of uptake by phytoplankton. This recycled nitrogen adds directly to the eutrophication of the estuarine waters in the same fashion as watershed inputs. In some systems that we have investigated, recycled nitrogen can account for about one-third to one-half of the nitrogen supply to phytoplankton blooms during the warmer summer months. It is during these warmer months that estuarine waters are most sensitive to nitrogen loadings. Failure to account for this recycled nitrogen generally results in significant errors in determination of threshold nitrogen loadings. In addition, since the sites of recycling can be different from the sites of nitrogen entry from the watershed, both recycling and watershed data are needed to determine the best approaches for nitrogen mitigation.

IV.3.2 Method for determining sediment-watercolumn nitrogen exchange

For the 3 Chatham embayments in order to determine the contribution of sediment regeneration to nutrient levels during the most sensitive summer interval (July-August), sediment samples were collected and incubated under *in situ* conditions. Sediment samples were collected in late July 2000 (Figure IV-7). Measurements of total dissolved nitrogen, nitrate + nitrite, ammonium and ortho-phosphate were made in time-series on each incubated core sample. As part of a separate research investigation, the rate of oxygen uptake was also determined and measurements of sediment bulk density, organic nitrogen, and carbon content were made.

Rate measurements of nutrient release (and oxygen uptake) were made using undisturbed sediment cores incubated for 24-36 hours in temperature controlled baths. Sediment cores (15 cm inside diameter) were collected by SCUBA divers and cores transported by a small boat. Cores are maintained from collection through incubation at *in situ* temperatures. Bottom water was collected and filtered from each core site to replace the headspace water of the flux cores prior to incubation. Cores were collected from the 3 embayments as follows: Stage Harbor System - 18 cores, Taylors Pond/Mill Creek - 5 cores, Sulphur Springs/Cockle Cove/Bucks Creek - 7 cores. Sampling was distributed throughout each embayment system and the core results combined for calculating the net nitrogen regeneration rates for the water quality modeling effort.

Sediment-watercolumn exchange follows the methods of Jorgensen (1977), Klump and Martens (1983), and Howes *et al.* (1995) for nutrients and metabolism. Upon return to the field laboratory (Chatham Water Quality Laboratory Annex) the cores were transferred to pre-equilibrated temperature baths. The headspace water overlying the sediment was replaced, magnetic stirrers emplaced, and the headspace enclosed. Oxygen consumption was determined in time-course incubations up to 24 hours. Periodic 60 ml water samples were withdrawn (volume replaced with filtered water), filtered into acid leached polyethylene bottles and held on ice for nutrient analysis. Ammonium (Scheiner 1976) and ortho-phosphate (Murphy and Reilly 1962) assays were conducted within 24 hours and the remaining sample frozen (-20^{o}C) for assay of nitrate + nitrite (Cd reduction: Lachat Autoanalysis), and DON (D'Elia *et al.* 1977). Rates were determined from linear regression of analyte concentrations through time.

Analyses were performed by the Coastal Systems Analytical Facility at the School for Marine Science and Technology (SMAST) at the University of Massachusetts in New Bedford, MA. The laboratory follows standard methods for salt water analysis and sediment geochemistry.

IV.3.3 Determination of Summer Nitrogen Regeneration from Sediments

Watercolumn nitrogen levels are the balance of inputs from direct sources (land, rain etc), losses (denitrification, burial), regeneration (watercolumn and benthic), and uptake (e.g. photosynthesis). As stated above, during the warmer summer months the sediments of shallow embayments typically act as a net source of nitrogen to the overlying waters and help to stimulate eutrophication in organic rich systems. However, some sediments may be net sinks for nitrogen and some may be in “balance” (organic N particle settling = nitrogen release). Sediments may also take up dissolved nitrate directly from the watercolumn and convert it to dinitrogen gas, hence effectively removing it from the ecosystem. This process can be very effective in removing nitrogen loads, particularly in salt marshes and is termed “denitrification”.

In addition to nitrogen cycling, there are ecological consequences to habitat quality of organic matter settling and mineralization within sediments which relate primarily to sediment and watercolumn oxygen status. However, for the modeling of nitrogen within an embayment it is the relative balance of nitrogen input from watercolumn to sediment versus regeneration which is critical. It is the net balance of nitrogen fluxes between water column and sediments during the modeling period that must be quantified. For example, a net input to the sediments represents an effective lowering of the nitrogen loading to down-gradient systems and net output from the sediments represents an additional load.

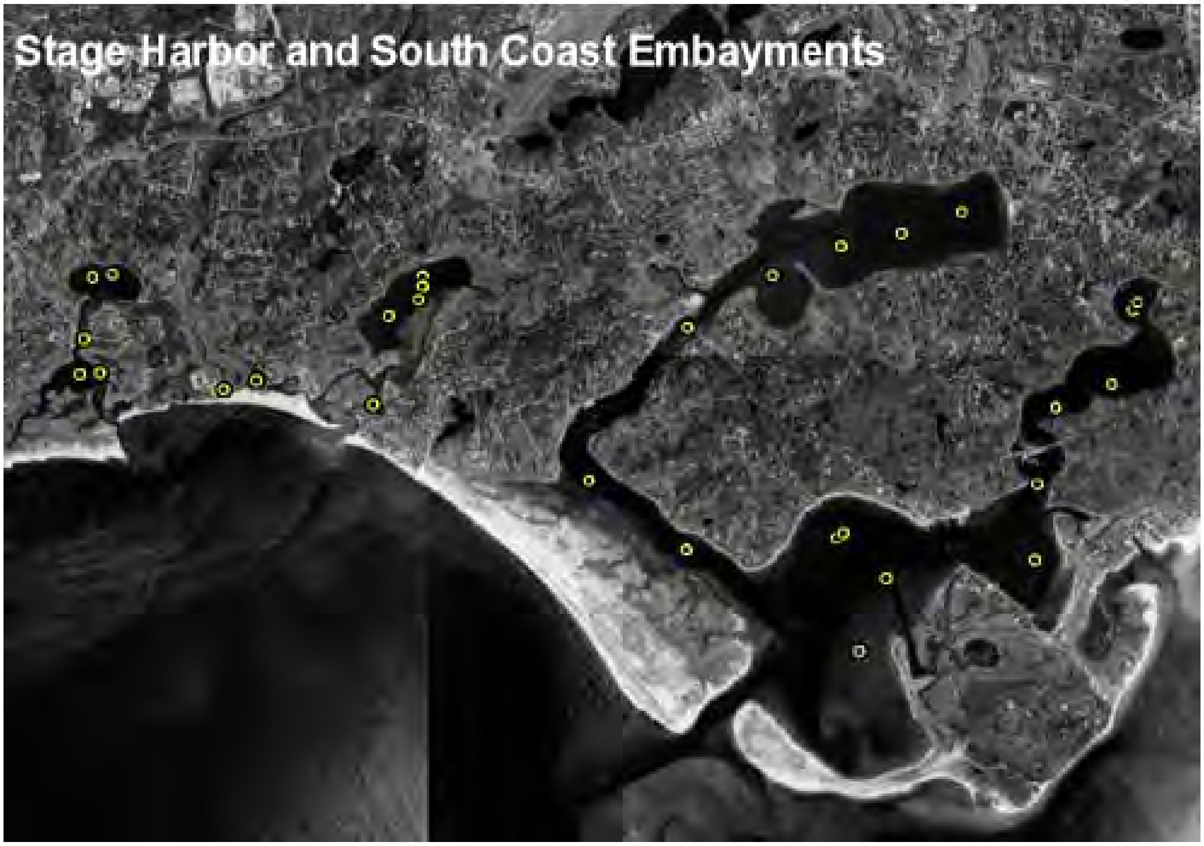


Figure IV-7. Chatham shoreline with locations of sediment core sampling stations shown as red filled circles. Some locations are sites of more than one sample, with sampling in July 2000.

The relative balance of nitrogen fluxes ("in" versus "out" of sediments) is dominated by the rate of particulate settling (in), the rate of denitrification of nitrate from overlying water (in), and regeneration (out). The rate of denitrification is controlled by the organic levels within the sediment (oxic/anoxic) and the concentration of nitrate in the overlying water. Organic rich sediment systems with high overlying nitrate frequently show large net nitrogen uptake throughout the summer months, even though organic nitrogen is being mineralized and released to the overlying water as well. The rate of nitrate uptake, simply dominates the overall sediment nitrogen cycle.

In order to model the nitrogen distribution within an embayment it is important to be able to account for the net nitrogen flux from the sediments within each part of each system. This requires that an estimate of the particulate input and nitrate uptake be obtained for comparison to the rate of nitrogen release. Only sediments with a net release of nitrogen contribute a true additional nitrogen load to the overlying waters, while those with a net input to the sediments serve as an "in embayment" attenuation mechanism for nitrogen.

Overall, coastal sediments are not overlain by nitrate rich waters and the major nitrogen input is via phytoplankton grazing or direct settling. In these systems, on an annual basis, the amount of nitrogen input to sediments is generally higher than the amount of nitrogen release. This net sink results from the burial of reworked refractory organic compounds, sorption of inorganic nitrogen and some denitrification of produced inorganic nitrogen before it can "escape" to the overlying waters. However, this net sink evaluation of coastal sediments is based upon annual fluxes. If seasonality is taken into account, it is clear that sediments undergo periods of net input and net output. The net output is generally during warmer periods and the net input is during colder periods. The result can be an accumulation of nitrogen within late fall, winter, and early spring and a net release during summer. The conceptual model of this seasonality has the sediments acting as a battery with the flux balance controlled by temperature (Figure IV-8).

Unfortunately, the tendency for net release of nitrogen during warmer periods, coincides with the periods of lowest nutrient related water quality within temperate embayments. This sediment nitrogen release is in part responsible for poor summer nutrient related health. Other major factors causing the seasonal water quality decline are the lower solubility of oxygen during summer, the higher oxygen demand by marine communities, and environmental conditions supportive of high phytoplankton growth rates.

In order to determine the net nitrogen flux between watercolumn and sediments, all of the above factors were taken into account. The net input or release of nitrogen within a specific embayment was determined based upon the measured ammonium release, measured nitrate uptake or release, and estimate of particulate nitrogen input. Dissolved organic nitrogen fluxes were not used in this analysis, since they were highly variable and generally showed a net balance within the bounds of the method.

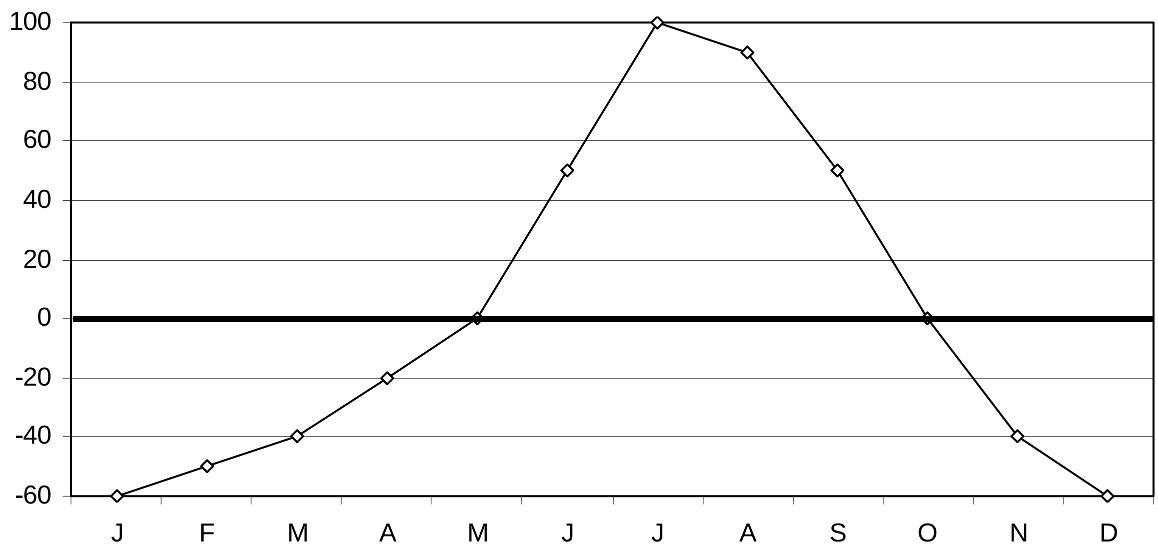


Figure IV-8. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

In order to obtain the net nitrogen balance of each embayments sediments, 30 cores were collected throughout the 3 southern embayments of Chatham (Figure IV-7). The distribution of cores was established to cover gradients in sediment type, flow field and phytoplankton density. Multiple cores were typically collected per sub-embayment and the results were averaged within an embayment for parameterizing the water quality model. For each core the nitrogen flux from the core incubations (described in the section above) were combined with measurements of the sediment organic carbon and nitrogen content and bulk density and an analysis of the sites tidal flow velocities. The maximum bottom water flow velocity at each coring site was determined from the calibrated and validated hydrodynamic model. The rate of organic nitrogen in particle settling was based upon measured particulate carbon and nitrogen concentrations measured during the appropriate summer, 2000 or 2001, (as well as overall 2000-2005 averages), by the Chatham Water Watchers and Chatham Water Quality Monitoring Program. These data were then used to determine the nitrogen balance of a sediment system.

The magnitude of the settling of particulate organic carbon and nitrogen into the sediments was accomplished by determining the average depth of water within each sediment site and the average summer particulate carbon and nitrogen concentration within the overlying water (from the monitoring program database). Two levels of settling were used. If the bulk density of the sediments indicated a fine grained substrate and data indicated a high carbon content and low velocities, then a water column particle residence time of 8 days was used (based upon phytoplankton studies of poorly flushed basins). If the sediments indicated a coarse grained sediments and low organic content and high velocities, then half this settling rate was used.

Adjusting the measured sediment releases was essential in order not to over-estimate the sediment nitrogen source and to account for those sediment areas which are net nitrogen sinks for the aquatic system. These results can be validated by examining the relative fraction of the sediment carbon turnover (total sediment metabolism) which would be accounted for by daily particulate carbon settling. This analysis indicates that sediment metabolism in the highly organic rich sediments of the wetlands and depositional basins was driven primarily by stored organic matter (ca. 90%). Also, in the more open lower portions of the larger embayments, storage appears to be low and a large proportion of the daily carbon requirement in summer is met by particle settling (approximately 33% to 67%). This range of values and their distribution is consistent with ecological theory and field data from shallow embayments (Figure IV-9). As depicted in figure IV-9, with the exception of Frost Fish Creek (not in this report), sediment nitrogen to organic carbon ratios indicate that phytoplankton is the prime source of carbon deposited in these sediments.

Net nitrogen release or uptake from the sediments of the 3 embayment systems used in the water quality modeling effort (Section VI) are presented in Table IV-10. The variation for each embayment system encompasses the spatial variation within each sub-basin, due to organic matter deposition, water depth, sediment type, etc. Basins with small release rates (near zero) will have proportionally larger variation, however, since the release is low this variation is not generally ecologically significant. The critical way to view the data relates to the inter-basin differences, which typically indicate that the upper basins have the largest nitrogen release rates and the narrower flow regions have lowest release rates (e.g. Oyster Pond versus Oyster River, Mill Pond/Little Mill Pond versus Mitchell River).

Table IV-10. Rates of net nitrogen return from sediments to overlying waters based on sub-embayment area coverage and core flux measurements. N is the number of cores.

Sub-embayment	N	Net N Efflux (mg m^{-2} d^{-1})	Standard Error
Stage Harbor			
Oyster Pond	4	38.8	12.1
Oyster River	3	2.9	2.9
Stage Harbor	4	2.9	1.0
Mitchell River	--	9.4	--
Mill Pond	3	50.6	16.0
Little Mill Pond	2	40.9	11.7
Sulphur Springs			
Sulphur Springs		-19.9	3.6
Bucks Creek		41.2	--
Cockle Cove Creek		-17.5	0.5
Taylors Pond			
Mill Creek		-0.6	0.2
Taylors Pond		22.8	17.8

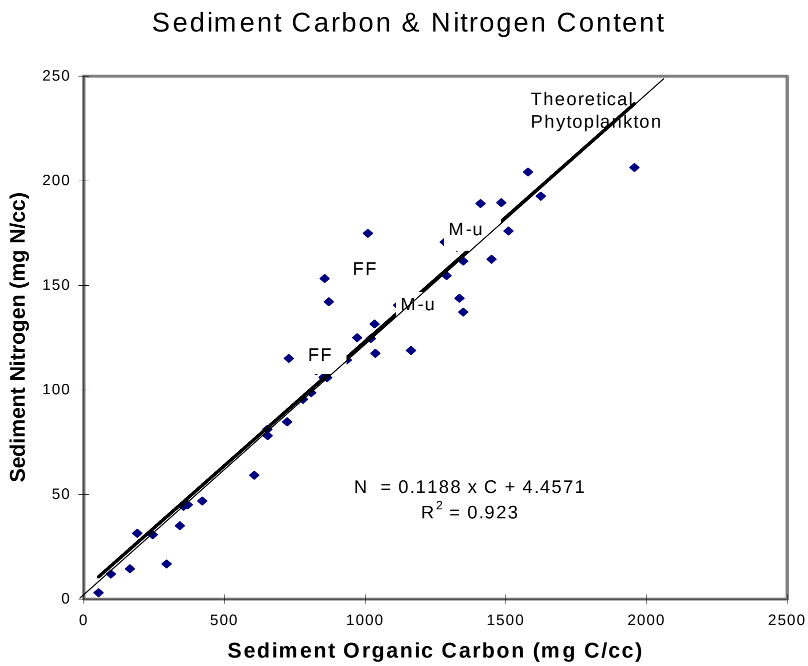


Figure IV-9 Sediment Carbon vs. Sediment Nitrogen content for core samples taken from Chatham sub-embayments

VI. WATER QUALITY MODELING

VI.1 DATA SOURCES FOR THE MODEL

Several different data types and calculations are required to support the water quality modeling effort. These include the output from the hydrodynamics model, calculations of external nitrogen loads from the watersheds, measurements of internal nitrogen loads from the sediment (benthic flux), and measurements of nitrogen in the water column.

VI.1.1 Hydrodynamics and Tidal Flushing in the Embayments

Extensive field measurements and hydrodynamic modeling of the embayments were an essential preparatory step to the development of the water quality model. The result of this work, among other things, was a set of five files of calibrated model output representing the transport of water within each of the three embayment systems of Chatham's south coast. Files of node locations and node connectivity for the RMA-2V model grids were transferred to the RMA-4 water quality model; therefore, the computational grid for the hydrodynamic model also was the computational grid for the water quality model. The period of hydrodynamic output for the water quality model calibration was a 14-tidal cycle period in summer 2000 that included both the neap and spring cycles.

VI.1.2 Nitrogen Loading to the Embayments

Three primary nitrogen loads to the embayments are recognized in this modeling study: external loads from the watersheds, nitrogen load from direct rainfall on the embayment surface, and internal loads from the sediments. Additionally, there is a fourth load to the embayments, consisting of the background concentrations of total nitrogen in the waters entering from Nantucket Sound. This load is represented as a constant concentration along the seaward boundary of each model grid.

VI.1.3 Measured Nitrogen Concentrations in the Embayments

In order to create a model that realistically simulates the total nitrogen concentrations in a system in response to the existing flushing conditions and loadings, it is necessary to calibrate the model to actual measurements of water column nitrogen concentrations. The Town of Chatham Water Quality Laboratory, in conjunction with the Chatham Water Watchers (citizen volunteers), initiated a water quality monitoring program in the Stage Harbor system in the fall of 1998, and continued it through the summer of 1999. In 2000, sampling stations were added in the Sulphur Springs and Taylors Pond systems (Duncanson, 2000). The sampling has continued and data through 2005 were used in the present analysis. The goals of this program were to monitor existing water quality conditions, to provide data on the extent to which water quality was meeting goals or criteria, to compare conditions in the different embayments and their watersheds for targeting remedial actions, to help focus future studies on areas perceived as degraded, and to provide a long term data set for monitoring the success of remediation activities (Duncanson, 2000). The data were reviewed and did meet quality control requirements under the MEP QAPP. The monitoring data were overseen by the Chatham Water Quality Laboratory and did have an approved QAPP. The refined and approved data for each system used in the water quality modeling effort are presented in Table VI-1. The multi-year averages present the "best" comparison to the water quality model output, since factors of tide, temperature and rainfall may exert short-term influences on the individual sampling dates and even cause inter-annual differences. Three years of baseline field data is the minimum required to provide a baseline for MEP analysis.

Table VI-1. Measured and modeled Nitrogen concentrations for Stage Harbor, Sulphur Springs, and Taylors Pond, used in the model calibration plots of Figures VI-6 (Stage Harbor total N),VI-7 (Sulphur Springs), and VI-8 (Taylors Pond). All concentrations are given in mg/L N. "Data mean" values are calculated as the average of all measurments.

System	Embayment	1999 mean	2000 mean	2001 mean	2002 mean	2003 mean	2004 mean	2005 mean	data mean	s.d.	N	model min	model average	model max
Stage Harbor*	Oyster Pond	0.597	0.786	0.708	0.604	0.770	0.671	0.761	0.735	0.227	45	0.708	0.721	0.714
	Lower Oyster Pond	-	-	0.552	0.498	0.482	0.580	0.447	0.513	0.135	27	0.372	0.652	0.534
	Oyster River	0.451	0.457	0.386	0.536	0.458	0.609	0.491	0.489	0.121	39	0.287	0.546	0.367
	Stage Harbor	-	-	-	-	-	-	0.385	0.385	0.062	29	0.288	0.415	0.336
	Upper Stage Harbor	0.418	0.457	0.503	0.548	0.500	0.500	0.467	0.503	0.136	103	0.381	0.425	0.403
	Mitchell River	-	-	0.429	0.487	0.477	0.494	0.400	0.459	0.087	29	0.409	0.463	0.435
	Mill Pond	0.471	0.503	0.418	0.507	0.520	0.390	0.553	0.485	0.123	96	0.458	0.474	0.466
	Little Mill Pond	0.792	0.690	0.742	0.741	0.805	0.764	0.554	0.736	0.232	97	0.653	0.675	0.666
Sulphur Springs	Mid Cockle Cove Cr.	-	1.492	2.043	1.613	2.115	1.499	1.901	1.857	0.531	36	0.606	1.373	2.482
	Cockle C. Cr. mouth	-	0.890	0.687	0.636	0.973	0.620	0.536	0.730	0.242	38	0.275	0.410	0.813
	Bucks Creek	-	0.401	0.479	0.576	0.561	0.573	0.621	0.516	0.149	38	0.282	0.347	0.684
	Sulphur Springs	-	0.360	0.453	0.584	0.623	0.643	0.768	0.584	0.179	39	0.270	0.452	0.906
Taylors Pond	Mill Creek	-	0.491	0.508	0.530	0.546	0.484	0.534	0.516	0.124	75	0.284	0.329	0.630
	Taylors Pond	-	0.509	0.487	0.530	0.575	0.568	0.528	0.525	0.099	37	0.414	0.455	0.502

VI.2 MODEL DESCRIPTION AND APPLICATION

A two-dimensional finite element water quality model, RMA-4 (King, 1990), was employed to study the effects of nitrogen loading in the three south shore Chatham embayment systems. The RMA-4 model has the capability for the simulation of advection-diffusion processes in aquatic environments. It is the constituent transport model counterpart of the RMA-2 hydrodynamic model used to simulate the fluid dynamics of the Chatham embayments. Like RMA-2 numerical code, RMA-4 is a two-dimensional, depth averaged finite element model capable of simulating time-dependent constituent transport. The RMA-4 model was developed with support from the US Army Corps of Engineers (USACE) Waterways Experiment Station (WES), and is widely accepted and tested. Applied Coastal staff have utilized this model in water quality studies of other Cape Cod embayments, including West Falmouth Harbor and the Falmouth "finger" ponds (Ramsey et al., 2000).

The overall approach involves modeling total nitrogen as a non-conservative constituent, where bottom sediments act as a source or sink of nitrogen, based on local biochemical characteristics. This modeling represents summertime conditions, when algal growth is at its maximum. Total nitrogen modeling is based upon various data collection efforts and analyses presented in previous sections of this report. Nitrogen loading information was derived from the Cape Cod Commission watershed loading analysis (based on the revised USGS watersheds), as well as the measured bottom sediment nitrogen fluxes. Water column nitrogen measurements by the Chatham Water Watchers were utilized as model boundaries and as calibration data. Hydrodynamic model output (discussed in Section V) provided the remaining information (tides, currents, and bathymetry) needed to parameterize the water quality model.

VI.2.1 Model Formulation

The formulation of the model is for two-dimensional depth-averaged systems in which concentration in the vertical direction is assumed uniform. The governing equation of the RMA-4 constituent model can be most simply expressed as a form of the transport equation, in two dimensions:

$$\left(\frac{\partial c}{\partial t}+u\frac{\partial c}{\partial x}+v\frac{\partial c}{\partial y}\right)=\left(\frac{\partial}{\partial x}D_x\frac{\partial c}{\partial x}+\frac{\partial}{\partial y}D_y\frac{\partial c}{\partial y}+\sigma\right)$$

where c in the water quality constituent concentration; t is time; u and v are the velocities in the x and y directions, respectively; D_x and D_y are the model dispersion coefficients in the x and y directions; and σ is the constituent source/sink term. Since the model utilizes input from the RMA-2 model, a similar implicit solution technique is employed for the RMA-4 model.

The model is therefore used to compute spatially and temporally varying concentrations c of the modeled constituent (i.e., total nitrogen), based on model inputs of 1) water depth and velocity computed using the RMA-2 hydrodynamic model; 2) mass loading input of the modeled constituent; and 3) user selected values of the model dispersion coefficients. The dispersion coefficients used in the sub-embayments of each of the five modeled systems were developed during the calibration process. During the calibration procedure, the dispersion coefficients were incrementally changed until model concentration outputs matched measured data.

The depth-averaged assumption is justified since vertical mixing by wind and tidal processes prevent significant stratification in these systems, even in the relatively deep kettle sub-embayments that are part of some of the Chatham embayments. This lack of stratification is evident in the temperature and salinity profiles of three such estuarine kettle ponds in Chatham, shown in Figure VI-1 and VI-2.

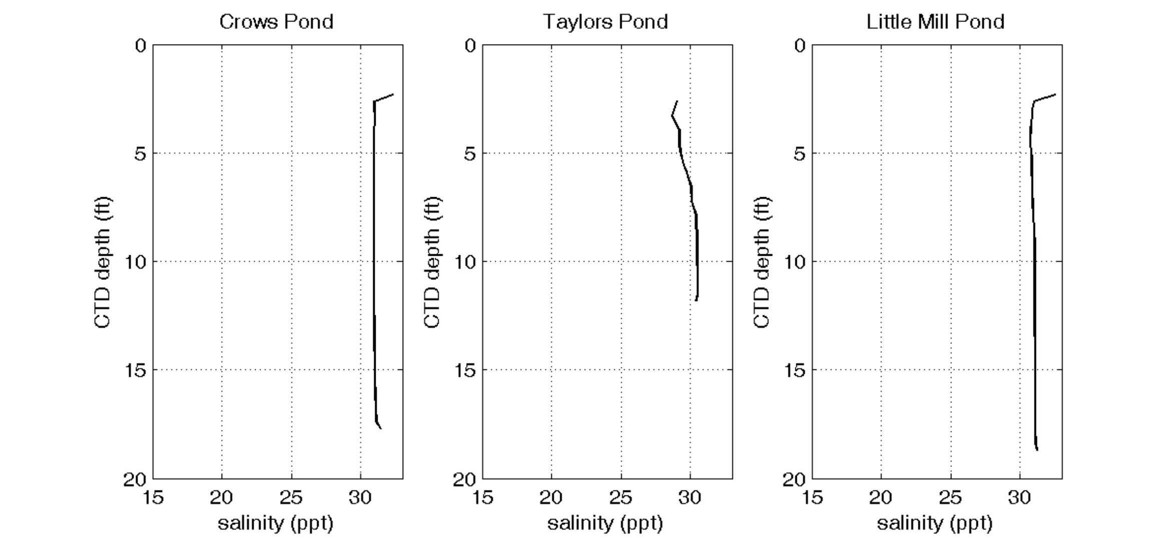


Figure VI-1. CTD cast salinity profiles for Crows Pond (Bassing Harbor), Taylors Pond, and Little Mill Pond (Stage Harbor). Cast data were recorded at 0.66 ft increments (0.2 m), during July 18 (Crows Pond), July 19 (Taylors Pond), and July 20 (Little Mill Pond) of 2000.

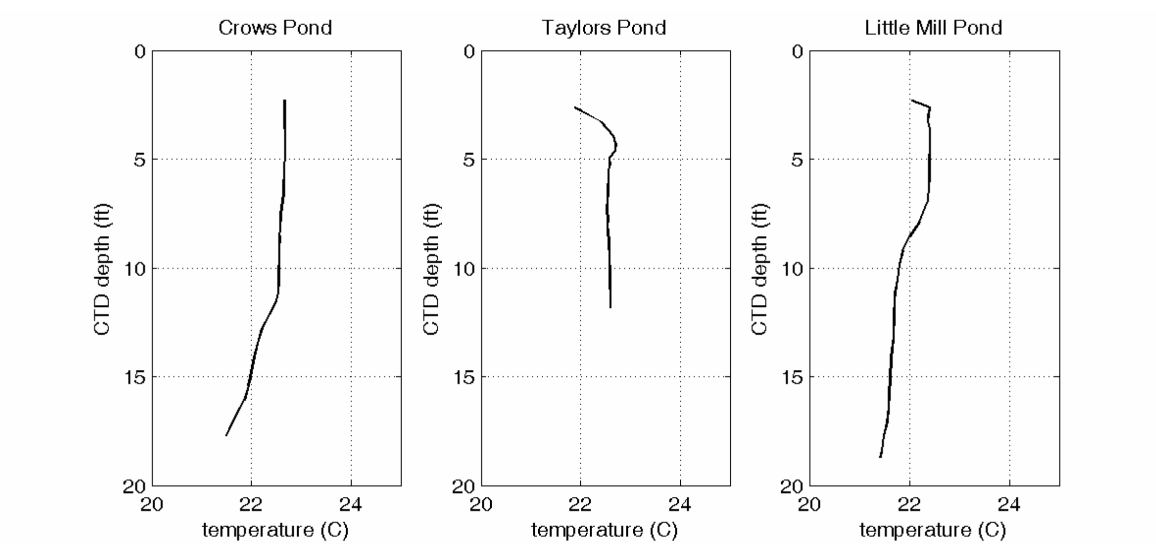


Figure VI-2. CTD cast temperature profiles for Crows Pond (Bassing Harbor), Taylors Pond, and Little Mill Pond (Stage Harbor). Cast data were recorded at 0.66 ft increments (0.2 m), during July 18 (Crows Pond), July 19 (Taylors Pond), and July 20 (Little Mill Pond) of 2000.

RMA-4 model can be utilized to predict both spatial and temporal variations in total. At each time step the model computes constituent concentrations over the entire finite element grid and utilizes a continuity of mass equation to check these results. Similar to the hydrodynamic model, the water quality model evaluates model parameters at every element at 12-minute time intervals throughout the grid system. For this application, the RMA-4 model was used to predict tidally averaged total nitrogen concentrations throughout the three south shore estuarine systems in Chatham.

VI.2.2 Water Quality Model Setup

Required inputs to the RMA-4 model include a computational mesh, computed water elevations and velocities at all nodes of the mesh, constituent mass loading, and spatially varying values of the dispersion coefficient. Because the RMA-4 model is part of a suite of integrated computer models, the finite-element meshes and the resulting hydrodynamic simulations previously developed for the three Chatham south coast embayment systems also were used for the water quality constituent modeling portion of this study.

For each model, an initial total N concentration equal to the concentration at the open boundary was applied to the entire model domain. The model was then run for a simulated month-long (30 day) spin-up period. At the end of the spin-up period, the model was run for an additional 5 tidal-day (124 hour) period. Model results were recorded only after the initial spin-up. The time step used for the water quality computations was 12 minutes, which corresponds to the time step of the hydrodynamics input to each of the three south coast systems.

VI.2.3 Boundary Condition Specification

Mass loading of nitrogen into each model included: 1) sources developed from the results of the watershed analysis, 2) estimates of direct atmospheric deposition, and 3) summer benthic regeneration. Nitrogen loads from each separate sub-embayment watershed were distributed across the sub-embayment. For example, the loads from the Little Mill Pond watershed were evenly distributed at the grid cells that formed the perimeter of the pond. Similarly, benthic flux loads were distributed among grid cells in the central portions of each sub-embayment.

The loadings used to model present conditions in the south coast embayments are given in Table VI-2. Watershed and depositional loads were taken from the results of the analysis of Section IV. Summertime benthic flux loads were computed based on the analysis of sediment cores described in Section IV. The area rate (g/sec/m^2) of nitrogen flux from that analysis was applied to the surface area coverage computed for each sub-embayment (excluding marsh coverages, when present), resulting in a total flux for each embayment (as listed in Table VI-2).

In addition to mass loading boundary conditions set within the model domain, concentrations along the model open boundaries were specified. The model uses concentrations at the open boundary during the flooding tide periods of the model simulations. Constituent concentrations of the incoming water are set at the value designated for the open boundary. For the south coast embayments (Taylors Pond, Sulphur Springs and Stage Harbor), the boundary concentration in Nantucket Sound was set at 0.285 mg/L, based on Chatham Water Watchers data from the Sound (station CM-7). These total nitrogen concentration represent long-term average summer concentrations found within Nantucket Sound.

Table VI-2. Sub-embayment loads used for total nitrogen modeling of the Stage Harbor and South Coastal embayment systems, with total watershed N loads, atmospheric N loads, and benthic flux. These loads represent present loading conditions for the listed sub-embayments.

sub-embayment	watershed load (kg/day)	atmospheric deposition (kg/day)	benthic flux (kg/day)
Stage Harbor			
Oyster Pond	10.041	1.784	22.568
Oyster River	9.400	1.055	0.968
Stage Harbor	2.000	3.244	4.061
Mitchell River	2.586	0.882	4.032
Mill Pond	3.600	0.627	3.521
Little Mill Pond	1.255	0.121	1.792
Sulphur Springs			
Sulphur Springs	9.093	0.378	-3.756
Bucks Creek	3.362	0.132	2.910
Cockle Cove Creek	8.427	0.060	-0.578
Waste Water TF	3.205	-	-
Taylors Pond			
Mill Creek	4.559	0.167	-0.061
Taylors Pond	6.219	0.186	1.424

VI.2.4 Model Calibration

Calibration of each of the three Chatham south coast embayment systems proceeded by changing model dispersion coefficients so that model output of nitrogen concentrations matched measured data. Generally, several model runs of each system were required to match the water column measurements. Dispersion coefficient (*E*) values were varied through the modeled systems by setting different values of *E* for each grid material type, as designated in Section V. Observed values of *E* (Fischer, *et al.*, 1979) vary between order 10 and order 1000 m^2/sec for riverine estuary systems characterized by relatively wide channels (compared to channel depth) with moderate currents. Coefficients in this range are appropriate for embayments with these characteristics, such as Oyster River (Stage Harbor). Generally, the embayments of Chatham are small compared to the riverine estuary systems evaluated by Fischer, *et al.*, (1979); therefore the values of *E* also are relatively lower for Chatham. Smaller values of *E* occur in deeper and narrower, relatively quiescent sub-embayments, such as Taylors Pond. Observed values of *E* in these calmer areas typically range between order 10 and order 0.001 m^2/sec (USACE, 2001). The final values of *E* used in each sub-embayment of the modeled systems are presented in Table VI-3. These values were used to develop the "best-fit" total nitrogen model calibration. For the case of TN modeling, "best fit" can be defined as minimizing the error between the model and data at all sampling locations, utilizing reasonable ranges of dispersion coefficients within each sub-embayment.

Table VI-3. Values of longitudinal dispersion coefficient, E, used in calibrated RMA4 model runs of salinity and nitrogen concentration for the south coastal embayments.

Embayment Division	E m²/sec
Stage Harbor System	
Oyster Pond - upper	1.00
Oyster Pond - lower	2.00
Oyster River	25.0
Little Mill Pond	0.01
Mill Pond	1.00
Mitchell River	5.00
Stage Harbor - upper	4.00
Stage Harbor – main basin	2.00
Stage Harbor - inlet	5.00
Sulphur Springs System	
Cockle Cove Creek – marsh	0.60
Cockle Cove Creek – channel	0.60
Sulphur Springs – basin	0.01
Sulphur Springs – marsh	0.05
Bucks Creek – marsh	0.50
Bucks Creek – channel	0.05
Bucks Creek – inlet to Nantucket Sound	0.50
Taylors Pond System	
Taylors Pond – basin	0.15
Mill Creek – upper channel	0.20
Mill Creek – lower channel	0.25
Mill Creek – marsh	0.05
Mill Creek – inlet to Nantucket Sound	1.00

Comparisons between model output and measured nitrogen concentrations are shown in Figures VI-3 through VI-8 for each of the three modeled embayment systems. In Figures VI-3, VI-5 and VI-7, the mean TN measurement and standard deviation of all the data at each individual station are plotted against the modeled maximum, mean, and minimum concentrations output from the model at locations which correspond to the Water Watcher stations. In Figures VI-4, VI-6 and VI-8, additional comparisons are presented between measured TN data and the model target concentration.

Because the water samples are taken during ebbing tides, calibration targets in each sub-embayment were set such that the means of the measured data would fall within the range between the modeled maximum and modeled mean concentration, for stations where there is a wide range of modeled concentrations.

Calibrated model output is shown in Figures VI-9 through VI-11 for Stage Harbor, Sulphur Springs/Cockle Cove Creek and Taylors Pond/Mill. In these figures, color contours indicate nitrogen concentrations throughout the model domain. The output in these figures show average total nitrogen concentrations, computed using the full 5-tidal-day model simulation output period. The range of the color scale used to indicate total N concentrations is the same for all five of these figures, to show conditions that exist in each system relative to the complete range of nitrogen concentrations observed in Chatham's embayments.

In addition to the model calibration based on nitrogen loading and water column measurements, numerical water quality model performance is typically verified by modeling salinity. This additional modeling step was not feasible in the modeled Chatham embayment systems because measured salinity data show only a slight gradient through to the uppermost reaches of each system (<1 ppt). The only exception in this case is Cockle Cove Creek, which has brackish to fresh in its upper reaches. Salinity modeling was not performed for these systems, however, because the existing salinity data does not provide enough information for adequate model verification.

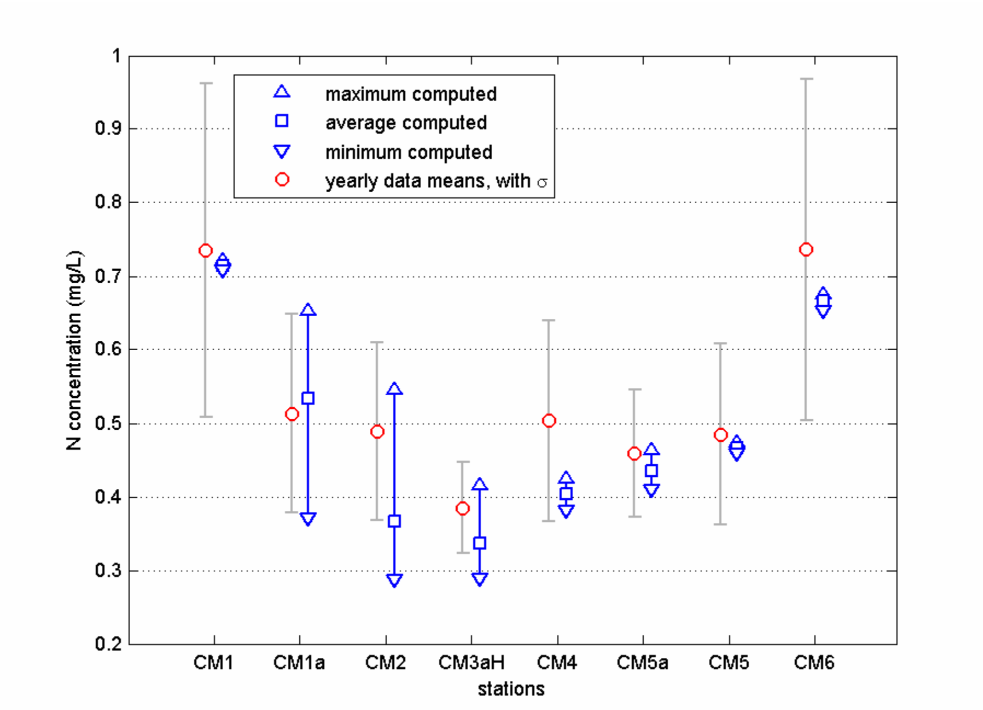


Figure VI-3. Comparison of measured total nitrogen concentrations (means for individual years and means of all data together) and calibrated model output at stations in the Stage Harbor system. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed concentration for the same period (square markers). The background concentration (0.29 mg/L) in Nantucket Sound is indicated using a solid line.

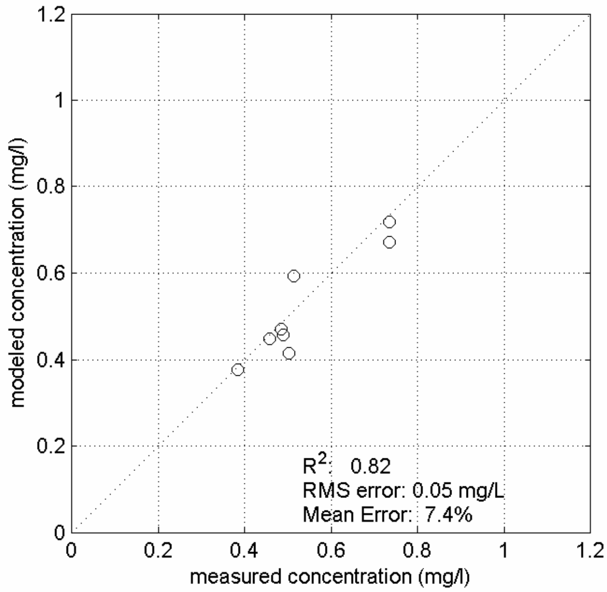


Figure VI-4. Total nitrogen calibration target values from the Stage Harbor model are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) and error (rms) for the model are also presented.

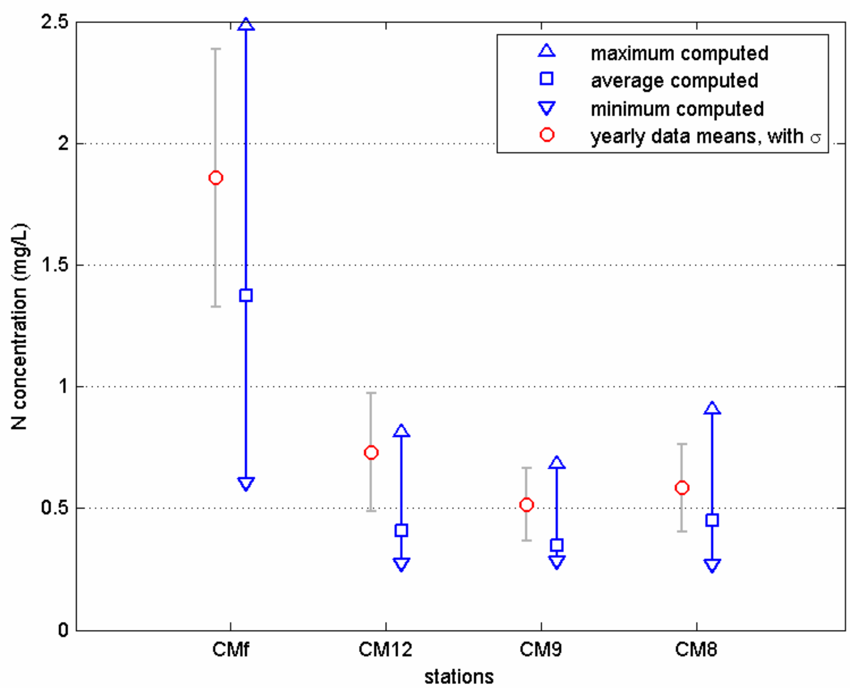


Figure VI-5. Comparison of measured total nitrogen concentrations (means for individual years and means of all data together) and calibrated model output at stations in the Sulphur Springs system, with Cockle Cove Creek (CCC). Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed concentration for the same period (square markers). The background concentration (0.29 mg/L) in Nantucket Sound is indicated using a solid line.

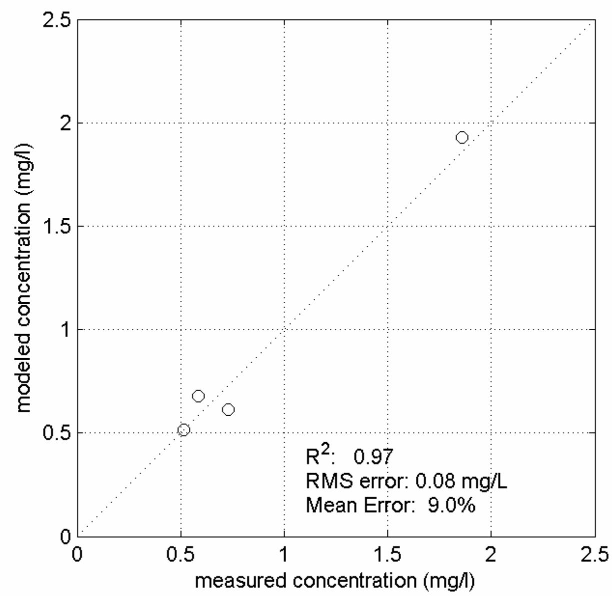


Figure VI-6. Total nitrogen calibration target values from the Sulphur Springs model are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) and error (rms) for the model are also presented.

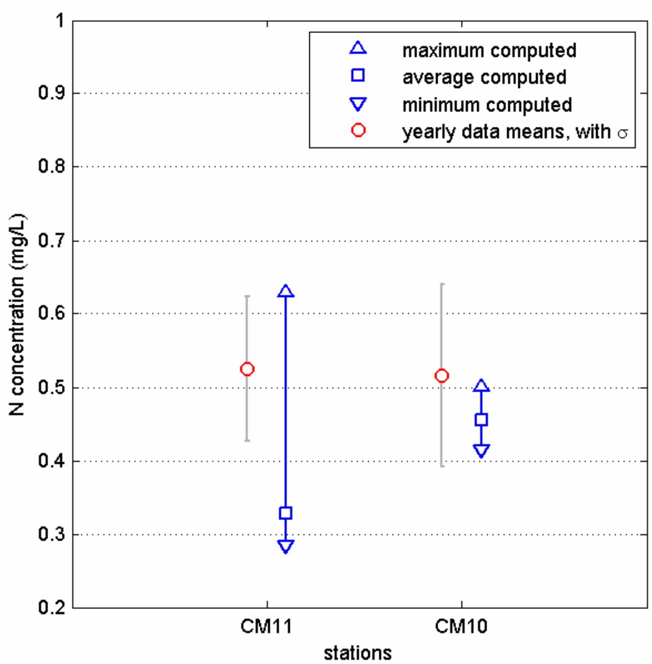


Figure VI-7. Comparison of measured total nitrogen concentrations (means for individual years and means of all data together) and calibrated model output at stations in the Taylors Pond system, with Mill Creek. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed concentration for the same period (square markers). The background concentration (0.29 mg/L) in Nantucket Sound is indicated using a solid line.

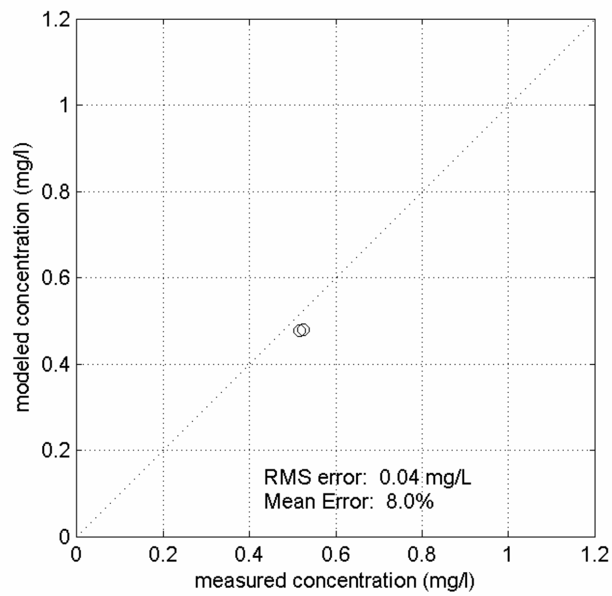


Figure VI-8. Total nitrogen calibration target values from the Taylors Pond model are plotted against measured concentrations, together with the unity line. Computed errors (rms) for the model are also presented.

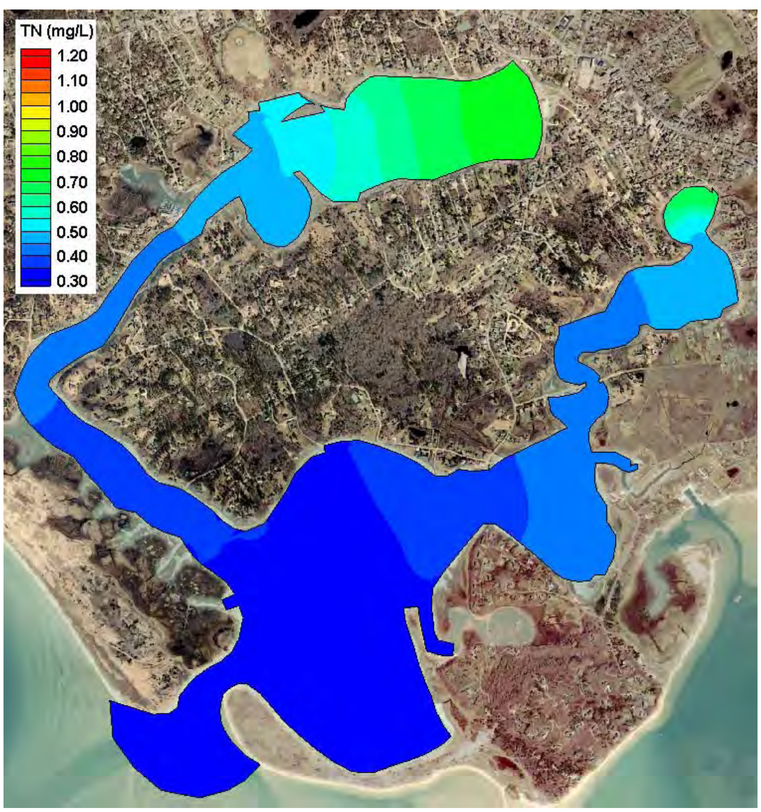


Figure VI-9. Contour plot of average total nitrogen concentrations from results of the present conditions loading scenario, for the Stage Harbor system.

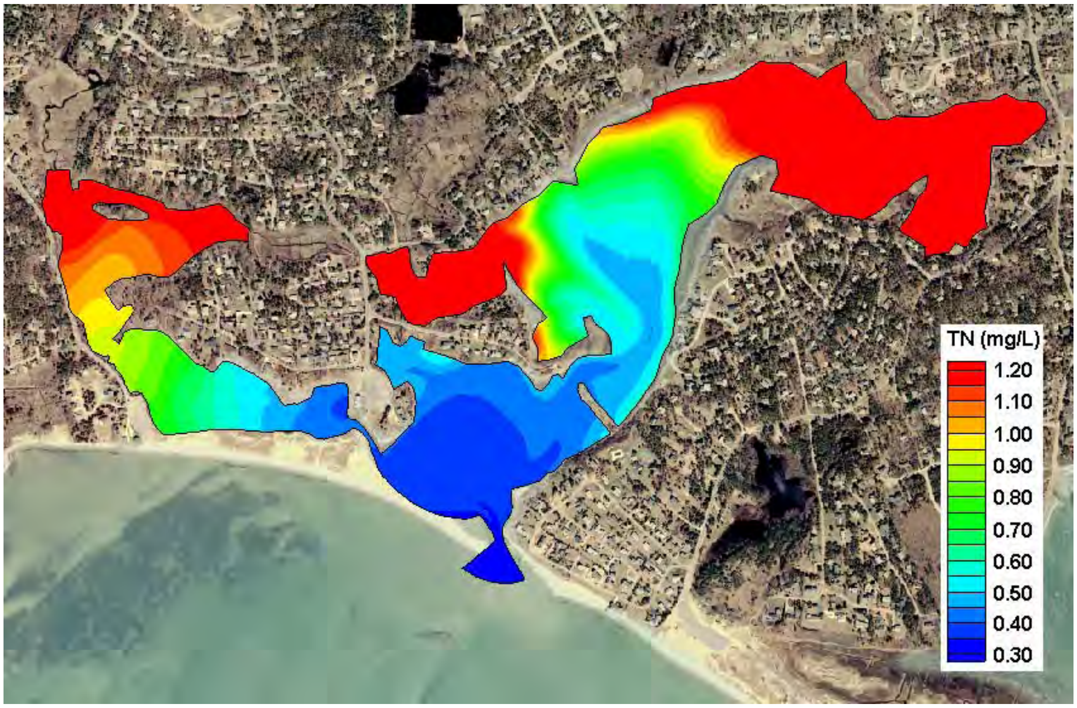


Figure VI-10. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Sulphur Springs/Cockle Cove Creek system, for present loading conditions.

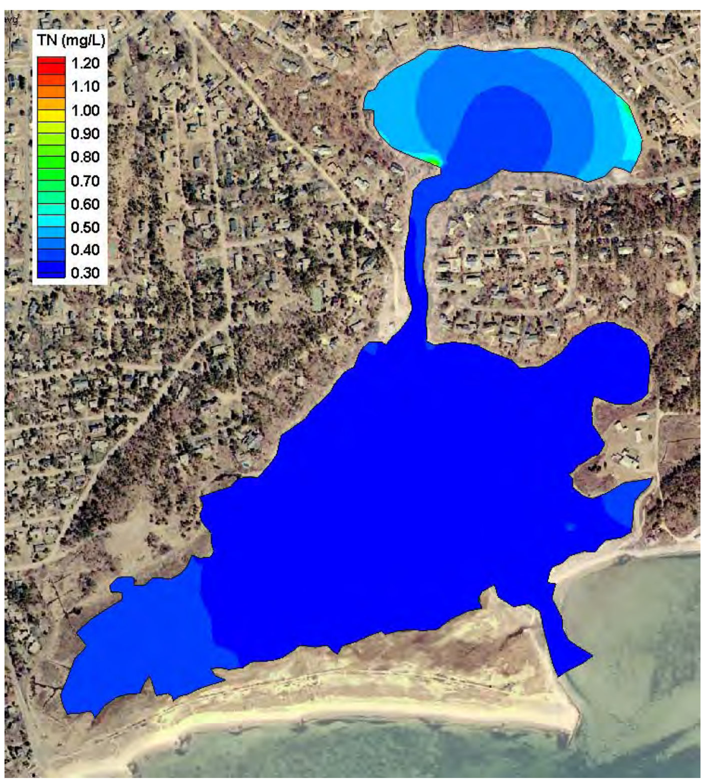


Figure VI-11. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Taylors Pond/Mill Creek system, for present loading conditions.

VI.2.5 Build-Out and No Anthropogenic Load Scenarios

To assess the influence of nitrogen loading on total nitrogen concentrations within each of the modeled embayment systems, two standard water quality modeling scenarios were run: a "build-out" scenario based on potential development (described in more detail in Section IV) and a "no anthropogenic load" or "no load" scenario assuming only atmospheric deposition on the watershed and sub-embayment, as well as a natural forest within each watershed.

For these scenarios, projected benthic fluxes (for both the build-out and no load scenarios) are based upon projected PON concentrations and watershed loads, determined as:

$$(\textit{Projected N flux}) = (\textit{Present N flux}) * [PON_{projected}]/[PON_{present}]$$

where the projected PON concentration is calculated by,

$$[PON_{projected}] = R_{load} * \Delta PON + [PON_{(present\ offshore)}],$$

using the watershed load ratio,

$$R_{load} = (\textit{Projected N load}) / (\textit{Present N load}),$$

and the present PON concentration above background,

$$\Delta PON = [PON_{(present\ flux\ core)}] - [PON_{(present\ offshore)}].$$

Comparisons of the watershed loading analyses are shown in Table VI-4. Loads are presented in kilograms per day (kg/day) in this Section, since it is inappropriate to show benthic flux loads in kilograms per year due to seasonal variability. The build-out scenarios for these systems indicate that watershed nitrogen loads would increase between 10% and 44% as a result of potential future development. The maximum increase in watershed loading resulting from future development would occur in the Stage Harbor main basin watershed. For the no load scenarios, almost all of the load entering the watershed is removed; therefore, the load is generally lower than existing conditions by over 80%.

For the build out scenario, a breakdown of the total nitrogen loads entering each sub-embayment is shown in Table VI-5. Due to the highly variable nature of bottom sediments and other estuarine characteristics of Chatham's coastal embayments, the measured benthic flux for existing conditions also is variable. For build-out conditions, some sub-embayments have approximately twice the benthic flux as total watershed load (e.g. Oyster Pond). For other sub-embayments, the benthic flux is relatively low or negative (indicating a net uptake of nitrogen in the bottom sediments).

Table VI-4. Comparison of sub-embayment watershed loads used for modeling of present, build out, and no-anthropogenic ("no-load") loading scenarios of the Stage Harbor and South Coastal embayment systems. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.

sub-embayment	present load (kg/day)	build out (kg/day)	build out % change	no load (kg/day)	no load % change
Stage Harbor					
Oyster Pond	10.041	13.463	+34.1%	1.236	-87.7%
Oyster River	9.400	11.830	+25.9%	1.619	-82.8%
Stage Harbor	2.000	2.871	+43.6%	0.318	-84.1%
Mitchell River	2.586	2.844	+10.0%	0.249	-90.4%
Mill Pond	3.600	3.981	+10.6%	0.386	-89.3%
Little Mill Pond	1.255	1.548	+23.3%	0.151	-88.0%
Sulphur Springs					
Sulphur Springs	9.093	11.348	+24.8%	1.003	-89.0%
Bucks Creek	3.362	4.408	+31.1%	0.378	-88.8%
Cockle Cove Creek	8.427	9.701	+15.1%	0.496	-94.1%
Waste Water TF	3.205	3.205	0.0%	0.000	-100.0%
Taylors Pond					
Mill Creek	4.559	5.696	+24.9%	0.559	-87.7%
Taylors Pond	6.219	8.490	+36.5%	0.699	-88.8%

Following development of the various nitrogen loading estimates for the build out scenario, the water quality model was run to determine nitrogen concentrations within each sub-embayment. Total nitrogen concentrations in the boundary waters (Nantucket Sound for the three south shore embayments) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from build out was relatively small as shown in Table VI-6. These results are shown pictorially in Figures VI-12 to VI-14. Again, the range of nitrogen concentrations shown represent the complete range of total nitrogen values observed in Chatham's south coast embayments. This allows direct comparison of nitrogen concentrations between regional embayment systems.

Table VI-5. Sub-embayment loads used for modeling of build out scenarios of the Stage Harbor and South Coastal embayment systems, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	watershed load (kg/day)	atmospheric deposition (kg/day)	benthic flux (kg/day)
Stage Harbor			
Oyster Pond	13.463	1.784	28.093
Oyster River	11.830	1.055	1.068
Stage Harbor	2.871	3.244	4.500
Mitchell River	2.844	0.882	4.718
Mill Pond	3.981	0.627	3.744
Little Mill Pond	1.548	0.121	2.020
Sulphur Springs			
Sulphur Springs	11.784	0.378	-4.190
Bucks Creek	4.408	0.132	3.264
Cockle Cove Creek	9.701	0.060	-0.657
Waste Water TF	3.205	-	-
Taylors Pond			
Mill Creek	5.696	0.167	-0.071
Taylors Pond	8.490	0.186	1.705

Table VI-6. Comparison of model average total N concentrations from present loading and build out scenario, with percent change, for South Coastal embayments and Stage Harbor.

sub-embayment	present (mg/L)	build out (mg/L)	% change
Stage Harbor			
Oyster Pond –upper	0.714	0.757	+6.0%
Oyster Pond – lower	0.534	0.578	+8.1%
Oyster River	0.367	0.382	+4.2%
Stage Harbor – main	0.336	0.344	+2.5%
Stage Harbor – upper	0.403	0.422	+4.5%
Mitchell River	0.435	0.457	+5.1%
Mill Pond	0.466	0.492	+5.7%
Little Mill Pond	0.666	0.730	+9.5%
Sulphur Springs			
Cockle Cove Cr. – mid	1.373	1.475	+7.4%
Cockle Cove Cr. – low	0.410	0.427	+4.1%
Bucks Creek	0.347	0.361	+4.2%
Sulphur Springs	0.452	0.494	+9.2%
Taylors Pond			
Mill Creek	0.329	0.352	+7.2%
Taylors Pond	0.455	0.471	+3.5%

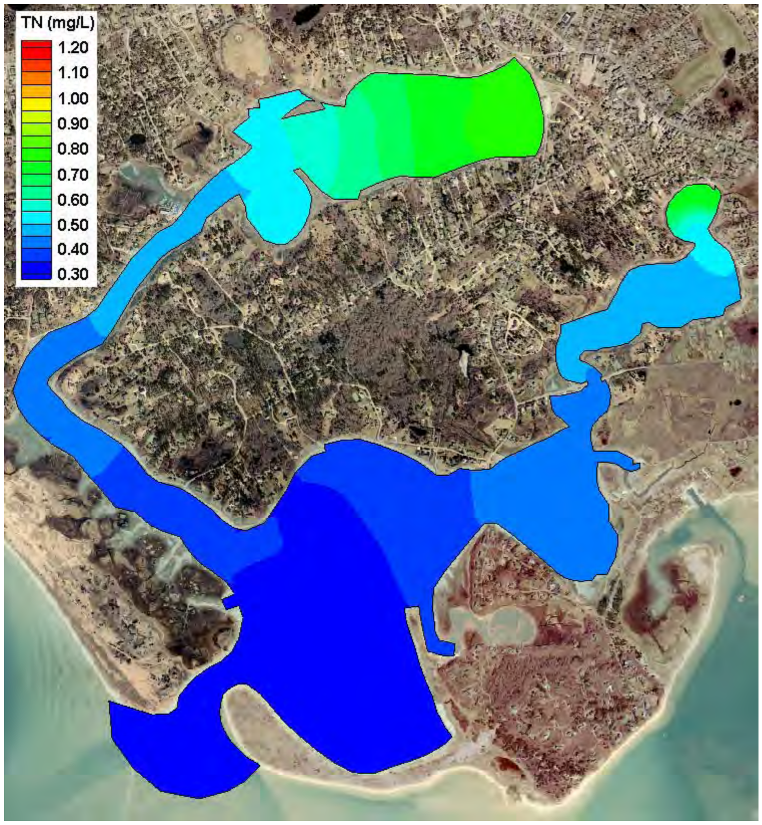


Figure VI-12. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Stage Harbor system, for projected build out loading conditions.

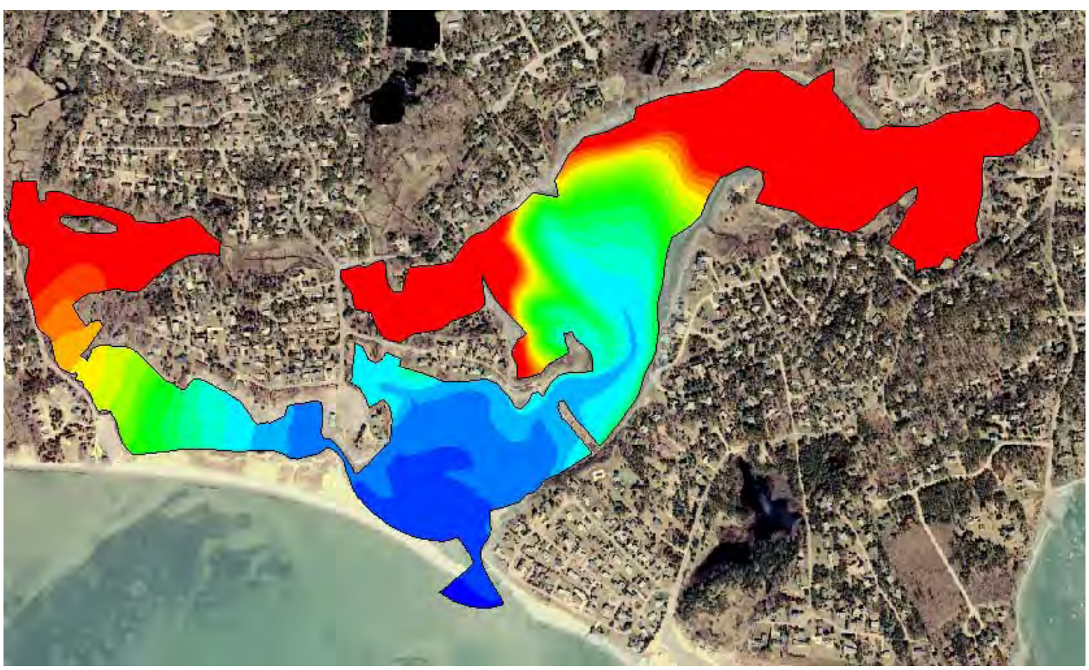


Figure VI-13. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Sulphur Springs/Cockle Cove Creek system, for projected build out loading conditions

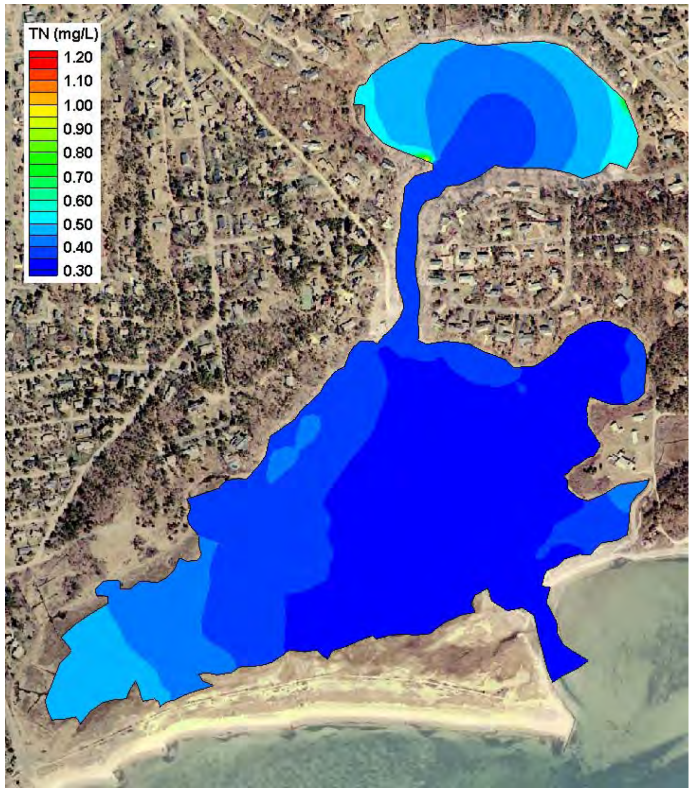


Figure VI-14. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Taylors Pond/Mill Creek system, for projected build out loading conditions.

A breakdown of the total nitrogen load entering each sub-embayment for the no anthropogenic load scenarios is shown in Table VI-7. For no load conditions, some sub-embayments have a benthic load that is significantly larger than the watershed load (e.g. Oyster Pond and Stage Harbor). Additionally, atmospheric deposition directly to each sub-embayment becomes a greater percentage of the total nitrogen load as the watershed load and related benthic flux decrease.

Following development of the various nitrogen loading estimates for the no load scenario, the water quality model was run to determine nitrogen concentrations within each sub-embayment. Again, total nitrogen concentrations in the boundary waters (Nantucket Sound) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from "no load" was relatively significant as shown in Table VI-8. These results are shown pictorially in Figures VI-15 to VI-17. Again, the range of nitrogen concentrations shown represent the complete range of total nitrogen values observed in Chatham's coastal embayments. This allows direct comparison of nitrogen concentrations between regional embayment systems.

Table VI-7. Sub-embayment loads used for modeling of no-anthropogenic loading scenarios of the Stage Harbor and South Coastal embayment systems, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	watershed load (kg/day)	atmospheric deposition (kg/day)	benthic flux (kg/day)
Stage Harbor			
Oyster Pond	1.236	1.784	13.320
Oyster River	1.619	1.055	0.634
Stage Harbor	0.318	3.244	1.866
Mitchell River	0.249	0.882	2.531
Mill Pond	0.386	0.627	2.081
Little Mill Pond	0.151	0.121	0.938
Sulphur Springs			
Sulphur Springs	1.003	0.378	-2.038
Bucks Creek	0.378	0.132	1.526
Cockle Cove Creek	0.496	0.060	-0.274
Waste Water TF	0.000	-	-
Taylors Pond			
Mill Creek	0.559	0.167	-0.031
Taylors Pond	0.699	0.186	0.631

Table VI-8. Comparison of model average total N concentrations from present loading and the no anthropogenic ("no load") scenario, with percent change, for South Coastal embayments and Stage Harbor. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions).

sub-embayment	present (mg/L)	no load (mg/L)	% change
Stage Harbor			
Oyster Pond –upper	0.714	0.470	-34.1%
Oyster Pond – lower	0.534	0.395	-26.1%
Oyster River	0.367	0.321	-12.5%
Stage Harbor – main	0.336	0.308	-8.4%
Stage Harbor – upper	0.403	0.337	-16.4%
Mitchell River	0.435	0.350	-19.5%
Mill Pond	0.466	0.362	-22.3%
Little Mill Pond	0.666	0.427	-35.9%
Sulphur Springs			
Cockle Cove Cr. – mid	1.373	0.444	-67.7%
Cockle Cove Cr. – low	0.410	0.310	-24.3%
Bucks Creek	0.347	0.293	-15.6%
Sulphur Springs	0.452	0.301	-33.5%
Taylors Pond			
Mill Creek	0.329	0.294	-10.7%
Taylors Pond	0.455	0.311	-31.7%

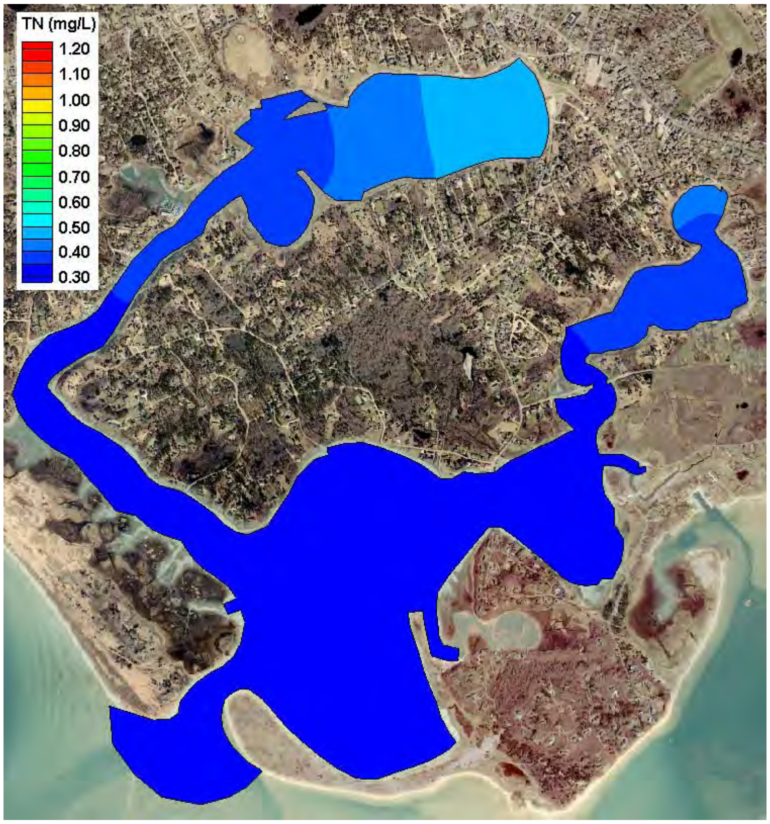


Figure VI-15. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Stage Harbor system, for no anthropogenic loading conditions.

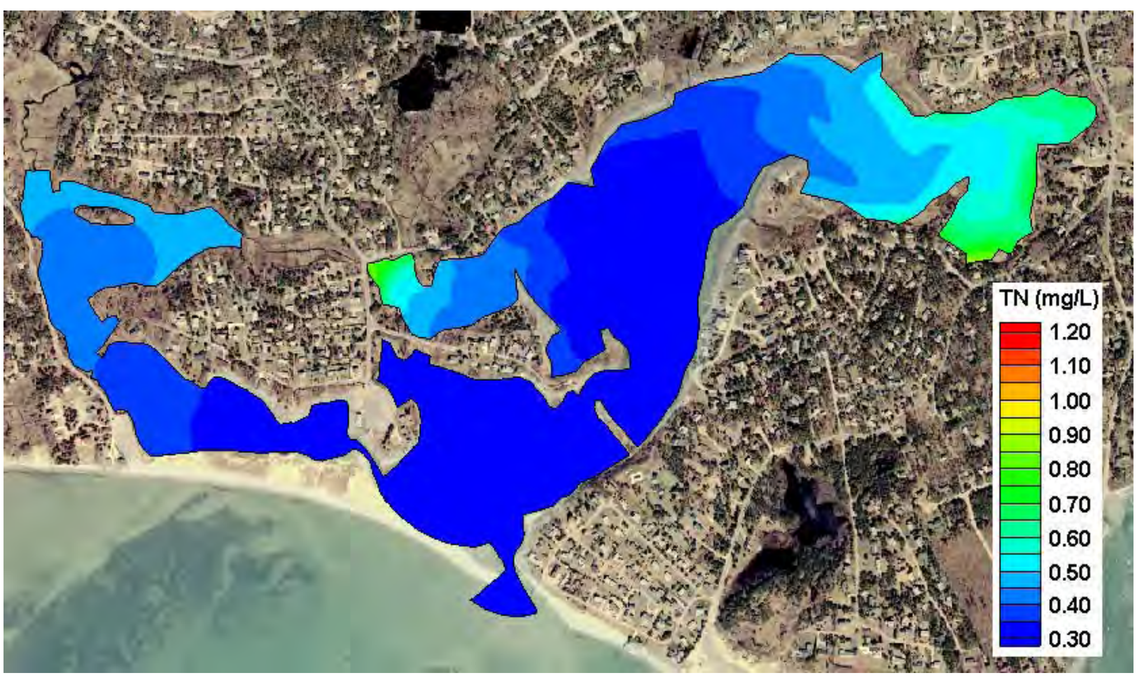


Figure VI-16. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Sulphur Springs/Cockle Cove Creek system, for no anthropogenic loading conditions.

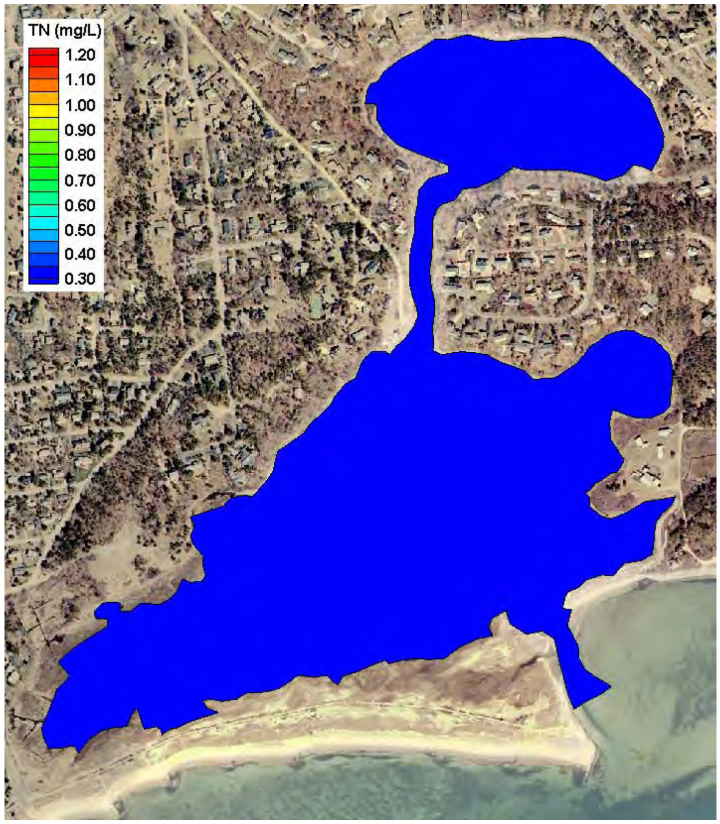


Figure VI-17. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Taylors Pond/Mill Creek system, for no anthropogenic loading conditions.

VIII. CRITICAL NUTRIENT THRESHOLD DETERMINATION AND DEVELOPMENT OF WATER QUALITY TARGETS

VIII.1. ASSESSMENT OF NITROGEN RELATED HABITAT QUALITY

Determination of site specific nitrogen thresholds for an embayment requires the integration of key habitat parameters (infauna and eelgrass), sediment characteristic data, and nutrient related water quality information, (particularly dissolved oxygen and chlorophyll a). Additional information on temporal changes within each sub-embayment and its watershed further strengthen the analysis. These data were all collected to support threshold development in the Stage Harbor, Sulphur Springs and Taylor Pond Systems by the MEP Technical Team and were discussed in Section VII. Nitrogen threshold development builds on these data and links habitat quality to summer water column nitrogen levels from long-term baseline water quality monitoring (Chatham Water Watchers, Chatham Water Quality Monitoring Program, and MEP Team; Table VIII-1).

The three embayment systems in this study displayed a range of habitat quality, both between systems and along the longitudinal axis of the larger systems. In general, sub-embayments show decline in habitat quality moving from the inlet to the inland-most tidal reach. This trend is seen in both the nitrogen levels (highest inland), eelgrass distribution, infaunal community stress indicators and community properties, as well as summer dissolved oxygen and chlorophyll a records. The following is a brief synopsis of the present habitat quality within each of the five embayment systems. The underlying quantitative data is presented on nitrogen (Section VI), oxygen and chlorophyll a (Section VII-1), eelgrass (Section VII-2), and benthic infauna (Section VII-3).

Stage Harbor System – Little Mill Pond, Mill Pond, and Oyster Pond have elevated nitrogen levels and have lost historic eelgrass beds which once covered most of their respective basins, although eelgrass beds within Oyster Pond appear to have been restricted to its lower ~1/3 with only fringing beds in the shallow areas of the upper portion and oxygen depletion is observed during summer in each system with Mill Pond (and presumably Little Mill Pond) having ecologically significant declines (<3 mg L^{-1}). Oyster Pond had less oxygen depletion possibly due to its greater fetch for ventilation with the atmosphere. Chlorophyll a levels were consistent with the observed oxygen depletion. The lower reaches of the Oyster River and Upper Stage Harbor show good habitat quality as evidenced by their persistent eelgrass beds, infaunal community structure and oxygen and chlorophyll a levels. The inner-most high quality habitat is found in the lower Mitchell River/upper Stage Harbor.

Sulphur Springs System – Cockle Cove consists primarily of a salt marsh and a central tidal creek. This system contains little water at low tide and has a high assimilative capacity for nitrogen as do other New England salt marshes. The Cockle Cove tidal creek and its associated marsh area are functioning well as a salt marsh ecosystem. The nitrogen threshold established for the open water areas of the Sulphur Springs system is not applicable to the Cockle Cove salt marsh area. Based upon a detailed MEP site-specific investigation of the Cockle Cove salt marsh, it appears that the N load can be increased to this tidal creek as long as the nitrogen concentration does not increase significantly (see MEP Cockle Cove Creek Threshold Report 2006). However, potential negative effects of increased loading to Cockle Cove Creek on down-gradient Bucks Creek is a concern. This concern is addressed in a Town requested modeling scenario detailed in Section IX, below. Sulphur Springs is a shallow basin containing significant macroalgal accumulations, no eelgrass, and appears to be transitioning to salt marsh. However, Sulphur Springs basin is still functioning as an embayment, but a eutrophic one. Nitrogen levels are high (Section VI), oxygen levels become significantly depleted (6% of time <3 mg L^{-1}) and phytoplankton blooms are common and large (chlorophyll a levels >20 ug L^{-1}). Eelgrass has not been observed for over a decade.

Taylors Pond System – Taylors Pond represents the inland-most sub-embayment and is a drowned kettle pond. The lower portion of this system is comprised of a tidal salt marsh, Mill Creek. Like the Sulphur Springs System, the inner basin functions as an embayment and the tidal creek as a salt marsh with low sensitivity to nitrogen inputs. Taylors Pond is currently showing poor habitat quality. There is currently no eelgrass community and no record of eelgrass for over a decade. Water column nitrogen levels are enriched over incoming tidal waters (Section VI) and dissolved oxygen depletion to ~4 mg L^{-1} is common. Chlorophyll a levels of 10-15 ug L^{-1} are common during summer. The benthic infaunal community is impoverished, with only a mean of 43 individuals collected in the grab samples, compared to several hundred in the high quality sub-embayments.

VIII.2. THRESHOLD NITROGEN CONCENTRATIONS

The threshold nitrogen level for an embayment represents the average water column concentration of nitrogen that will support the habitat quality being sought. The water column nitrogen level is ultimately controlled by the watershed nitrogen load and the nitrogen concentration in the inflowing tidal waters (boundary condition). The water column nitrogen concentration is modified by the extent of sediment regeneration.

Threshold nitrogen levels for each of the three southern embayment systems in the Town of Chatham were developed to restore or maintain SA waters or high habitat quality. In these three systems, high habitat quality was defined as supportive of eelgrass and productive infaunal communities. Dissolved oxygen and chlorophyll a were considered in the assessment.

The approach developed by the MEP has been to select a sentinel sub-embayment within each embayment system. First, a sentinel sub-embayment is selected based upon its location within the system. The sentinel should be close to the inland-most reach as this is typically where water quality is lowest in an embayment system. Therefore, restoration or protection of the sentinel sub-embayment will necessarily create high quality habitat throughout the estuary. Second, a sentinel sub-embayment should be sufficiently large to prevent steep horizontal water quality gradients, such as would be found in the region of entry of a stream or river or in the upper most region of a narrow, shallow estuary. This second criteria relates to the ability to accurately determine the baseline nitrogen level and to conduct the predictive modeling runs. Finally, the sentinel system should be able to obtain the minimum level of habitat quality acceptable for the greater system (unless a multiple classification is to be used).

After the sentinel sub-system (or systems) is selected, the nitrogen level associated with high and stable habitat quality typically derived from a lower reach of the same system or an adjacent embayment is used as the nitrogen concentration target. Finally, the watershed nitrogen loading rate is manipulated in the calibrated water quality model to determine the watershed nitrogen load which will produce the target nitrogen level within the sentinel system. Differences between the required modeled nitrogen load to achieve the target nitrogen level and the present watershed nitrogen load represent nitrogen management goals for restoration or protection of the embayment system as a whole.

Table VIII-1. Assessment of nitrogen related habitat quality within the embayments of the Town of Chatham. Water quality stations and benthic stations were in the same basin, but not in the exact same locations. Data for this comparison is from 2000, when the eelgrass mapping and benthic infauna were assayed. Note that the 1998-2002 water quality data was used in the validation of the water quality model and that the moored instrumentation captured a greater range of dissolved oxygen and chlorophyll a than the water quality sampling programs. Ecological Assessment Classification (SMAST) attempts to integrate water quality and habitat indicators, as well as any temporal trends which have been identified. No data is represented by "-- --". Note that the water quality data used here are contemporaneous with the ecological sampling, but that the water quality data through 2005 were used in the modeling effort.

Embayment System	Depth m	Salinity ppt	Minimum D.O. Mg/L	Secchi depth m	Nitrogen DIN mg N/L	Nitrogen TN mgN/L	Phytoplankton Tot-Pig ug/L	Sediment Type	Sediment Carbon mgC/cc	MacroAlgae Abundance	Eelgrass* Cover/Density/Status	Infaunal** Community Classification	Ecological*** Assessment Class/Status
Stage Harbor System:													
Oyster Pond	2.85	29.8	7.06	2.13	0.05	0.79	5.18	Mud/Sand	737	Low	Sparse/Low/Decline	Intermediate	Mod-Fair/Decline
Oyster River	1	30.1	6.23	2.13	0.04	0.46	4.43	Sand	236	-- --	Mod/Mod/Decline	-- --	Mod-High/Decline
Stage Harbor	1.6	30.2	7.09	1.84	0.04	0.66	5.63	Sand	546	-- --	Mod/Mod/Decline	-- --	Mod-High/Decline
Stage Harbor - Upper	1.6	30	7.45	2.22	0.04	0.46	4.2	Sand/mud	950	-- --	Mod/High/Decline	Intermed/Healthy	Mod-High/Decline
Mitchell River								Sand	294	-- --	Low/Low/Decline	Intermediate	Mod/Decline
Mill Pond	4.46	30	6.57	2.12	0.04	0.50	5.2	Mud	815	-- --	0	Stressed	Poor
Little Mill Pond	4.15	29.8	5.63	2.34	0.09	0.69	6.74	Mud	1334	-- --	0	Stressed	Poor
Taylors Pond System:													
Taylors Pond	2.18	28.3	5.85	1.76	0.06	0.51	7.03	Mud	1624	Moderate	0	Stressed	Poor
Mill Creek	1.01	28.3	5.26	1	0.06	0.49	6.35	Sand	702	Moderate	-- --	-- --	-- --
Cockle Cove System:													
Sulphur Springs	1.03	28.6	4.8	1.03	0.04	0.36	5.56	Mud/Sand	1246	High	0	-- --	Poor
Bucks Creek	0.82	28.1	5.86	0.8	0.05	0.40	4.66	Sand	853	Moderate	0	-- --	Moderate
Cockle Cove Cr Mid	NA	0	NA	-- --	0.25	1.49	-- --	-- --	-- --	-- --	-- --	-- --	N/A****
Cockle Cove Cr Low	0.31	24.5	2.78	0.31	0.20	0.89	6.35	Sand	300	-- --	-- --	-- --	N/A****
Bassing Harbor System													
Bassing Harbor	1.8	28.7	6.78	1.42	0.05	0.54	5.63	Sand/mud	1186	-- --	High/Mod/Stable-Incr.	Intermediate	Mod-High
Crows Pond	4.98	29.2	6.58	1.97	0.11	0.76	5.92	Sand/mud	1292	-- --	Mod/Mod/Decline	Intermediate	Moderate
Ryder Cove - Inner	2.34	29.1	6.04	2	0.06	0.47	6.29	Mud/Sand	899	-- --	Mod/High/Decline.	Intermediate	Moderate
Ryder Cove - Outer	3.5	28.2	6.55	2.35	0.04	0.44	6.45	Mud/Sand	1210	-- --	High/Low/Stable	-- --	Moderate
Frost Fish Outer	1.1	28.5	5.48	1.1	0.16	1.24	11.02	-- --	-- --	-- --	0	-- --	Poor
Frost Fish Inner	0.6	15.3	NA	-- --	0.30	0.92	-- --	Mud	792	-- --	0	Stressed	N/A****
Muddy Creek System:													
Upper	-- --	-- --	-- --	-- --	-- --	-- --	-- --	Mud	940	-- --	0	Stressed	Poor
Lower	1.37	25.6	6.33	1.18	0.04	0.57	9.99	Mud	1447	-- --	Sparse Patch	Stressed	Poor

* Eelgrass coverage was classified as High, Moderate (Mod), Low and Absent (0); the stability of the beds was based upon areal changes since the DEP survey of 1994 (Costello).
** Infaunal communities classification is based upon the composition and number of species representative of "healthy", "Intermediate", and "stressed" conditions.
*** Classification is based upon High - low nutrient stress & high habitat quality; Moderate - moderate to fair nutrient stress & moderate habitat quality; Low - high nutrient related stress and poor habitat quality.
**** Infaunal communities reflective of salt marsh conditions.

The threshold nitrogen levels for the each embayment system was determined as follows:

Stage Harbor System – This embayment system has two upper reaches. Therefore, two sentinel sub-embayments were selected, lower Oyster Pond and Mitchell River/Mill Pond. Little Mill Pond could not be used because it is small and has steep horizontal nitrogen gradients (see Section VI). Within the Stage Harbor System, the uppermost sub-embayment supportive of high quality habitat was upper Stage Harbor (Section VII, VIII-1). Water column total nitrogen levels within this embayment region vary with the tidal stage due to high nitrogen out-flowing waters and low nitrogen inflowing waters (Section VI). The calibrated water quality model for this system indicates an average total nitrogen level in the upper Stage Harbor of about 0.40 mg N L^{-1} is most representative of the conditions within this sub-embayment. However, upper Stage Harbor does not appear to be stable based upon changes in eelgrass distribution. Therefore, a nitrogen level reflective of conditions closer to the inlet should achieve the stability required. The lower nitrogen level is equivalent to the tidally averaged total nitrogen concentration mid-way between upper Stage Harbor and Stage Harbor or 0.38 mg N L^{-1}. This threshold selection is supported by the fact that the high quality and stable habitat near the mouth of the Oyster River is also at a tidally averaged total nitrogen concentration of 0.37 mg N L^{-1}. The 0.38 mg N L^{-1} was used to develop watershed nitrogen loads required to reduce the average nitrogen concentrations in each sentinel system to this level. Tidal waters inflowing from Nantucket Sound have an average concentration of total nitrogen of 0.285 mg N L^{-1}. For the development of the Stage Harbor total nitrogen threshold, two sentinel stations were selected, one for each branch of the system. For the Mitchell River/Mill Pond branch, the existing CM5-A monitoring station was selected. For the Oyster Pond branch, the area between station CM1-A and the inlet to Oyster Pond was selected. In order for any loading scenario to meet the requirements of the threshold set for Stage Harbor, the TN concentration must be no more than 0.38 mg/ at both of these stations.

Sulphur Springs System – The Sulphur Springs basin is both the inland-most sub-embayment and also represents the largest component of the Sulphur Springs System (which also includes Mill Creek and Bucks Creek). Since this System exchanges tidal waters with Nantucket Sound (0.285 mg N L^{-1}), as does Stage Harbor, and since there is currently no high quality habitat within this system, Stage Harbor habitat quality information was used to support the Sulphur Springs thresholds analysis. The tidally averaged nitrogen threshold concentration for this system was determined to be the same as for the sentinel sub-embayments to the Stage Harbor System or 0.38 mg N L^{-1}. The 0.38 mg N L^{-1} was used to develop watershed nitrogen loads required to reduce the average nitrogen concentrations in the Sulphur Springs sentinel system to this level (CM8, in Bucks Creek). This 0.38 mg N L^{-1} threshold concentration was developed for the open water portions of the system and as previously mentioned above is not applicable to the Cockle Cove subsystem as it is functioning well as a salt marsh. As such, the Cockle Cove Creek sub-system received its own nitrogen threshold analysis, which was provided previously to the Town of Chatham by the MEP (Howes, White & Samimy 2006) and which was supported by an appended companion habitat study by MCZM (Carlisle, Smith, Callahan 2005).

Taylors Pond System – This system was approached in a similar manner to the Sulphur Springs System and for the same reasons. Taylors Pond represents the innermost and functional embayment within this system. This system also exchanges tidal waters with Nantucket Sound (0.285 mg N L^{-1}), as does the Stage Harbor System and there is no high quality stable embayment habitat within this system. Therefore, the tidally averaged nitrogen threshold concentration for this system was determined to be the same as for the sentinel sub-embayments to the Stage Harbor System or 0.38 mg N L^{-1}. The 0.38 mg N L^{-1} was used to develop watershed nitrogen loads required to reduce the average nitrogen concentrations in Taylors Pond to this level.

VIII.3. DEVELOPMENT OF TARGET NITROGEN LOADS

The tidally averaged total nitrogen thresholds derived in Section VIII-2 were used to adjust the calibrated constituent transport model developed in Section V. Watershed nitrogen loads were sequentially lowered, using reductions in septic effluent discharges only, until the nitrogen levels reached the threshold levels in each sentinel system.

As shown in Table VIII-2, the nitrogen load reductions within the Stage Harbor system necessary to achieve the threshold nitrogen concentrations were relatively high, with 100% removal of septic load required within three watersheds (Oyster Pond, Oyster River, and Stage Harbor), and 50% from the remaining watersheds (Little Mill Pond, Mill Pond and Mitchell River/Upper Stage Harbor). The resulting distribution of tidally-averaged nitrogen concentrations associated with the threshold loadings are shown in Figures VIII-1 through VIII-3.

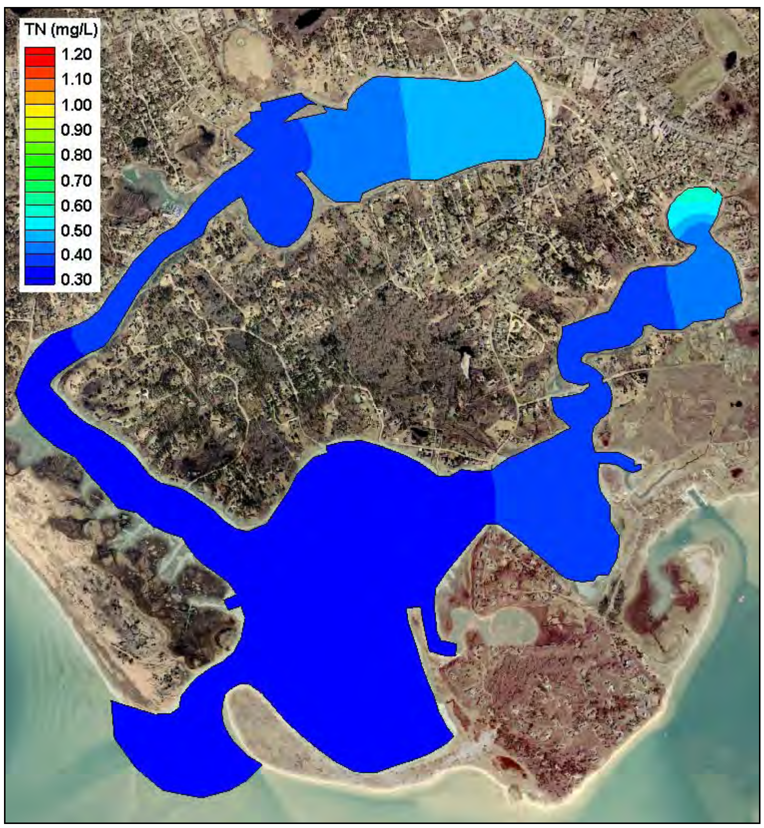


Figure VIII-1. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Stage Harbor system, for threshold loading conditions (0.38 mg/L in both Mill Pond and Oyster Pond).

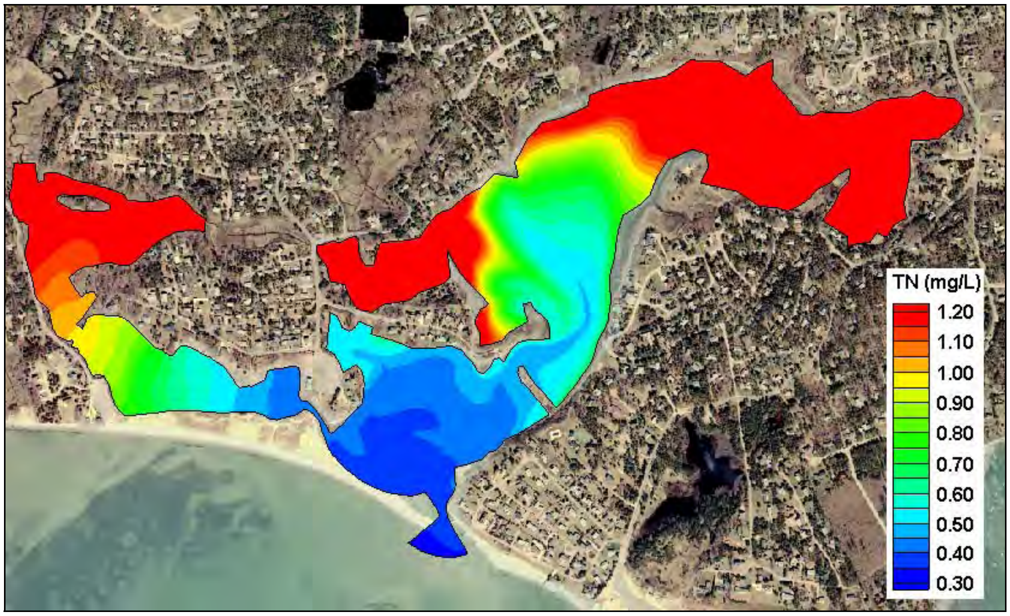


Figure VIII-2. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Sulphur Springs/Cockle Cove Creek system, for threshold loading conditions (0.38 mg/L in Sulphur Springs).

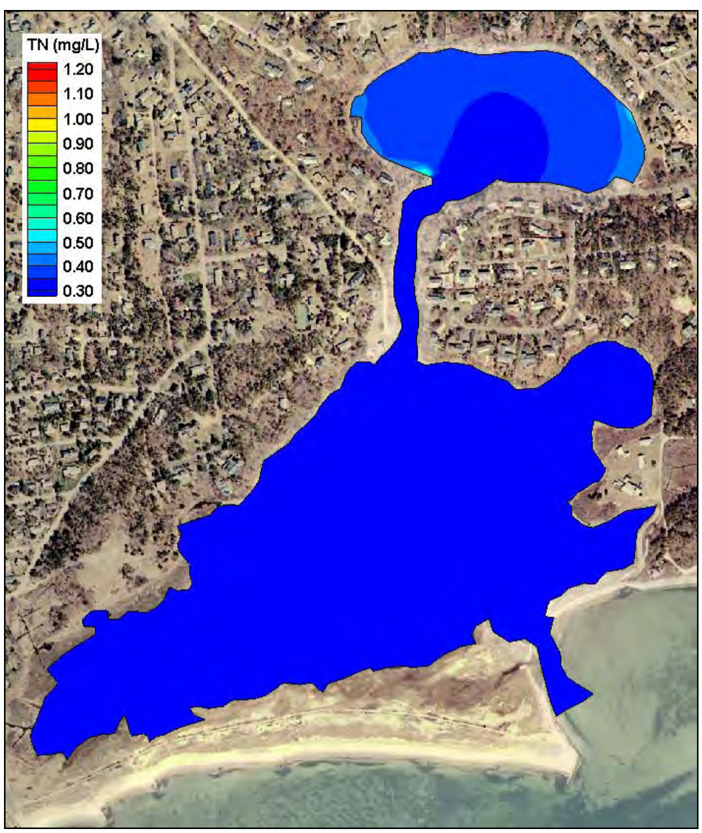


Figure VIII-3. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Taylors Pond/Mill Creek system, for threshold loading conditions (0.38 mg/L in Taylors Pond).

For the other south coastal embayments (Sulphur Springs and Taylors Pond systems), 60% of the septic load must be removed from the Sulphur Springs watershed to achieve the nitrogen concentration targets set for the system. For the Taylors Pond system, 65% for the total load to the system must be removed (100% from Mill Creek together with 40% from Taylors Pond)

It should be understood that the septic load reductions presented in Table VIII-2 provide only one option for achieving the selected threshold levels for the sentinel sub-embayments within each estuarine system, therefore these specific examples do not represent the only method for achieving this goal. However, the thresholds analysis does provide general guidance for the nitrogen management of these systems. Future water quality modeling scenarios can be run based on other nitrogen removal strategies based upon the newly updated MEP Models.

Using the septic load reductions shown in Table VIII-2, the resulting total watershed loads (i.e., including fertilizer and run-off) to each system sub-embayment can be calculated. As shown in Table VIII-3, the resulting watershed loads are reduced be between 32% and 81% in the areas that require a load reduction to meet the designated threshold.

It should be noted that there a larger percent removal of septic nitrogen is required in the present analysis than previously determined by the MEP analysis in 2003, for watershed planning it is the proportion of septic systems being removed that is the critical consideration. In the earlier analysis (2003), it was indicated that a refinement might be needed to upgrade the watershed loading data, due to water-use data issues at the time of report preparation.

The present revised analysis shows a watershed loading shift for these embayments resulting from: (1) the Town of Chatham requested that the MEP move forward with 3 quarters of water-use data, as that was all that was available. The MassDEP decided that the MEP should move forward, partially because these embayments could be revisited if additional analysis indicated that this approach resulted in an under or over estimate of wastewater flows. The follow-up analysis revealed that this approach had resulted in an overestimate of the extent of the wastewater load. While the water use data was inflated (as subsequent analysis of years of data collected by the CCC, the Chatham TAC and CAC has demonstrated), it also had secondary effects on estimates of population that were based on water use. (2) an issue resulted from use of the wastewater effluent and consumptive use terms that inflated the per capita load contribution. These issues were discovered very early on and resolved. (3) as the water-use data was being developed and refined, additional water quality data was being collected by the Town, nearly doubling the data base, which greatly strengthened (but also refined) the water quality model calibration. The nitrogen thresholds, however, did not change from the 2003 MEP analysis, except that the Cockle Cove Creek sub-system received a separate threshold analysis as noted above.

It should be noted that it is not possible to set a threshold under one set of conditions and then compare the load reductions required using another set of conditions. It only works when a consistent set of input data are used throughout the analysis.

Table VIII-2. Comparison of sub-embayment watershed septic loads used for modeling of present and threshold loading scenarios of the South Coastal embayments and Stage Harbor systems. These loads represent groundwater load contribution from septic systems only, and do not include runoff, fertilizer, atmospheric deposition and benthic flux loading terms.

Sub-embayment	Present Septic Load g/day)	New Septic Load (kg/day)	Threshold % Change
Stage Harbor			
Oyster Pond	8.099	0.000	-100.0%
Oyster River	7.052	0.000	-100.0%
Stage Harbor	1.523	0.000	-100.0%
Mitchell River	2.170	1.085	-50.0%
Mill Pond	2.956	1.478	-50.0%
Little Mill Pond	0.904	0.452	-50.0%
Sulphur Springs			
Sulphur Springs	7.863	2.971	-62.2%
Bucks Creek	2.767	2.767	0.0%
Cockle Cove Creek	4.282	4.282	0.0%
Waste Water TF	-	-	-
Taylors Pond			
Mill Creek	3.584	0.000	-100.0%
Taylors Pond	5.019	3.012	-40.0%

Table VIII-3. Comparison of sub-embayment watershed loads (including septic, runoff, and fertilizer) used for modeling of present and threshold loading scenarios of the South Coastal embayments and Stage Harbor systems. These loads do not include atmospheric deposition and benthic flux loading terms. Note that this is but one of many approaches for reaching the “target” N value.

Sub-embayment	Present Total Load (kg/day)	Threshold Total Load (kg/day)	Threshold % Change
Stage Harbor			
Oyster Pond	10.041	1.942	-80.7%
Oyster River	9.400	2.348	-75.0%
Stage Harbor	2.000	0.477	-76.2%
Mitchell River	2.586	1.501	-41.9%
Mill Pond	3.600	2.122	-41.1%
Little Mill Pond	1.255	0.803	-36.0%
Sulphur Springs			
Sulphur Springs	9.529	4.637	-51.3%
Bucks Creek	3.362	3.362	0.0%
Cockle Cove Creek	8.427	8.427	0.0%
Waste Water TF	3.205	3.205	0.0%
Taylors Pond			
Mill Creek	4.559	0.975	-78.6%
Taylors Pond	6.219	4.212	-32.3%

Table VIII-4. Sub-embayment loads used for nitrogen threshold scenarios run for the Stage Harbor and South Coastal embayment systems, with total watershed N loads, atmospheric N loads, and benthic flux.

Sub-embayment	Watershed Load (kg/day)	Atmospheric Deposition (kg/day)	Benthic Flux (kg/day)
Stage Harbor			
Oyster Pond	1.942	1.784	14.062
Oyster River	2.348	1.055	0.665
Stage Harbor	0.477	3.244	2.345
Mitchell River	1.501	0.882	3.352
Mill Pond	2.122	0.627	2.877
Little Mill Pond	0.803	0.121	1.443
Sulphur Springs			
Sulphur Springs	4.637	0.378	-2.810
Bucks Creek	3.362	0.132	2.520
Cockle Cove Creek	8.427	0.060	-0.578
Waste Water TF	3.205	-	-
Taylors Pond			
Mill Creek	0.975	0.167	-0.034
Taylors Pond	4.212	0.186	1.135

The complete tabulation of loads (including direct atmospheric deposition and benthic regeneration terms) for the three southern estuaries within the Town of Chatham is presented in Table VIII-4. Benthic flux terms have been modified to reflect the change in the total load to each system using the method described in Section VI.2.5.

The TN concentrations resulting from the threshold loadings determined for each system are compared to present modeled conditions in Table VIII-5. The greatest changes occur in Oyster Pond, where concentrations are reduced by more the 33%. Generally, TN concentrations changes are less than 16% in the other sub-embayments of the south coast systems.

Table VIII-5. Comparison of model average total N concentrations from present loading and build out scenario, with percent change, for South Coastal embayments and Stage Harbor.			
sub-embayment	present (mg/L)	threshold (mg/L)	% change
Stage Harbor			
Oyster Pond –upper	0.714	0.476	-33.3%
Oyster Pond – lower	0.534	0.404	-24.4%
Oyster River	0.367	0.325	-11.4%
Stage Harbor – main	0.336	0.314	-6.5%
Stage Harbor – upper	0.403	0.359	-11.1%
Mitchell River	0.435	0.382	-12.1%
Mill Pond	0.466	0.405	-13.0%
Little Mill Pond	0.666	0.554	-16.9%
Sulphur Springs			
Cockle Cove Cr. – mid	1.373	1.373	0.0%
Cockle Cove Cr. – low	0.410	0.410	0.0%
Bucks Creek	0.347	0.324	-6.5%
Sulphur Springs	0.452	0.381	-15.8%
Taylors Pond			
Mill Creek	0.329	0.301	-8.3%
Taylors Pond	0.455	0.381	-16.3%

IX. ALTERNATIVES TO IMPROVE TIDAL FLUSHING AND WATER QUALITY

A model-based investigation was performed to determine what water quality improvements would result from altering the flushing characteristics of both the Stage Harbor and the Sulphur Springs systems. These management scenarios were developed by the Town of Chatham, with assistance from the MEP Technical Team. Their completion required the refinements to the data supporting the MEP Linked Models for these 3 estuaries and the full re-calibration described in the sections above. Other scenarios relating to Muddy Creek were previously presented to the Town in a separate report (8/21/06).

IX.1 STAGE HARBOR

One approach to lowering nitrogen levels within the waters of an estuary is by improving tidal flushing, generally in concert with watershed nitrogen load reductions. In the Stage Harbor System, the tidal inlet has been very dynamic and has had multiple inlet locations. In an attempt to determine the potential enhancement to the water quality throughout the Stage Harbor Estuarine System, a second inlet was added to the model, to connect upper Stage Harbor to Nantucket Sound. The new inlet channel as modeled was placed just north of Morris Island, in an area where an historical inlet had once existed (Section V). The channel was modeled with a maximum depth of -2.5 ft NGVD, or at approximately MLLW. The existing inlet was unaltered in the analysis.

Changes to the hydrodynamic characteristics of the harbor resulting form the new inlet are presented in Table IX-1. The tidal prism of the Harbor does not change between present condition and the Morris Island inlet scenario, which further supports previous analysis that the present inlet flushes very efficiently, and little improvement to total volumetric exchange is possible by adding the second inlet.

However, when the TN water quality model is re-run with the second inlet, it is clear that the new configuration with multiple inlets results in an improvement in water quality conditions in the Harbor, particularly along the Upper Stage Harbor/Mill Pond branch of the system. The TN model was run with two different N boundary conditions at the new Morris Island Inlet: 1) 0.285 mg/L, which is equal to the existing TN concentration at Stage Harbor inlet, and 2) 0.300 mg/L, which was used as an estimate of maximum TN concentrations in the area between the Monomoy spit and Morris Island. These two boundary concentrations, therefore, provide a range of likely improvements. It would be useful to collect some offshore TN samples adjacent the "new" inlet, should the Town continue to evaluate this approach.

The results of the water quality model runs are presented in Table IX-2. The two scenarios representing conditions that would occur with the second inlet are compared to existing conditions. The largest improvements are seen in the Upper Stage Harbor, Mitchell River and Mill Pond sub-embayments, where TN concentrations are reduced by between 10% and 13% from existing flushing conditions. Due to the hydrodynamics of the Stage Harbor System, projected improvements from the second inlet would not be as great in the Oyster Pond/Oyster River portion of the estuary.

Table IX-1. Comparison of mean sub-embayment volumes, tidal prisms and flushing rates for present conditions and the modeled new inlet scenario, where a second inlet is cut north of Morris Island.			
system sub section	present inlet	with new Morris Island Inlet	% change
Mill Pond			
mean volume (ft3)	20,178,000	20,170,000	0.0%
mean prism (ft3)	13,155,000	13,139,000	-0.1%
local flushing rate (days)	0.8	0.8	0.1%
Stage Harbor (whole system)			
mean volume (ft3)	155,442,000	155,767,000	0.2%
mean prism (ft3)	123,654,000	123,712,000	0.0%
local flushing rate (days)	0.7	0.7	0.2%

Table IX-2. Comparison of model average total N concentrations in Stage Harbor for present watershed N loading for the modeled inlet scenarios: 1) present inlet, 2) new inlet at Morris Island with a 0.285 mg/L boundary condition and 3) new inlet at Morris Island with an alternate 0.300 mg/L boundary condition.

sub-embayment	present inlet (mg/L)	Morris Island Inlet with 0.285 mg/L boundary (mg/L)	% change	Morris Island Inlet with 0.300 mg/L boundary (mg/L)	% change
Oyster Pond –upper	0.714	0.707	-0.9%	0.708	-0.8%
Oyster Pond – lower	0.534	0.527	-1.3%	0.528	-1.1%
Oyster River	0.367	0.363	-0.9%	0.364	-0.8%
Stage Harbor – main	0.336	0.322	-4.1%	0.324	-3.5%
Stage Harbor – upper	0.403	0.349	-13.4%	0.358	-11.4%
Mitchell River	0.435	0.379	-12.8%	0.388	-10.9%
Mill Pond	0.466	0.411	-11.8%	0.419	-10.0%
Little Mill Pond	0.666	0.620	-6.9%	0.628	-5.7%

An important finding is that the threshold set for Mill Pond is met in the 0.285 mg/L boundary TN level scenario, and is nearly met in the 0.300 mg/L TN level scenario. This indicates that the threshold concentration can be achieved in Mill Pond with little to no change in the existing total N load to the Harbor System. N load removal would be still required to meet the threshold in Oyster Pond.

If the channel where dredged deeper than the -2.5 NGVD that was used in the model, there would be further improvements in TN concentrations in the Mitchell River branch of the Harbor.

IX.2 SULPHUR SPRINGS

A similar investigation regarding changes in water quality resulting from multiple inlet development was performed for Sulphur Springs. The scenario was run in relation to the nitrogen threshold analysis for Cockle Cove Creek, which indicated that nitrogen loading could be increased to this sub-system over present levels, but that the down-gradient receiving waters of Bucks Creek would likely be negatively effected (see the Cockle Cove Threshold Report for details). The concept was to evaluate the return of the historic independent inlet to Cockle Cove Creek, i.e. the separation of the Cockle Cove Creek sub-system from the Sulphur

Springs/Bucks Creek System. To evaluate this this scenario, the complete hydrodynamic model grid of Cockle Cove and Sulphur Springs was split into two separate grids, and the hydrodynamics were re-run. For the Cockle Cove Creek, a new inlet was placed through Cockle Cove Beach, in an area where the Creek historically has had an inlet directly to Nantucket Sound. Flushing improvements (from Table IX-3) are greatest for Cockle Cove Creek, where the tide prism increases by nearly 50%. In Sulphur Springs, the prism increases by only 3%.

The new Cockle Cove Creek was not optimized for flushing and stability, though the resulting tidal velocities (approximately 2 ft/sec) in the new creek indicate that the inlet, as modeled, likely would be stable.

The results of the water quality models of the two split systems show that the TN improvements would also be greatest in Cockle Cove Creek, where TN concentrations decrease by 36%. While Sulphur Springs/Bucks Creek show only modest improvements in water quality, the separation of the system from Cockle Cove Creek is of import to future additional nitrogen discharges to the Cockle Cove watershed.

Table IX-3. Comparison of mean sub-embayment volumes, tidal prisms and flushing rates for present conditions and the modeled split system scenario, where a new inlet is opened for Cockle Cove Creek, separating the creek from the Sullphur Springs system.

system sub section	present inlet	split system	% change
Cockle Cove Creek			
mean volume (ft3)	981,000	1,025,000	4.5%
mean prism (ft3)	1,176,000	1,759,000	49.6%
local flushing rate (days)	0.4	0.3	-30.1%
Sulphur Springs			
mean volume (ft3)	5,527,000	5,527,000	0.0%
mean prism (ft3)	7,387,000	7,608,000	3.0%
system flushing rate (days)	0.4	0.4	-2.9%

Table IX-4. Comparison of model average total N concentrations in the Sulphur Springs system for present watershed N loading for the modeled inlet scenarios: 1) present inlet to Bucks Creek and 2) a split system where a new inlet to Cockle Cove Creek is cut through Cockle Cove Beach.

sub-embayment	present inlet (mg/L)	split system (mg/L)	% change
Cockle Cove Cr. – mid	1.373	0.884	-35.6%
Cockle Cove Cr. – low	0.410	0.375	-8.5%
Bucks Creek	0.347	0.341	-1.8%
Sulphur Springs	0.452	0.438	-3.1%