IV. WATERSHED NITROGEN LOADING TO EMBAYMENTS: LAND USE, STREAM INPUTS, NITROGEN SEDIMENT FLUX AND 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 embayments within the Town of Chatham. Determination of watershed nitrogen inputs to Chatham's embayments 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 is conducted by biological systems which 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 (specifically nitrogen regeneration from sediments). 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 project 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) within each of the 52 subwatersheds to the 5 embayment systems (Section III). After completing a quality check of land use and reviewing water quality modeling, the 10 year time of travel subwatersheds were eliminated and the number of subwatersheds was reduced to 29. The nitrogen loading effort also involved further refinement of watershed delineations to accurately reflect shoreline areas to ponds and embayments.

In order to determine nitrogen loads from large watersheds, it is not possible to conduct measurements of individual lot-by-lot nitrogen loading. Instead, the Linked Watershed-Embayment Management Model (Howes & Ramsey 2001) uses a land-use Nitrogen Loading Sub-Model based upon subwatershed-specific land-uses and pre-determined nitrogen loading rates. The model used Chatham and Harwich specific land-use data transformed to nitrogen loads using both regional nitrogen load factors and local site-specific data (such as water use). Determination of the nitrogen loads required obtaining site-specific information regarding the 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" 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) was determined based upon site-specific studies within the Lovers Lake/Stillwater Pond discharge to Ryder Cove and within Frost Fish Creek. Attenuation during transport through each of the major fresh ponds, within the 5 embayment watersheds, was determined through (a) comparison with other Cape Cod lake studies and (b) data collected on each pond. Nitrogen recycling was also determined within each of the 5 embayment systems. Measurements were made to capture the spatial distribution of sediment nitrogen regeneration from the sediments to the overlying watercolumn. 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).

IV.1.1 Land Use and Database Preparation

MEP Technical Staff obtained digital parcel and tax assessors' data from the Towns of Chatham and Harwich. Chatham land use data is from 2002, while Harwich data is from 1999. These two databases were combined by using Geographic Information System (GIS) analysis by the MEP (Cape Cod Commission GIS Department).

Figure IV-1 shows the land uses within the study area; assessors land uses classifications (MADOR, 2002) are aggregated into eight land use categories: 1) residential, 2) commercial, 3) industrial, 4) undeveloped, 5) cranberry bog, 6) golf course, 7) public service, and 8) road right-of-way. Within the five main watersheds considered, the predominant land use is residential, most of which are single family residences. Single family residences occupy approximately 68% of the total land area and 89% of the total parcels (Figure IV-2). Commercial properties are generally concentrated along Route 28, which loops through the Town of Chatham.

In order to estimate wastewater flows within the study area, MEP staff also obtained 2001 water use information from the Town of Chatham and 2000 water use information from the Harwich Water Department. In addition, information on flow, effluent quality, and the service area delineation for the Chatham Wastewater Treatment Facility (WWTF) were obtained. The water use information was linked to the parcel and assessors data using GIS techniques.

IV.1.2 Nitrogen Loading Input Factors

Wastewater/Water Use

All wastewater is returned to the aquifer underlying Chatham either through the Town's municipal WWTF or individual on-site septic systems. The wastewater in Chatham is predominantly treated through on-site septic systems. Only 4% (266 of the 6,926) of the parcels within the study area are connected to the Town of Chatham wastewater treatment facility. The parcels connected to the WWTF are predominantly located within the watershed to the Stage Harbor System (Figure IV-6).

In order to check the reliability of parcel water use as a proxy for wastewater flow, influent flow at the WWTF was compared to parcel water use within the service area. Previous assessment of WWTF had found that measured water use within the service area closely matched influent at the WWTF. This previous assessment assumed that 90% of the water use throughout Chatham was returned to the aquifer via septic systems (Stearns and Wheler, 1999). Comparison of the 2001 water use with influent flow at the WWTF revealed that influent flow was 71% of the measured water use within the service area.

In order to address this observed difference, WWTF flows and other factors that might have caused water uses to increase in 2001 were investigated. WWTF influent flows between 1998 to 2002 were made available by the Chatham Department of Public Works. Review of these flows shows that average annual influent flow at the WWTF during the period is 40.32 million gallons (MG) with a range of 38.46 (2001) to 42.48 (2000) MG. Annual influent flows showed only about a 10% range over the five years reviewed. Since there is little change in

Sub-Embayment	***ID#***
Mill Pond (Fresh)	1
Goose Pond	2
Trout Pond	3
Schoolhouse Pond	4
Stillwater Pond	5
Lovers Lake	6
Emery Pond	7
White Pond	8
Newty Pond	9
Bassing Pond	10
Lower Muddy Creek	11
Upper Muddy Creek	12
Mill Creek	13
Taylors Pond	14
Cockle Cove	15
Bucks Creek	16
Sulfur Springs	17
Stage Harbor	18
Lower Oyster River	19
Oyster River	20
Oyster Pond	21
Mitchell River	22
Mill Pond	23
Little Mill Pond	24
Ryders Cove	25
Crows Pond	26
Bassing Harbor	27
Frostfish Creek	28
Upper Frostfish Creek	29

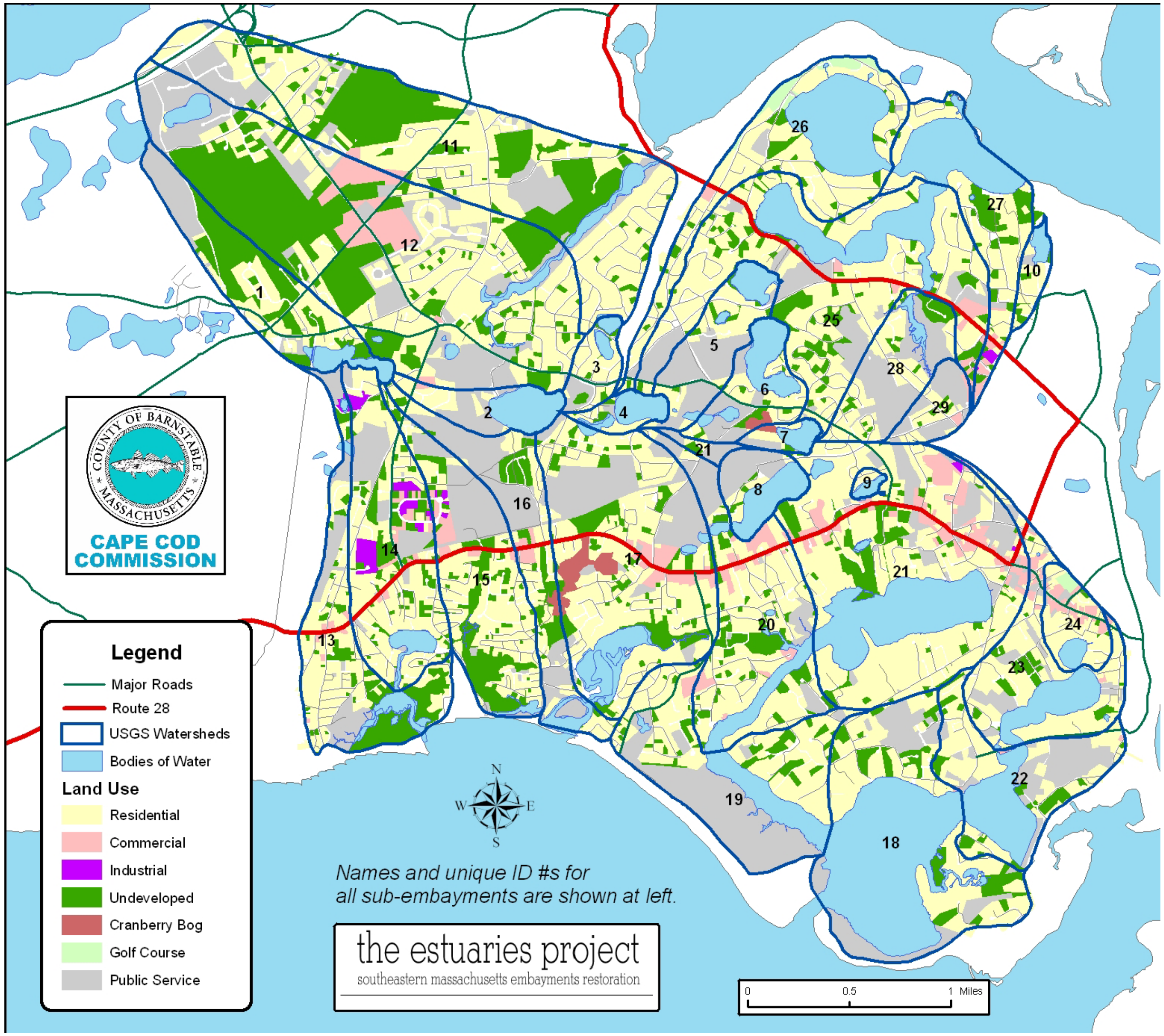


Figure IV-1. Land-use by parcel for the 5 embayment systems. Watershed data encompasses parts of the Towns of Chatham and Harwich, MA.

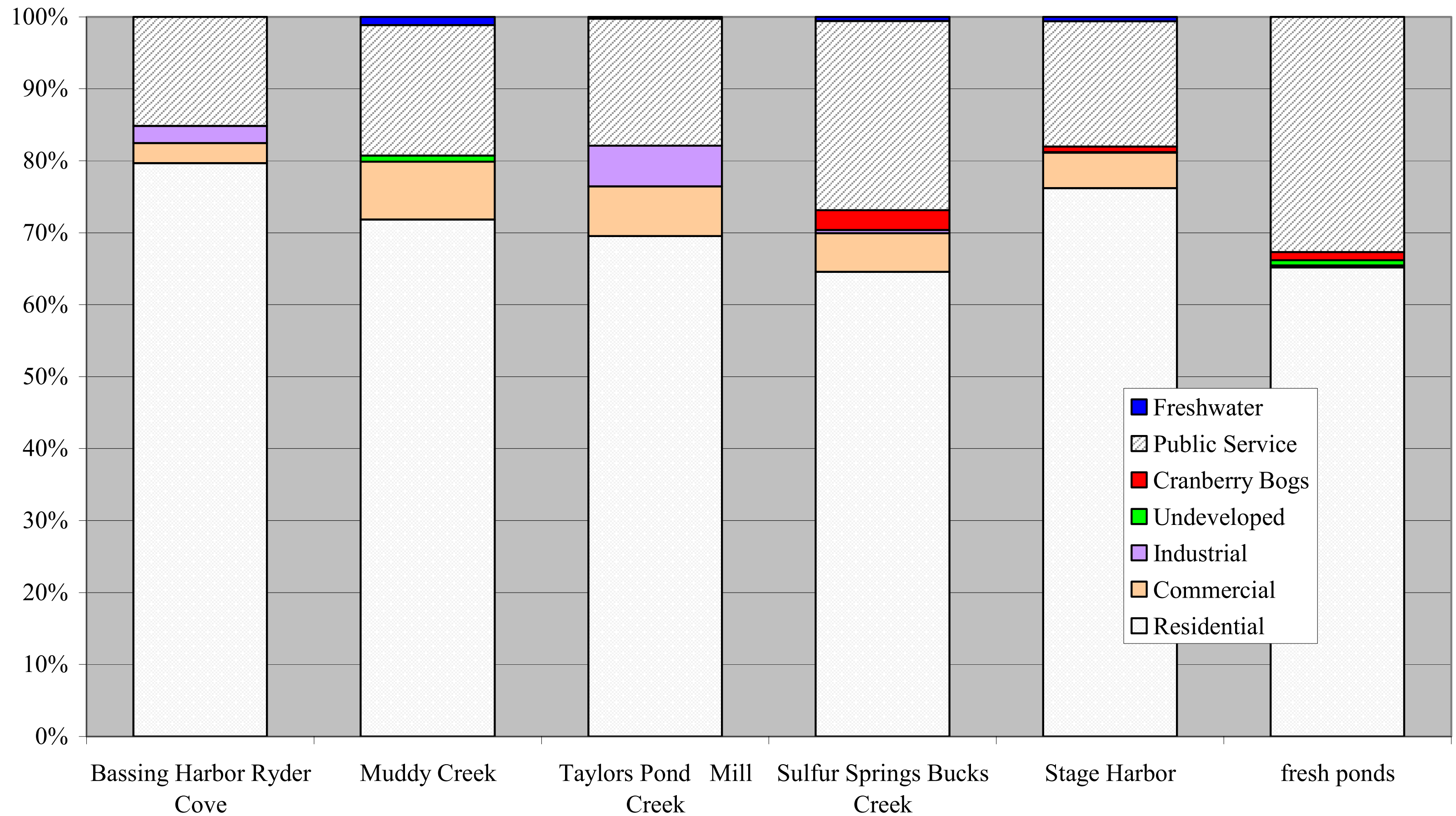


Figure IV-2. Distribution of land-uses within the watersheds to the 5 embayment systems and their freshwater ponds.

water use during this period and flow during 2001 was at the low end of the observed range, it seemed likely that that the observed difference in water use to WWTF influent volume within the service area might represent a shift in water use to purposes other than wastewater, probably lawn and shrub irrigation.

Between 1999 to 2001, annual precipitation in Chatham was cumulatively 15.4 inches below average (Figure IV-3). Since precipitation is the sole source of groundwater on Cape Cod, corresponding regional groundwater levels declined, with winter high elevations barely reaching long-term average conditions (Figure IV-4). Winter is usually the period of greatest recharge and, thus, replenishment of the aquifer and corresponding water levels. The Chatham DPW responded to the 1999 to 2001 drought by instituting a voluntary water ban in July 2001 and a mandatory ban in August 2002.

Review of annual pumping and precipitation records between 1993 and 2001 (Figure IV-5 shows that more precipitation generally results in less pumping; statistical review shows a fairly good linear relationship (R^2 = 0.55). In 2001, water pumping from municipal wells was 18% higher than in 1997. Given that the previous nitrogen loading assessment (Stearns and Wheler 1999)assumed a 10% consumptive loss in their nitrogen loading calculations, the observed 29% difference between water use and wastewater influent (71% return) appears to closely match the combination of a 10% normal consumptive loss plus an 18% increase in non-wastewater associated water use. Based on this analysis, MEP staff concluded that for the 2001 water-use data, the most appropriate breakdown of measured water use is 71% associated with wastewater and 29% for normal consumptive loss and drought associated activities (e.g. irrigation). Correspondingly, wastewater estimates for parcels with water use information were determined by multiplying water use by 0.71.

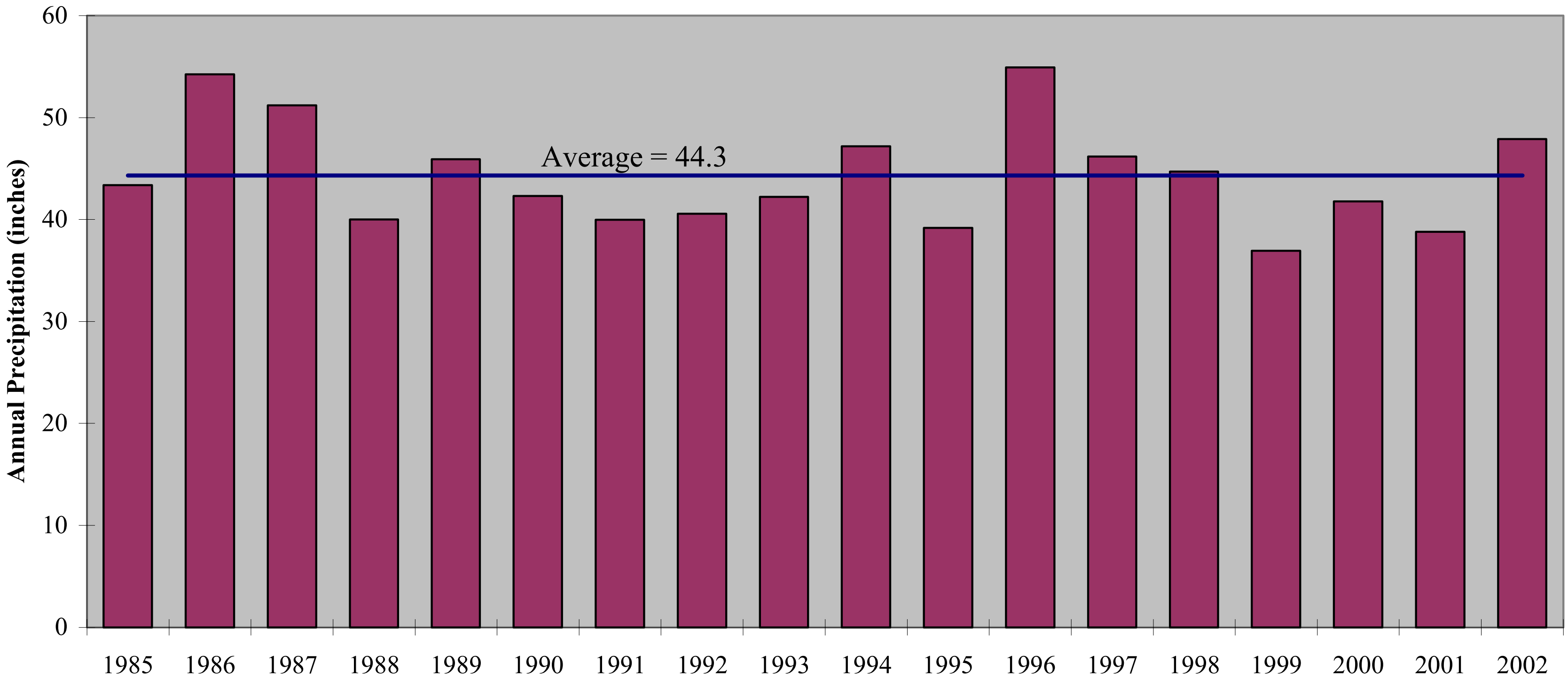
Although this estimate is appropriate for parcels with measured water use, 821 (14%) of the parcels in the study area do not have water use in the available database. These parcels are assumed to utilize private wells. A water use estimate for these parcels was developed based on measured water use from similar land uses. Of these 821 parcels without water use data, 97% are classified as residential parcels (land use codes 101 to 112) or condominium parcels and the remainder are commercial (land use codes 300 to 389). In order to address these parcels, MEP reviewed water use for residential and commercial properties in Chatham with water supply accounts (Table IV-1).

Table IV-1. Water Use in Town of Chatham

Land Use	State Class Codes	# of Parcels	Water Use (gallons per day)		
			Average	Median	Range
Residential*	101	4,420	210	154	4 to 3,077
Commercial	300 to 389	137	580	186	4 to 6,915
Industrial	400 to 433	12	522	123	18 to 2,656

*All values are based on land use from entire town

Review of Chatham water use found that significant differences existed between average and median water use flows in the various land use categories. The average water uses for commercial and industrial parcels are more than double the median, while the residential average is nearly 60 gpd greater than the median. In order to evaluate whether the average or median use data was more appropriate for determining residential wastewater flows for developed parcels without water use information.



source: 1985 thru 2/95 data obtained from National Weather Service Chatham Station; remaining data to present recorded at Town of Chatham Water Quality Laboratory

Figure IV-3. Annual Precipitation Chatham, Massachusetts.

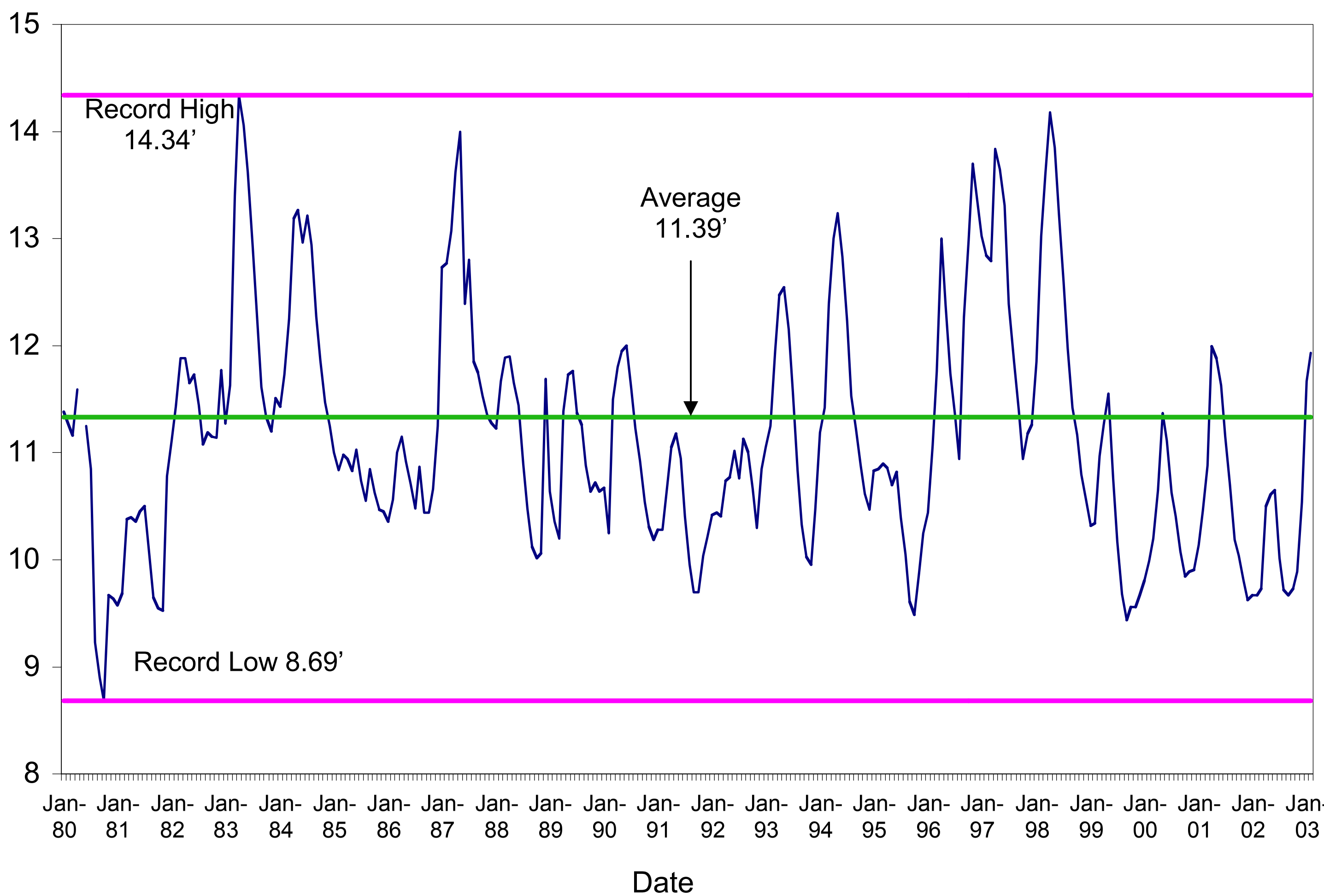
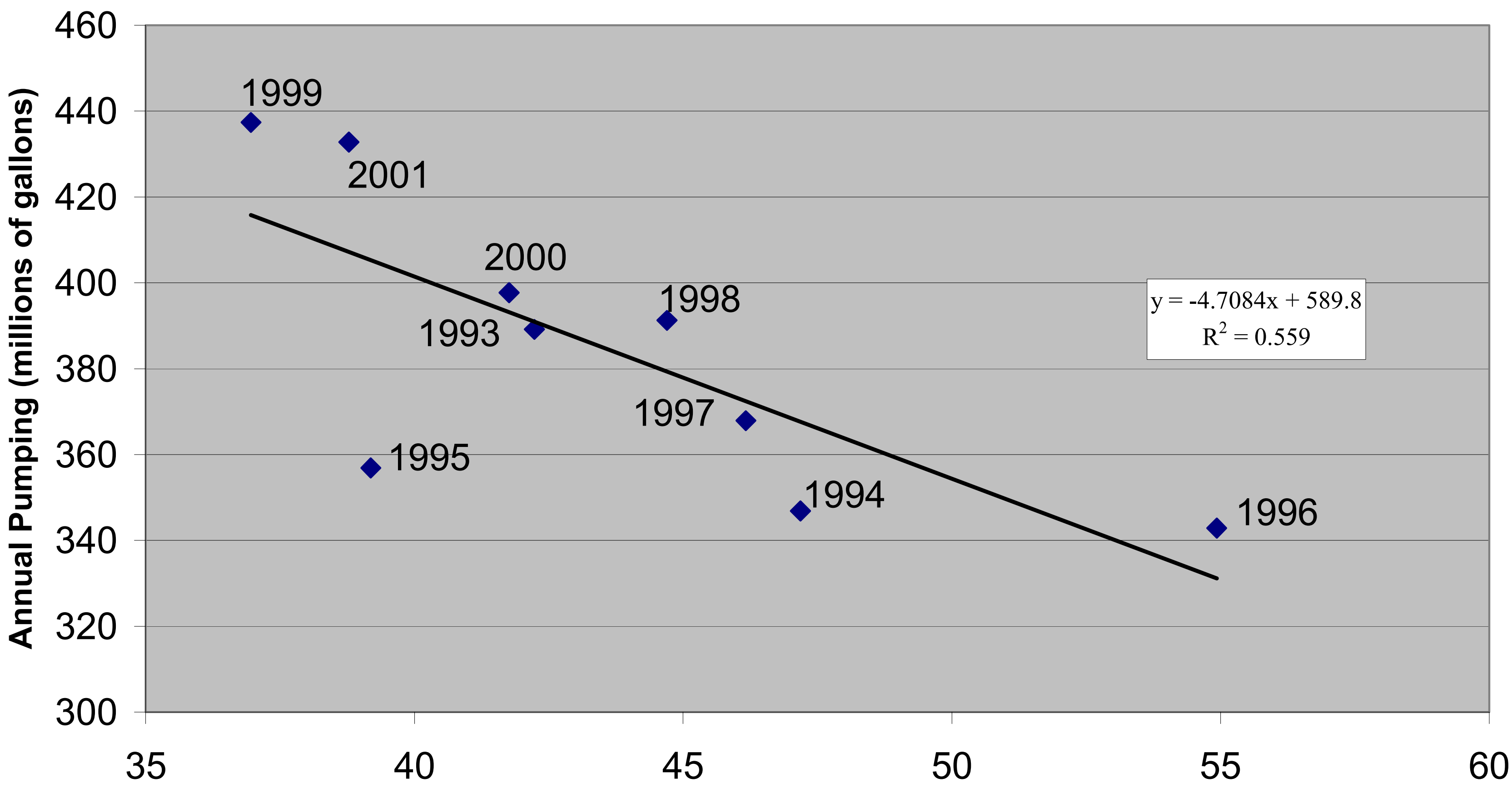


Figure IV-4. CGW138 Hydrograph. Trace indicates the water table elevation at the well site from 1980-2002.



data source: Town of Chatham

Figure IV-5. Water Supply relative to Precipitation Amounts (annual)

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). Therefore, based on these regulations each person would generate 55 gpd. Average occupancy within the town of Chatham during the 2000 US Census was 2.1 people per household. If 2.1 is multiplied by 55 gpd, the average household would generate 115 gpd of wastewater, which is nearly equal to the median residential estimate of 108 gpd based on 2001 water use (154 gpd water use multiplied by 0.71).

Because the water use is measured on an annual basis, seasonal occupancy rates for residences are indirectly accounted for in the annual water uses. In order to provide an additional check of whether the water use agreed with other measures of seasonality, 2000 US Census information was examined. The 2000 Census estimates that 3,147 of the 6,743 housing units (46.7%) in Chatham were occupied for “seasonal, recreational, or occasional use.” Previous estimates of summer population increases have estimated that the Cape’s population triples during the summer. In order for Chatham’s summer population to triple, the seasonal housing units would need to be occupied at twice the year-round occupancy or 4.2 people per household. Average household water use during the summer using the Title 5 flow of 55 gpd/person would be 232 gpd. If this use is assumed to occur for three months and is averaged with 115 gpd for the housing units occupied throughout the year, the resulting annual residential average is 144 gpd. This flow is remarkably close to 155 gpd, the median water use flow in 2001 (see Table IV-1-Water Use).

Based on this analysis, project staff felt that the median residential water use was most appropriate for use in the nitrogen loading calculations for developed residential parcels without water use information and for new residential parcels determined from the buildout assessment. Similar comparisons were not available for the commercial or industrial water uses, which have a much wider range of land uses. Average water use derived from existing commercial and industrial sites were assigned to similar land uses without water use information and for new parcels determined from the build-out assessment.

Nitrogen Loading Input Factors: Residential Lawns

In most southeastern Massachusetts watersheds, nitrogen applied to the land to fertilize residential lawns is the second major source of nitrogen to receiving coastal waters after wastewater associated nitrogen discharges. However, residential lawn fertilizer use has rarely been directly measured in previous 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 at3 pounds per 1,000 ft^2, 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 undertook an assessment of lawn fertilizer application rates and a review of leaching rates for inclusion in the land-use Nitrogen Loading Sub-Model.

The initial effort was to determine nitrogen fertilization rates for residential lawns in the Towns of Falmouth, Mashpee and Bourne, and related to inland, fresh ponds and embayments sub-watershed regions. Based upon ~300 interviews and over 2,000 surveys, a number of findings emerged: 1) average residential lawn area is ~5000 sq. ft., 2) half of the residences did not fertilize at all, and 3) the weighted average 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 for use in the 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 (loss to groundwater of 3 lb/lawn/yr).

Nitrogen Loading Input Factors: Other

The nitrogen loading factors for 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, *et al*., 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). Factors used in the nitrogen loading analyses for Chatham's embayments are listed in Table IV-2.

Table IV-2. Primary Nitrogen Loading Factors used in Chatham MEP analysis. General factors are from the 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
Wastewater	35	Impervious Surfaces	40
Road Run-off	1.5	Natural and Lawn Areas	26.5
Roof Run-off	0.75	Water Use:	
Direct Precipitation on Embayments and Ponds	1.09	For Parcels wo/water accounts:	gpd
Natural Area Recharge	0.072	Single Family Residence	154
Fertilizer:		Commercial Properties	580
Average Residential Lawn Size (ft^2)*	5,000	Industrial Properties	522
Residential Watershed Nitrogen Rate (lbs/lawn)*	1.08	For Parcels w/water accounts:	Measured annual water use
Nitrogen Fertilizer Rate for golf courses, cemeteries, and public parks determined by site-specific information			
Town of Chatham Municipal WWTF:		Wastewater Estimates:	
Annual Flow (million gallons)	38.46		
Total Nitrogen Effluent Concentration (mg/l)	7.44	Wastewater determined by multiplying water use by 0.71	

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 separately and were split (as appropriate) in order to obtain less than a 2% difference between the total land area of each watershed and the sum of the area of the parcels within each watershed. The resulting "parcelized" watersheds are shown in Figure IV-6. This 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 (small public water supplies, golf courses, etc.) were also assigned at this stage. DEP and Town of Chatham records were reviewed to determine water use for small public water supplies (*e.g.*, non-community public water supplies) and golf course superintendents for two golf courses in the study area were contacted to determine fertilizer application rates.

Following the assignment of all parcels to individual watersheds, tables were generated for each of 29 sub-watersheds to summarize water use, parcel area, frequency, sewer connections, private wells, and road area.

The 29 individual sub-watershed assessments were then integrated to generate nitrogen loading tables relating to each of the sub-embayments within each of the 5 major embayment systems. The sub-embayments represent the functional embayment units for the Linked Watershed-Embayment Model's water quality component.

For management purposes, the aggregated sub-embayment watershed nitrogen loads are separated into various nitrogen sources to support potential nitrogen mitigation alternative development: wastewater (septic systems and the WWTF), fertilizer, impervious surfaces, direct atmospheric deposition to water surfaces, and recharge from natural areas (Table IV-3 N Load summary). The output of the watershed nitrogen loading effort is the kg N per year (or day) loaded into each sub-embayment's contributing area, by land use category (Figures 7a-e) which is then adjusted for natural nitrogen attenuation during transport before use in the Linked Model.

Freshwater Pond Nitrogen Loads

Freshwater ponds on Cape Cod are generally kettle hole depressions that intercept the surrounding groundwater table revealing what some call "windows on the aquifer." Since the ponds are connected to the aquifer, the ecosystems in these ponds have the opportunity to alter the nitrogen loads flowing into them via groundwater flow. This change to the nitrogen load taking place as a result of the hydraulic interaction with the pond occurs before the loads flow back into the groundwater system through the down gradient side of the pond or stream outlet and eventual discharge into an embayment. Table IV-3 N Load summary includes both the unattenuated (nitrogen load to each subwatershed) and attenuated nitrogen loads. The attenuated loads include site-specific studies within the Lovers Lake/Stillwater Pond system and within Frost Fish Creek (see Section IV.2). Except for the site-specific studies, nitrogen attenuation in the ponds was assumed to be 40%.

This assumption was checked through the use of pond water quality information collected during late August 2001 under the Cape Cod Pond and Lake Stewardship (PALS) program, which is a collaborative Cape Cod Commission/SMAST Program. The Town of Chatham Water Quality Laboratory collected dissolved oxygen and temperature profiles, Secchi disk depth readings and water samples at various depths within the following ponds: Emery, Goose, Lovers, Mill, Schoolhouse, Stillwater, White, Trout, and Newty (Figure IV-1). Water samples were analyzed at the SMAST laboratory for total nitrogen, total phosphorus, chlorophyll *a*, alkalinity, and pH.

In order to estimate nitrogen attenuation in the ponds physical and chemical data for each pond was assessed. Available bathymetric information was reviewed relative to measured pond temperature profiles to determine the epilimnion (*i.e.*, well mixed, homothermic, upper portion of the water column) in each pond. Following this determination, the volume of this portion was determined and compared to the annual volume of recharge from each pond's watershed in order to determine how long it takes the aquifer to completely exchange the water in this portion of the pond (*i.e.*, turnover time). Using the total nitrogen concentrations collected within the epilimnion, the total mass of nitrogen within this portion of the pond was determined and, using

Sub-Embayment	ID#
Mill Pond (Fresh)	1
Goose Pond	2
Trout Pond	3
Schoolhouse Pond	4
Stillwater Pond	5
Lovers Lake	6
Emery Pond	7
White Pond	8
Newty Pond	9
Bassing Pond	10
Lower Muddy Creek	11
Upper Muddy Creek	12
Mill Creek	13
Taylors Pond	14
Cockle Cove	15
Bucks Creek	16
Sulfur Springs	17
Stage Harbor	18
Lower Oyster River	19
Oyster River	20
Oyster Pond	21
Mitchell River	22
Mill Pond	23
Little Mill Pond	24
Ryders Cove	25
Crows Pond	26
Bassing Harbor	27
Frostfish Creek	28
Upper Frostfish Creek	29

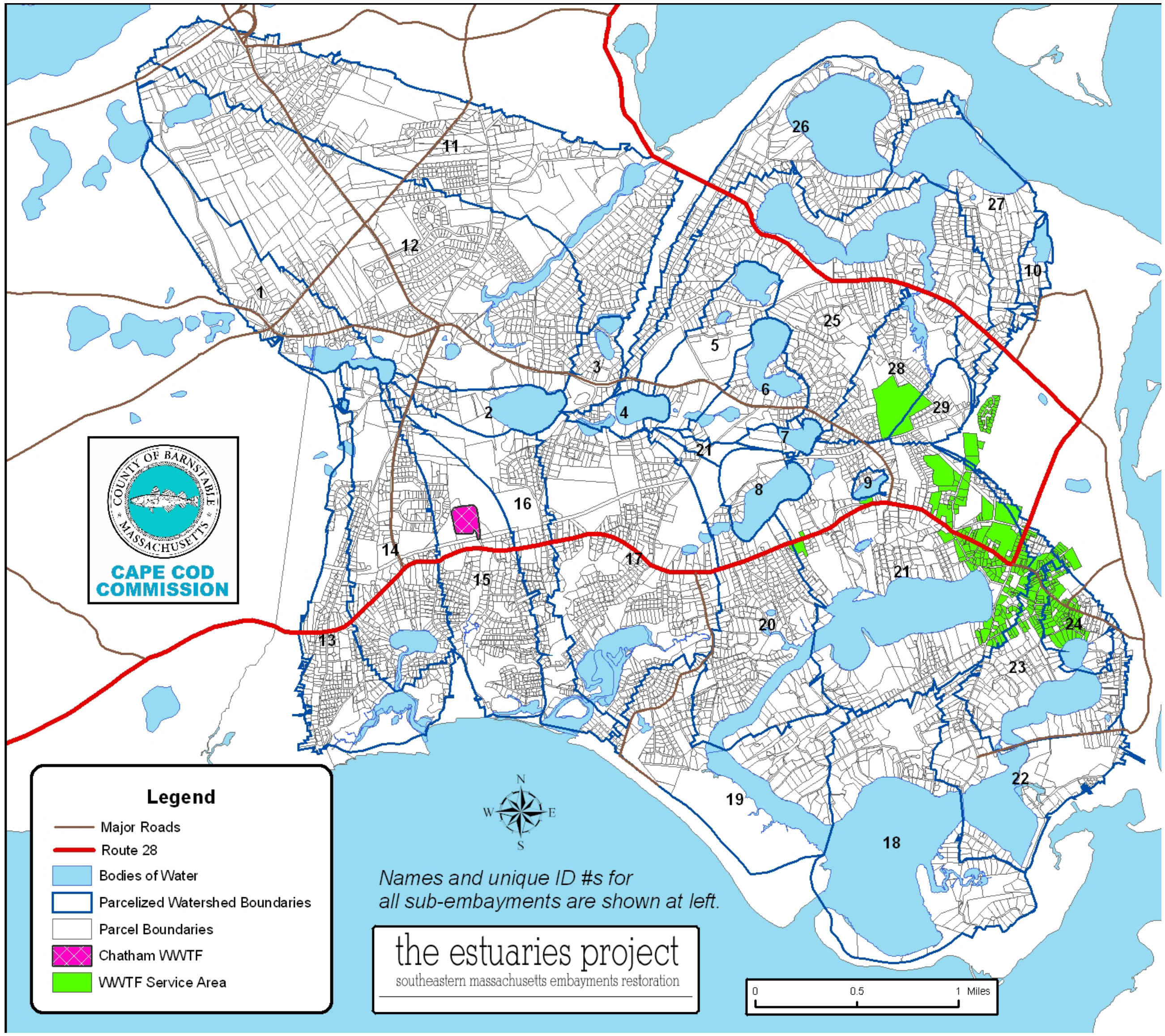


Figure IV-6. Parcel distribution within the watersheds to the 5 embayment systems.

Table IV-3a. Oyster Pond/Stage Harbor System Nitrogen Loads.

*All values in kilograms/year		Chatham N Loads by Input**:						% of Pond Outflow	Present N Loads			Buildout N Loads		
Name	Watershed ID#	Wastewater	Lawn Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout		UnAtten N Load	Atten %	Atten N Loads	UnAtten N Load	Atten %	Atten N Loads
Oyster Pond / Stage Harbor	7, 8, 9, 18, 19, 20, 21,	11556	791	684	2535	347	1615		15913		15714	17528		17324
Oyster Pond	**7, 8, 9, 21**	**4076**	**251**	**230**	**274**	**106**	**713**		**4936**		**4805**	**5649**		**5516**
Oyster Pond	21	3933	242	209	126	100	707		4610		4610	5316		5316
Newty Pond	9	58	4	1	28	1	0	100%	92	40%	55	92	40%	55
Emery Pond	7	28	3	2	63	3	0	38%	37	40%	22	37	40%	22
White Pond	8	140	7	35	180	7	12	53%	197	40%	118	203	40%	122
Oyster River	**8, 20**	**2169**	**154**	**135**	**252**	**56**	**331**		**2767**		**2698**	**3098**		**3027**
Oyster River	20	2104	151	119	169	52	326		2595		2595	2921		2921
White Pond	8	140	7	35	180	7	12	47%	172	40%	103	177	40%	106
Lower Oyster Rive	**19**	**1361**	**107**	**81**	**230**	**59**	**134**		**1838**		**1838**	**1972**		**1972**
Little Mill Pond	**24**	**494**	**52**	**41**	**44**	**10**	**58**		**642**		**642**	**700**		**700**
Mill Pond Salt	**23**	**565**	**36**	**29**	**229**	**18**	**111**		**877**		**877**	**988**		**988**
Mitchell River	**22**	**2043**	**132**	**117**	**322**	**46**	**93**		**2660**		**2660**	**2754**		**2754**
Stage Harbor	**18**	**848**	**58**	**50**	**1184**	**52**	**175**		**2193**		**2193**	**2368**		**2368**

** sums of unattenuated loads adjusted for pond shore percentages

Note that the N Loads by input show the total nitrogen input to each sub-watershed. However, the fresh ponds sub-watersheds typically contribute to more than 1 sub-embayment and therefore their sub-watershed contribution if apportioned based upon the proportion of shoreline. The total nitrogen load to each fresh pond (e.g. # 7,8,9) are adjusted to their contribution to the receiving sub-embayment in both the "roll-ups" for each sub-embayment and in the present and build-out Nitrogen Loading (right sections of table)

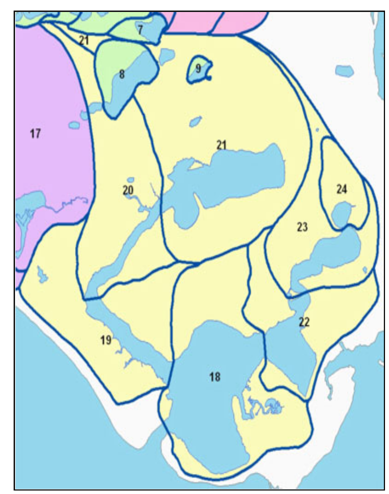


Table IV-3b. Sulphur Springs/Bucks Creek/Cockle Cove System Nitrogen Loads.

*All values in kilograms/year		Chatham N Loads by Input**:								Present N Loads			Buildout N Loads		
Name	Watershed ID#	Wastewater	Chatham WWTF	Lawn Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout	% of Pond Outflow	UnAtten N Load	Atten %	Atten N Loads	UnAtten N Load	Atten %	Atten N Loads
Sulfur Springs / Bucks	1, 2, 15, 16, 17	8257	1107	484	374	334	186	1444		10743		10606	12187		12031
Bucks Creek	**1, 2, 16**	**1303**	**0**	**81**	**61**	**94**	**42**	**284**		**1582**		**1514**	**1866**		**1788**
Bucks Creek	16	1197	0	75	56	48	38	258		1413		1413	1671		1671
Mill Pond Fresh	1	1152	0	57	48	105	37	420	6%	87	40%	52	114	40%	68
Goose Pond	2	155	0	15	8	179	9	0	22%	81	40%	49	81	40%	49
Cockle Cove	**1, 15**	**2094**	**1107**	**163**	**116**	**28**	**51**	**490**		**3560**		**3530**	**4050**		**4011**
Cockle Cove	15	2034	1107	160	114	22	49	468		3486		3486	3954		3954
Mill Pond Fresh	1	1152	0	57	48	105	37	420	5%	74	40%	44	96	40%	57
Sulfur Springs	**2, 17**	**4860**	**0**	**240**	**197**	**212**	**93**	**670**		**5601**		**5561**	**6272**		**6232**
Sulfur Springs	17	4817	0	235	195	163	91	670		5501		5501	6172		6172
Goose Pond	2	155	0	15	8	179	9	0	27%	100	40%	60	100	40%	60

** sums of unattenuated loads adjusted for pond shore percentages

Note that the N Loads by input show the total nitrogen input to each sub-watershed. However, the fresh ponds sub-watersheds typically contribute to more than 1 sub-embayment and therefore their sub-watershed contribution if apportioned based upon the proportion of shoreline. The total nitrogen load to each fresh pond (e.g. # 1, 2) are adjusted to their contribution to the receiving sub-embayment in both the "roll-ups" for each sub-embayment and in the present and build-out Nitrogen Loading (right sections of table)

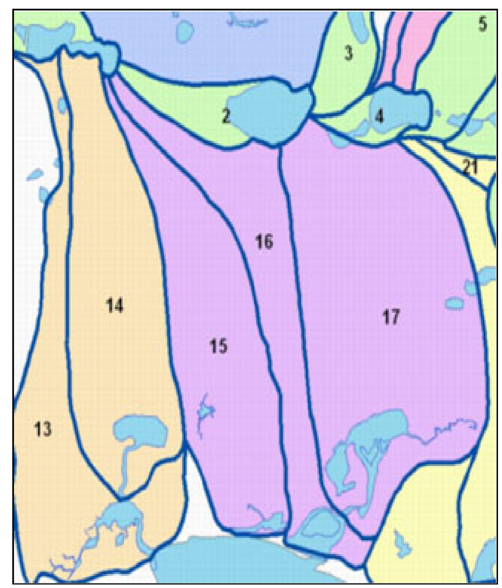


Table IV-3c. Taylors Pond/Mill Creek System Nitrogen Loads.

*All values in kilograms/year		Chatham N Loads by Input**:						% of Pond Outflow	Present N Loads			Buildout N Loads		
Name	Watershed ID#	Wastewater	Lawn Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout		UnAtten N Load	Atten %	Atten N Loads	UnAtten N Load	Atten %	Atten N Loads
Taylors Pond/ Mill Creek	**1, 13, 14**	4763	344	258	198	116	1418		5679		5344	7097		6661
Mill Creek	**1, 13**	**2034**	**157**	**111**	**90**	**50**	**382**		**2442**		**2320**	**2824**		**2665**
Mill Creek	13	1783	145	100	67	42	291		2136		2136	2427		2427
Mill Pond Fresh	1	1152	57	48	105	37	420	22%	305	40%	183	397	40%	238
Taylors Pond	**1, 14**	**2728**	**187**	**147**	**108**	**67**	**1035**		**3237**		**3024**	**4273**		**3995**
Taylors Pond	14	2289	166	129	68	53	875		2704		2704	3579		3579
Mill Pond Fresh	1	1152	57	48	105	37	420	38%	533	40%	320	693	40%	416

** sums of unattenuated loads adjusted for pond shore percentages

Note that the N Loads by input show the total nitrogen input to each sub-watershed. However, the fresh ponds sub-watersheds typically contribute to more than 1 sub-embayment and therefore their sub-watershed contribution if apportioned based upon the proportion of shoreline. The total nitrogen load to each fresh pond (e.g. # 1) are adjusted to their contribution to the receiving sub-embayment in both the "roll-ups" for each sub-embayment and in the present and build-out Nitrogen Loading (right sections of table)

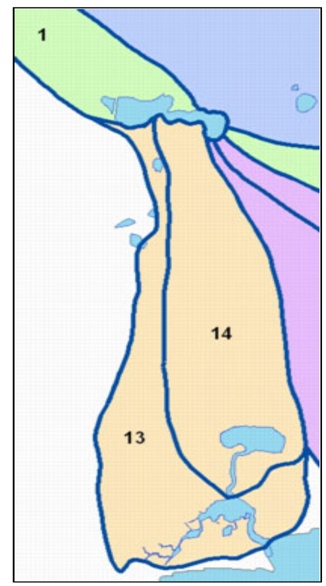


Table IV-3d. Muddy Creek System Nitrogen Loads.

*All values in kilograms/year		Chatham N Loads by Input**:						% of Pond Outflow	Present N Loads			Buildout N Loads		
Name	Watershed ID#	Wastewater	Lawn Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout		UnAtten N Load	Atten %	Atten N Loads	UnAtten N Load	Atten %	Atten N Loads
Muddy Creek	**1, 2, 3, 11, 12**	10509	560	572	334	318	1695		12293		11988	13988		13639
Lower Muddy Creek	**3, 11**	**4291**	**265**	**283**	**118**	**135**	**327**		**5092**		**4953**	**5419**		**5272**
Lower Muddy Creek	11	4048	243	249	75	128	309		4744		4744	5053		5053
Trout Pond	3	243	22	34	43	7	17	100%	349	40%	209	366	40%	220
Upper Muddy Creek	**1, 2, 12**	**6217**	**295**	**289**	**216**	**183**	**1368**		**7201**		**7035**	**8569**		**8367**
Upper Muddy Creek	12	5904	277	276	135	172	1272		6764		6764	8035		8035
Mill Pond Fresh to MC	1	1152	57	48	105	37	420	23%	321	40%	193	418	40%	251
Goose Pond	1, 2	155	15	8	179	9	0	32%	131	40%	78	135	40%	81
Goose Pond	*2*	*155*	*15*	*8*	*179*	*9*	*0*	*100%*	*366*			*366*		
Mill Pond Fresh to GP	*1*	*1152*	*57*	*48*	*105*	*37*	*420*	*6%*	*79*	*40%*	*47*	*102*	*40%*	*61*

** sums of unattenuated loads adjusted for pond shore percentages

Note that the N Loads by input show the total nitrogen input to each sub-watershed. However, the fresh ponds sub-watersheds typically contribute to more than 1 sub-embayment and therefore their sub-watershed contribution if apportioned based upon the proportion of shoreline. The total nitrogen load to each fresh pond (e.g. # 1, 2, 3) are adjusted to their contribution to the receiving sub-embayment in both the "roll-ups" for each sub-embayment and in the present and build-out Nitrogen Loading (right sections of table)

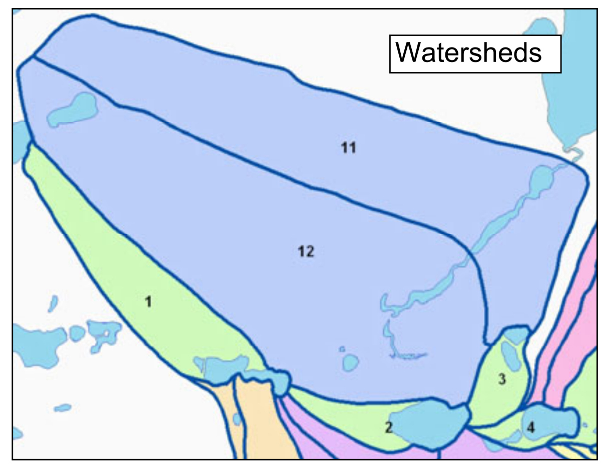
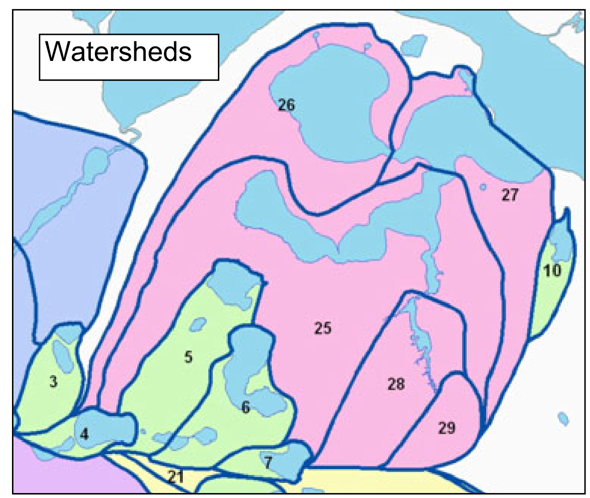


Table IV-3e. Ryder Cove/Bassing Harbor System Nitrogen Loads.

*All values in kilograms/year		Chatham N Loads by Input**:						% of Pond Outflow	Present N Loads			Buildout N Loads		
Name	Watershed ID#	Wastewater	Lawn Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout		UnAtten N Load	Atten %	Atten N Loads	UnAtten N Load	Atten %	Atten N Loads
Ryders Cove/Bassing Hbr/Crows Pd	4, 5, 6, 7, 10, 25, 26, 27, 28, 29	8137	465	477	1868	235	1109		11183		10330	12292		11368
Crows Pond	4, 26	**1871**	**111**	**89**	**512**	**40**	**94**		**2622**		**2618**	**2716**		**2712**
Crows Pond	26	1866	111	88	507	40	93		2612		2612	2705		2705
Schoolhouse Pond	4	95	9	4	112	3	12	5%	11	40%	6	11	40%	7
Ryders Cove	4, 5, 6, 7, 25, 28, 29	**5384**	**341**	**341**	**957**	**166**	**811**		**7190**		**6357**	**8000**		**7098**
Ryder Cove GW	4, 7, 25	3344	201	209	542	92	587		4388		4340	4975		4926
Ryders Cove	25	3302	197	207	474	89	584		4269		4269	4852		4852
Schoolhouse Pond	4	95	9	4	112	3	12	26%	58	40%	35	61	40%	37
Emery Pond	7	28	3	2	63	3	0	62%	62	40%	37	62	40%	37
Stillwater Pond	4, 5, 6	686	56	59	380	38	96	100%	1218	14%	717	893	14%	768
Stillwater Pond	5	264	24	29	105	22	23		444			468		
Lovers Lake	6	356	25	26	197	13	64	100%	618	52%	296	682	52%	327
Schoolhouse Pond	4	95	9	4	112	3	12	69%	156	40%	93	164	40%	98
Frostfish Creek	28, 29	1353	84	74	35	37	128		1584	18%	1299	1712	18%	1404
Bassing Harbor	10, 27	**883**	**13**	**47**	**399**	**29**	**204**		**1371**		**1355**	**1575**		**1559**
Bassing Harbor	27	853	11	46	393	28	202		1332		1332	1533		1533
Bassing Pond	10	183	11	7	41	5	17	16%	39	40%	24	42	40%	25

** sums of unattenuated loads adjusted for pond shore percentages

Note that the N Loads by input show the total nitrogen input to each sub-watershed. However, the fresh ponds sub-watersheds typically contribute to more than 1 sub-embayment and therefore their sub-watershed contribution if apportioned based upon the proportion of shoreline. The total nitrogen load to each fresh pond (e.g. # 4, 5, 6, 7, 10) are adjusted to their contribution to the receiving sub-embayment in both the "roll-ups" for each sub-embayment and in the present and build-out Nitrogen Loading (right sections of table)



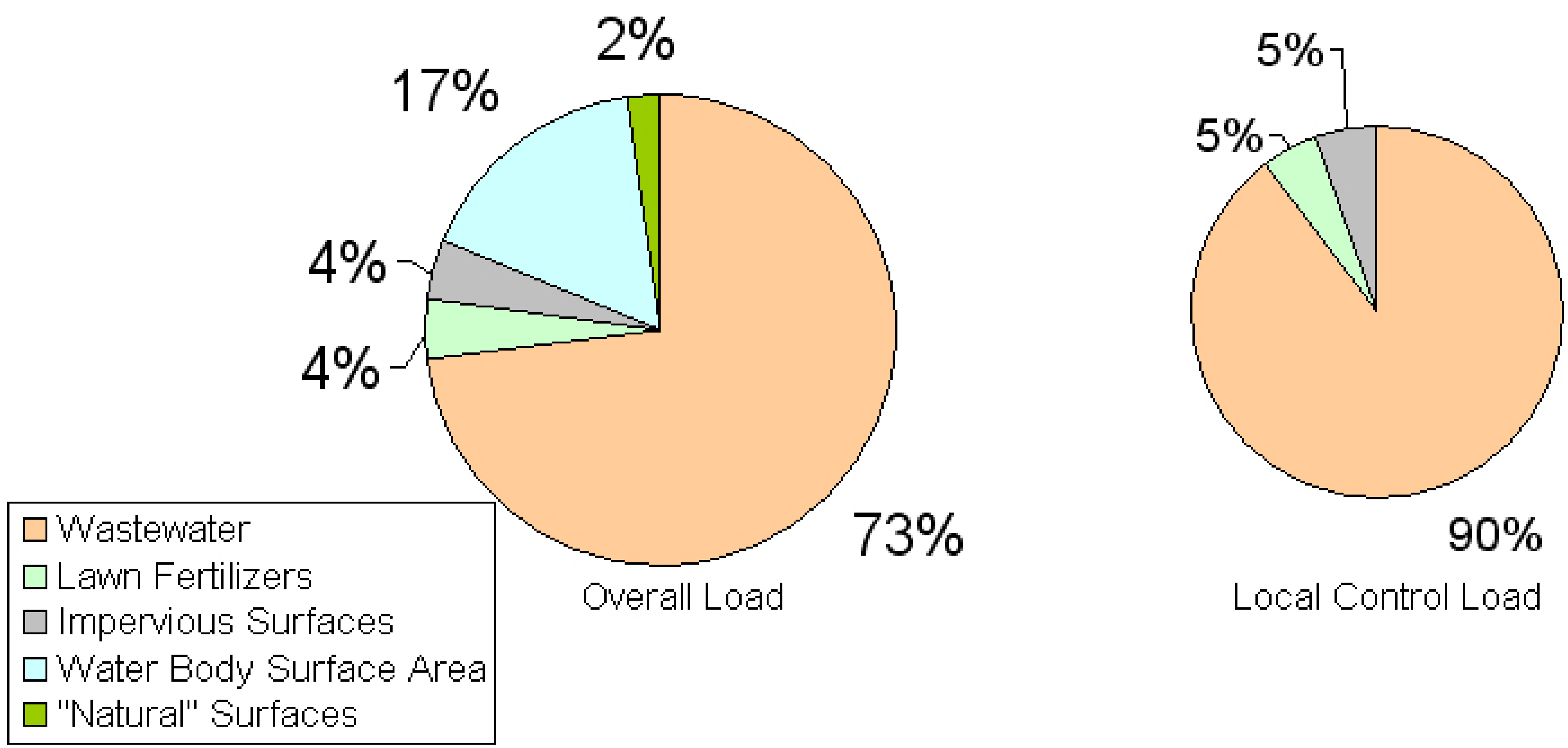


Figure IV-7a. Land use specific unattenuated watershed based nitrogen load (by percent) to Bassing Harbor embayment system.

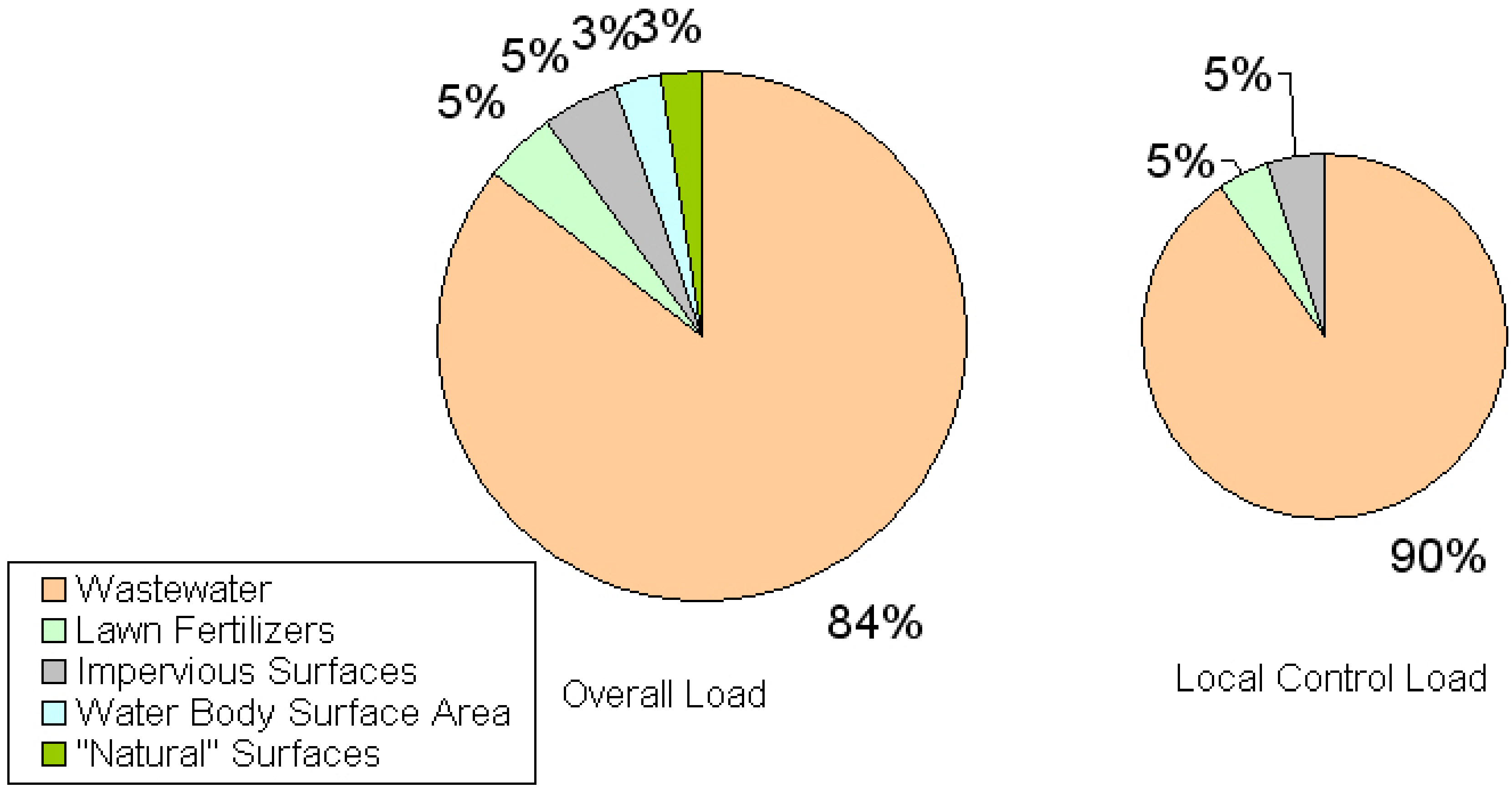


Figure IV-7b. Land use specific unattenuated watershed based nitrogen load (by percent) to Muddy Creek embayment system.

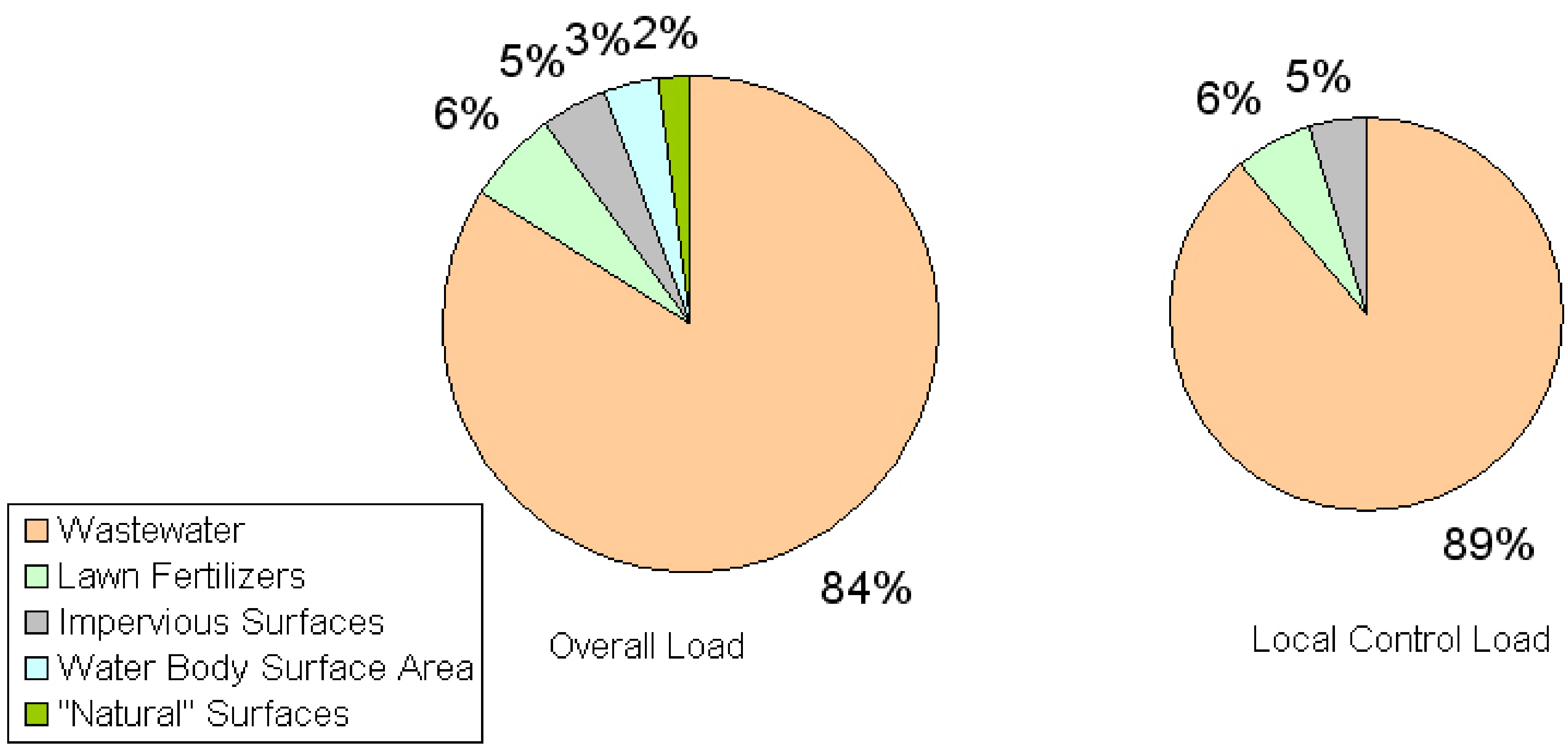


Figure IV-7c. Land use specific unattenuated watershed based nitrogen load (by percent) to Taylors Pond embayment system.

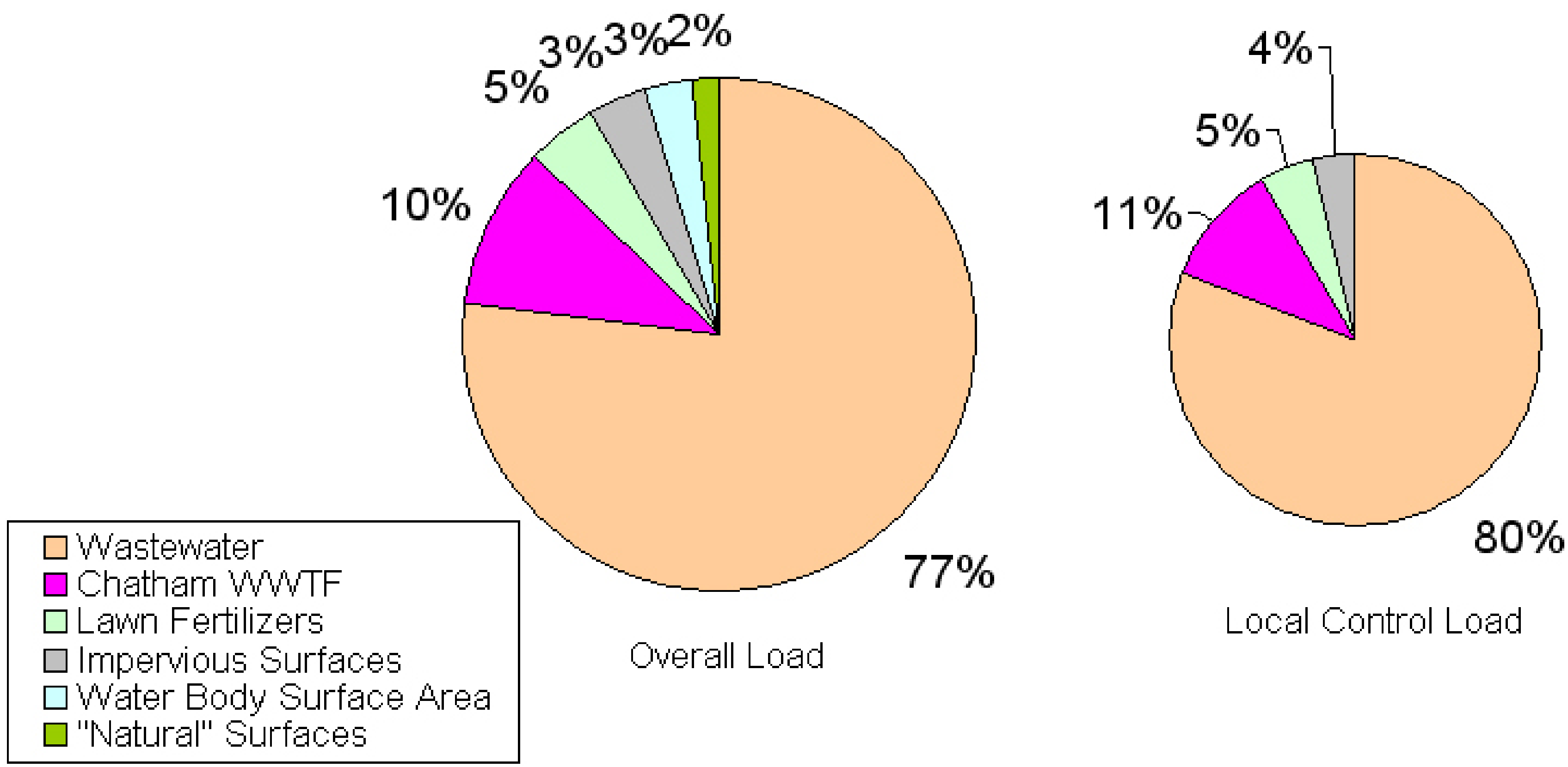


Figure IV-7d. Land use specific unattenuated watershed based nitrogen load (by percent) to Sulphur Springs embayment system.

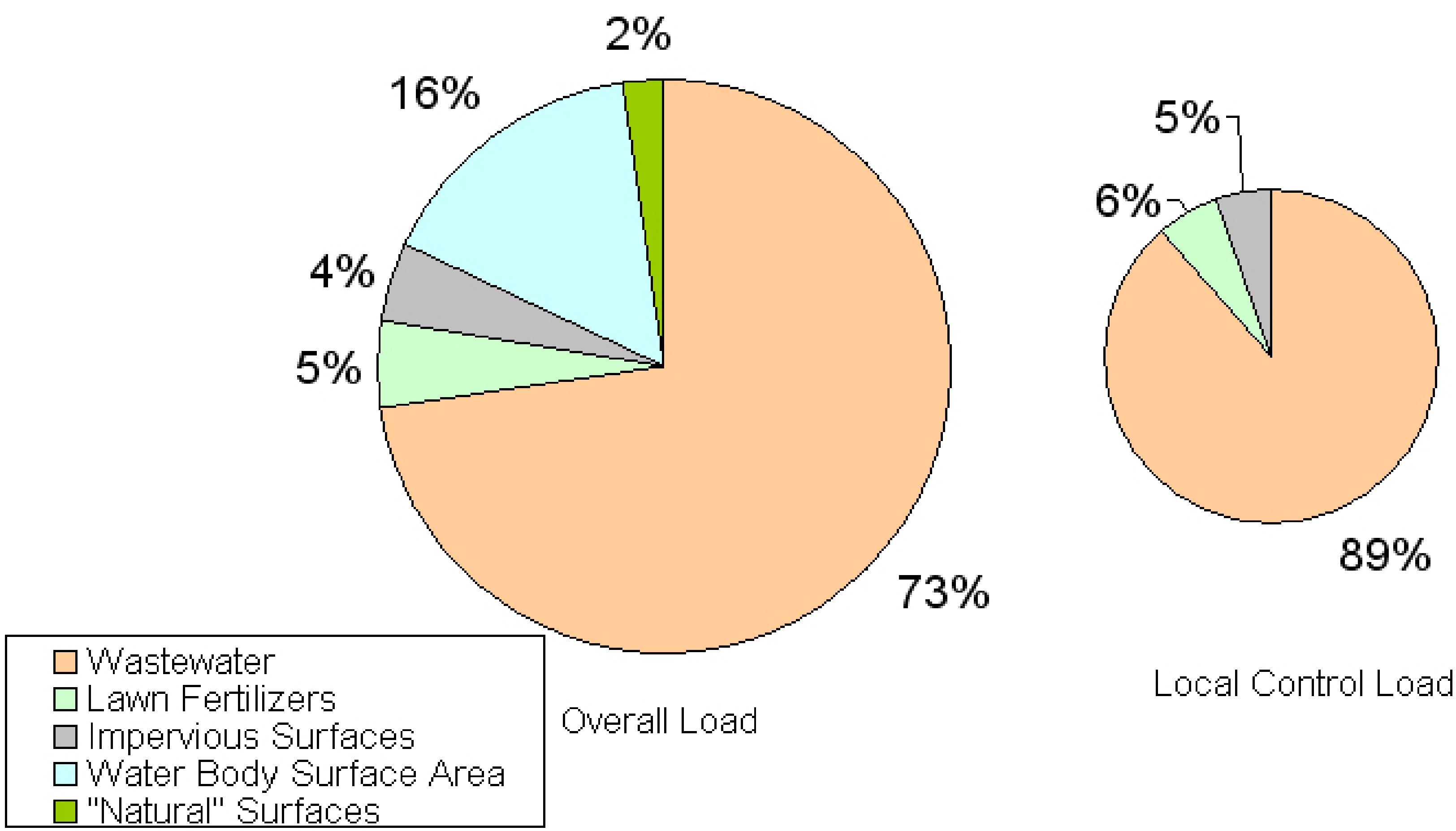


Figure IV-7e. Land use specific unattenuated watershed based nitrogen load (by percent) to Stage Harbor embayment system.

the turnover time, how much of the nitrogen is returned to the aquifer through the downgradient discharge of pond water was determined. In ponds with homothermic water columns, the nitrogen mass within the pond was based on the entire water volume.

Table IV-4 summarizes the pond attenuation estimates calculated from land-use modeled nitrogen inflow loads and nitrogen loads which appear to be recharged to the downgradient aquifer or to outflow streams from each pond based on pond characteristics and measured nitrogen levels. Nitrogen attenuation within these ponds appears to vary between 39 and 95%. However, a caveat to these attenuation estimates is that they are based upon nitrogen outflow loads from summer water column samples, and are not necessarily representative of the annual nitrogen loads that are transferred downgradient. More detailed studies of other southeastern Massachusetts freshwater systems including Ashumet Pond (AFCEE 2000) and Agawam/Wankinco River Nitrogen Discharges (CDM 2001) have supported a 40% attenuation factor. Within the Chatham study area, the pond outflows from Lovers Lake allowed a more detailed analysis (Section IV.2) of nitrogen attenuation in this system and attenuation was found to be 52% of total nitrogen input (watershed + atmosphere). This factor is also consistent with the freshwater pond attenuation factors used for the nitrogen balance for Great, Green and Bournes Ponds (embayments) in the Town of Falmouth (Howes and Ramsey 2001).

Table IV-4. Nitrogen attenuation by Chatham Freshwater Ponds based upon late summer 2001 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. Stillwater Pond and Lovers Lake had annual nitrogen and discharge measurements to determine attenuation; only Lovers Lake has full discharge through surface water flow, which yielded an attenuation of 52% (Table IV-5). The MEP Linked N Model uses a value of 40% for the non-stream discharge systems.

Pond	ID	Area acres	Total Depth m	Overall turnover time yrs	N Load Attenuation %
Emery	CH-491	14.11	6.2	3.5	39%
Goose	CH-458	41.25	11.0	8.7	90%
Lovers	CH-428	37.73	9.6	2.9	69%
Mill	CH-440	23.45	2.8	0.3	95%
Schoolhouse	CH-463	22.78	13.2	9.4	93%
Stillwater	CH-396	18.71	13.8	1.3	65%
White	CH-516	40.53	16.2	7.1	88%
Trout	CH-425	4.88	4.8	0.3	94%
Newty	CH-522	5.47	1.7	0.9	81%
				Mean	79%
				s.d.	19%

Since groundwater outflow from a pond can enter more than one down gradient sub-watershed, the length of shoreline on the down gradient side of the pond was used to apportion the attenuated nitrogen load to respective down gradient watersheds. The apportionment was based on the percentage of pond discharging shoreline bordering each down gradient sub-watershed. The percentages of shoreline are shown in Table IV-3 N Load Summary.

Buildout

In order to gauge potential future nitrogen loads resulting from continuing development, the potential number of residential, commercial, and industrial lots within each subwatershed was determined from the GIS database. Assessment began with the state class codes to determine all parcels that are classified as developable: residential land use codes 130 and 131, commercial codes 390 and 391, and industrial codes 440 and 441 (Figure IV-8). Existing zoning maps from the Towns of Chatham and Harwich (Figure IV-9) were then combined with the developable parcels through GIS. Build-out of parcels classified as developable were based on sub-divisions using minimum lot size within each zoning district. All municipal overlay districts (e.g., Districts of Critical Planning Concern, water resource protection districts) were considered in the determination of minimum lot sizes. A nitrogen load for each parcel was determined for the existing development using the factors presented in Table IV-2 and discussed above. A summary of potential additional nitrogen loading from build-out is presented as unattenuated and attenuated loads in Table IV-3.

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 5 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

Sub-Embayment	***ID#***
Mill Pond (Fresh)	1
Goose Pond	2
Trout Pond	3
Schoolhouse Pond	4
Stillwater Pond	5
Lovers Lake	6
Emery Pond	7
White Pond	8
Newty Pond	9
Bassing Pond	10
Lower Muddy Creek	11
Upper Muddy Creek	12
Mill Creek	13
Taylors Pond	14
Cockle Cove	15
Bucks Creek	16
Sulfur Springs	17
Stage Harbor	18
Lower Oyster River	19
Oyster River	20
Oyster Pond	21
Mitchell River	22
Mill Pond	23
Little Mill Pond	24
Ryders Cove	25
Crows Pond	26
Bassing Harbor	27
Frostfish Creek	28
Upper Frostfish Creek	29

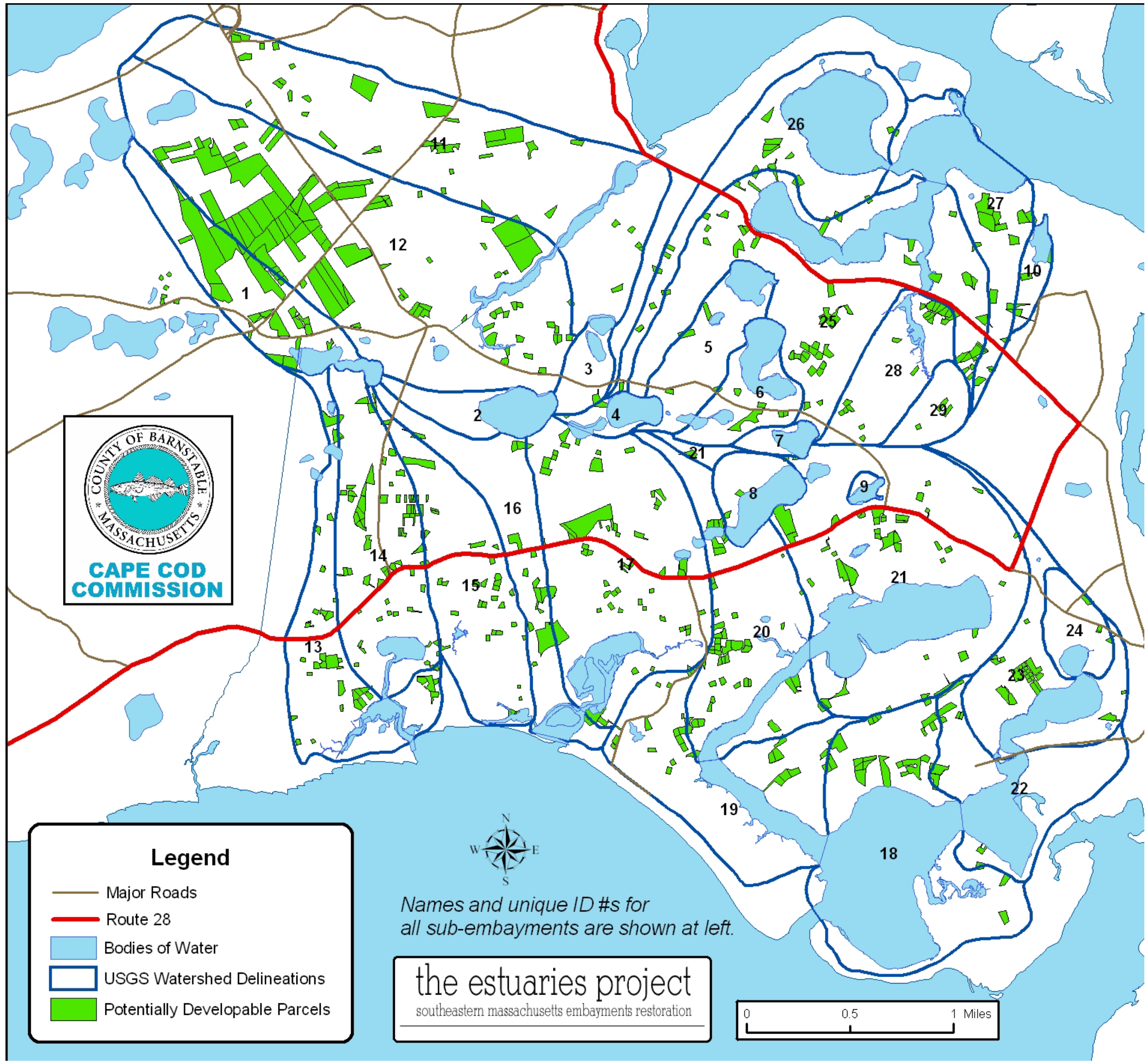


Figure IV-8. Distribution of present parcels which are potentially developable within the watersheds to the 5 embayment systems.

Sub-Embayment	ID#
Mill Pond (Fresh)	1
Goose Pond	2
Trout Pond	3
Schoolhouse Pond	4
Stillwater Pond	5
Lovers Lake	6
Emery Pond	7
White Pond	8
Newty Pond	9
Bassing Pond	10
Lower Muddy Creek	11
Upper Muddy Creek	12
Mill Creek	13
Taylors Pond	14
Cockle Cove	15
Bucks Creek	16
Sulfur Springs	17
Stage Harbor	18
Lower Oyster River	19
Oyster River	20
Oyster Pond	21
Mitchell River	22
Mill Pond	23
Little Mill Pond	24
Ryders Cove	25
Crows Pond	26
Bassing Harbor	27
Frostfish Creek	28
Upper Frostfish Creek	29

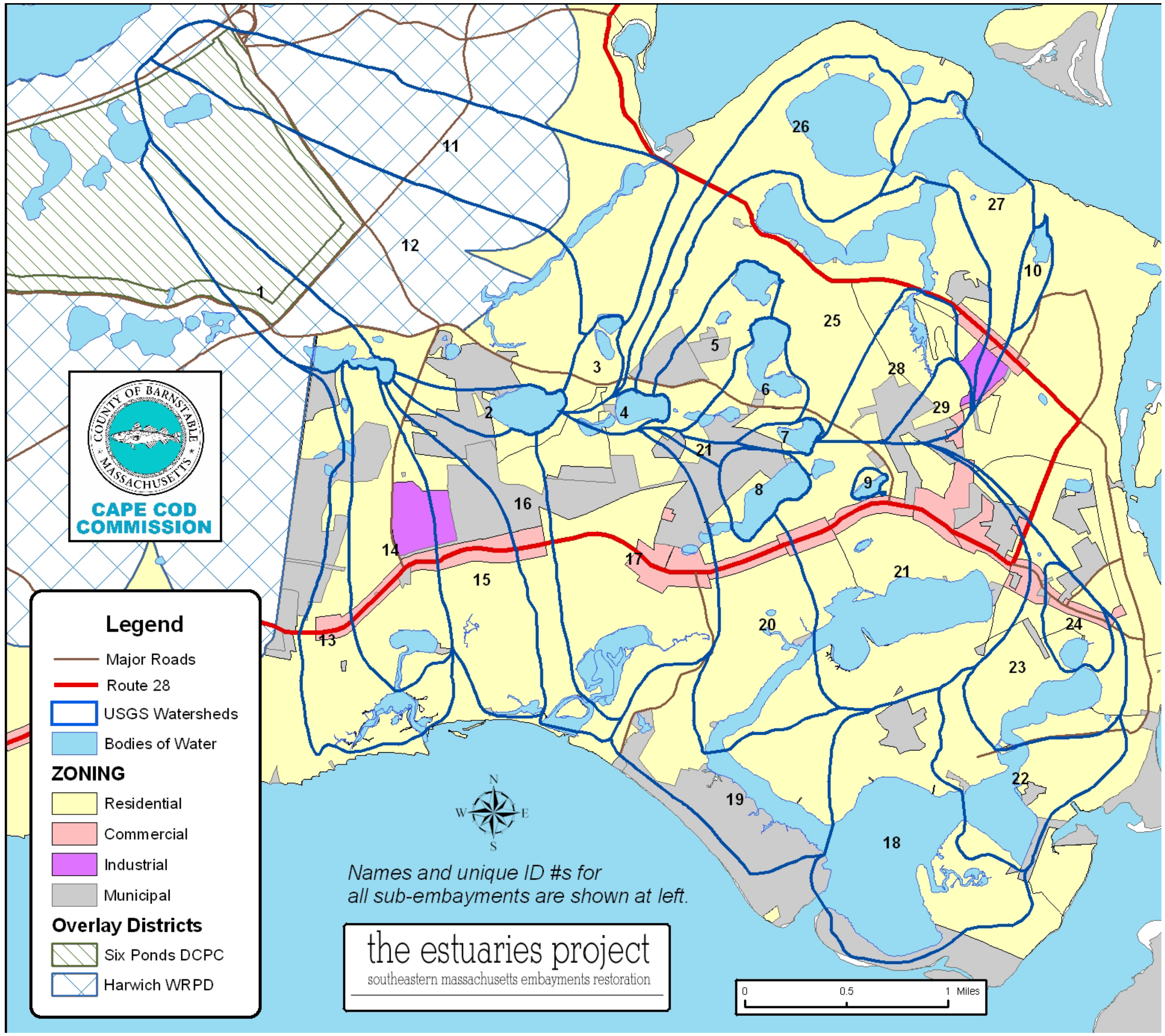


Figure IV-9. Spatial coverage of existing zoning and special overlay districts (DCPC, WRPD) in Harwich and Chatham, MA.

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 (Stillwater Pond) and nitrogen modeling indicated that nitrogen attenuation was likely in the Cockle Cove sub-watershed and associated with the Bassing Harbor System.

In the previous watershed loading studies the watershed delineation's were made by the Cape Cod Commission by surveying watertable elevations in available wells. While this is a powerful approach, it is limited by the distribution of existing wells. A review of the watershed delineation's by the Project Team and 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. The USGS re-delineation effort is described above in Section III. 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 5 embayment systems in the study. Natural attenuation by fresh kettle ponds was addressed above. However, additional site-specific studies were conducted in each of the major pond and marsh systems which have significant streams (Lovers Lake and Stillwater Pond discharge to Ryder Cove) or tidal exchanges (Frost Fish Creek). In addition, a screening approach was applied within Stage Harbor, and Cockle Cove Systems (Section IV.2.4.).

Quantification of watershed based nitrogen attenuation is contingent upon being able to compare nitrogen load to the embayment system directly measured in freshwater stream flow (or net tidal outflow) to nitrogen load as derived from the detailed land use analysis (Section IV.1). The development of a nitrogen attenuation term for freshwater transport through streams prior to discharge to marine waters was undertaken on two of the more significant surface water features in the Town of Chatham. Flow was measured at three different surface water locations (Figure IV-10) for their nitrogen loading and attenuation effects on Ryder Cove (creek between Lovers Lake and Stillwater Pond, creek between Stillwater Pond and Ryder Cove) and Bassing Harbor (Frost Fish Creek). Stage (water depth in the creek or stream) was monitored continuously for 16 months in the outflow streams from Lovers Lake to Stillwater Pond and Stillwater Pond to Ryder Cove (controlled outflow). Surface water flows in Frost Fish creek as well as nitrogen loading were measured and analyzed in order to refine the unique nitrogen

Benthic Coring Locations   Stream Gage Locations

Figure IV-10. Location of Stream gages and benthic coring locations in the Ryder Cove / Bassing Harbor System.

attenuation capacities of this system discharging to Bassing Harbor. Analysis of nitrogen attenuation resulting from biological processes in Frost Fish Creek was based on four separate tidal flux studies performed in July, August, and September 2002.

The Ryder Cove watershed was targeted because it contains surface water bodies which are generally associated with nitrogen attenuation and which previous studies (Applied Coastal Research and Engineering, 2001) indicated are subject to attenuation. In addition, rerouted outflows from Lovers Lake previously to Frost Fish Creek and Stillwater Pond and now to only to Stillwater Pond might provide a potential nitrogen management "soft solution" for Ryder Cove. Surface water samples were collected about weekly by the Chatham Water Quality Laboratory (R. Duncanson) and assayed by the SMAST Coastal Systems Analytical Laboratory.

In addition to the surface water field study within the Ryder Cove watershed, samples of surface water were collected by the water quality monitoring program from a variety of watersheds in order to screen watersheds for significant nitrogen attenuation of the watershed loading estimates. The watershed nitrogen loading and freshwater discharge estimates in this attenuation study were those derived in Section IV.1.

IV.2.2 Surface water Discharge and Attenuation of Watershed Nitrogen: Lovers Lake to Stillwater Pond to Ryder Cove

Lovers Lake and Stillwater Pond are 2 of the larger ponds within the study area and unlike many of the freshwater ponds, these have stream outflows rather than discharging solely to the aquifer on down-gradient shores. These stream outflows may serve to decrease their attenuation of nitrogen, but they also allow for a direct measurement of the nitrogen attenuation. Nitrogen attenuation was calculated in both Lovers Lake and Stillwater Pond from nitrogen loading rate estimates within respective watersheds and measured annual discharge of nitrogen through stream outflows of both ponds.

Stream gauging and nitrogen sampling stations were established within each of the two outflow streams, within the Ryder Cove sub-watershed. An upper station was placed at the discharge from Lovers Lake to Stillwater Pond and a lower station at the outlet of Stillwater Pond to Ryder Cove (Figure IV-10). The upper station was installed to evaluate results of the historical re-routing of discharge from Lovers Lake to Frost Fish Creek, as opposed to present discharge to Stillwater Pond. The lower station was to evaluate the surface water flow and nitrogen load to Ryder Cove from the sub-watersheds to Stillwater Pond + Lovers Lake + a portion of Schoolhouse Pond.

At each sampling site, a continuously recording vented water level gauge was installed and calibrated to yield the level of water in the discharge culvert that carries the flows and associated nitrogen load under roadways. Flow was periodically measured using a Marsh-McBirney electromagnetic flow meter. Periodic (~ weekly) water samples were collected for nitrogen analysis. These measurements allowed for the determination of both total volumetric discharge and nitrogen mass discharge to down-gradient systems. In addition, a water balance was constructed based upon the groundwater flow model to determine freshwater discharge expected at each gauge site. Comparison of measured and predicted discharge is used to confirm that the stream is capturing the entire recharge to its up-gradient contributing area. This comparison also can be used to indicate if pond outflow is through a combination of stream and groundwater outflow. This freshwater balance is necessary to support the attenuation calculations.

The gauges were installed on November 8, 2000 and were set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Due to multiple instrument failures during the period May 2001 to February, 2002, meaningful data was not collected. As a result, the field deployment period for the stream gaging was extended to include the summer 2002 field season. Water samples were collected approximately biweekly with an increase in sampling frequency to weekly during critical summer periods.

The stream gauge records available for this analysis of freshwater stream flow and associated attenuated nitrogen load covers a period of 361 days for the discharge to Ryder Cove and 470 days for the discharge from Lovers Lake to Stillwater Pond. The Ryder Cove gauge was damaged at 111 days and replaced to continue the long term recording of stage. Using the available flow measurements a composite year for each site was constructed from which annual and average daily freshwater flow from Lovers Lake to Stillwater Pond and from Stillwater Pond into Ryder Cove were determined (Figures IV-11, IV-12, Table IV-5). Both stream flow records show a similar seasonal pattern of high flow in spring and lowest flow during summer. This seasonal pattern reflects the annual variation of groundwater levels (Section IV.1), which is a major driver to streamflow in this hydrological setting. The nitrogen concentration measurements indicate the opposite pattern with higher levels in summer.

Total nitrogen concentrations within both streams outflows were relatively high, with Stillwater Pond outflow (0.851 mg N L^{-1}) higher than Lovers Lake outflow (0.732 mg N L^{-1}). This likely represents the higher nitrogen loading to Stillwater Pond (2465 g N d^{-1}) compared to Lovers Lake (1693 g N d^{-1}). In both streams, organic nitrogen forms dominated the total nitrogen pool, indicating that groundwater nitrogen (presumably dominated by nitrate) entering the ponds is taken up by plants within the pond system prior to export to the streams. However, nitrate was still a major fraction of the total nitrogen pool being 17% and 31% of the Lovers Lake and Stillwater Pond outflow nitrogen pools, respectively. The high concentration of inorganic nitrogen in the outflowing stream waters suggest that plant production within these ponds is not nitrogen limited. In the case of Stillwater Pond outflow water, the average nitrate concentration was >0.25 mg N L-1, representing a source of readily available nitrogen for stimulation of phytoplankton production within the receiving waters of Ryder Cove.

Annual flow measured within the Lovers Lake to Stillwater Pond stream agreed well (91%) with the predicted groundwater inflow to Lovers Lake from its watershed (Table IV-5). The slightly lower measured discharge likely results from the lower than average groundwater levels during the study period (Figure IV-4). From these data it appears that Lovers Lake discharges primarily through this stream. Therefore, the much lower nitrogen load (812 g N d^{-1}) discharged from Lovers Lake in this stream outflow relative to the nitrogen mass entering the Lake from its watershed (1693 g N d^{-1}) should be a direct measure of nitrogen attenuation by the pond ecosystem. Therefore, rate of natural attenuation of nitrogen moving through Lovers Lake is 52%, within the 39%-95% range determined from the pond survey method (see above) and consistent with use of a 40% attenuation factor for the survey ponds.

It should be noted that the discharge from Lovers Lake to Stillwater Pond, being the sole surface water drain for Lovers Lake, is a relatively recent phenomenon. The historic discharge from Lovers Lake was to both Stillwater Pond and to Frost Fish Creek (note 1943 USGS Topographic Map). However, one of the outflows, from Lovers Lake to Frostfish Creek, was discontinued since the 1980's (Duncanson, personal communication). This shift in outflow from Lovers Lake, increased the freshwater flow through and therefore decreased the residence time of water within Stillwater Pond (although the extent is currently unknown). This decreased residence time in Stillwater Pond, likely reduces the level of nitrogen attenuation. The effects of restoring the historic dual flow paths on distribution and total load to upper and lower Ryders Cove and the potential for increased nitrogen removal in passage through Stillwater Pond and Frost Fish Creek should be considered by the Town as it develops nitrogen management alternatives for the Bassing Harbor System. In this evaluation, it should be considered that outflow from Lovers Lake could be seasonally shifted between Stillwater Pond and Frost Fish Creek to maximize natural attenuation to “relocate” the site of nitrogen input to the estuary, while still providing for herring migration. While any such analysis must take into account existing aquatic uses of the fresh and saltwater systems being modified, it should be noted that the Frost Fish Creek system is primarily salt marsh with a relatively high salinity and that the flow change is not expected to shift this saltwater system significantly.

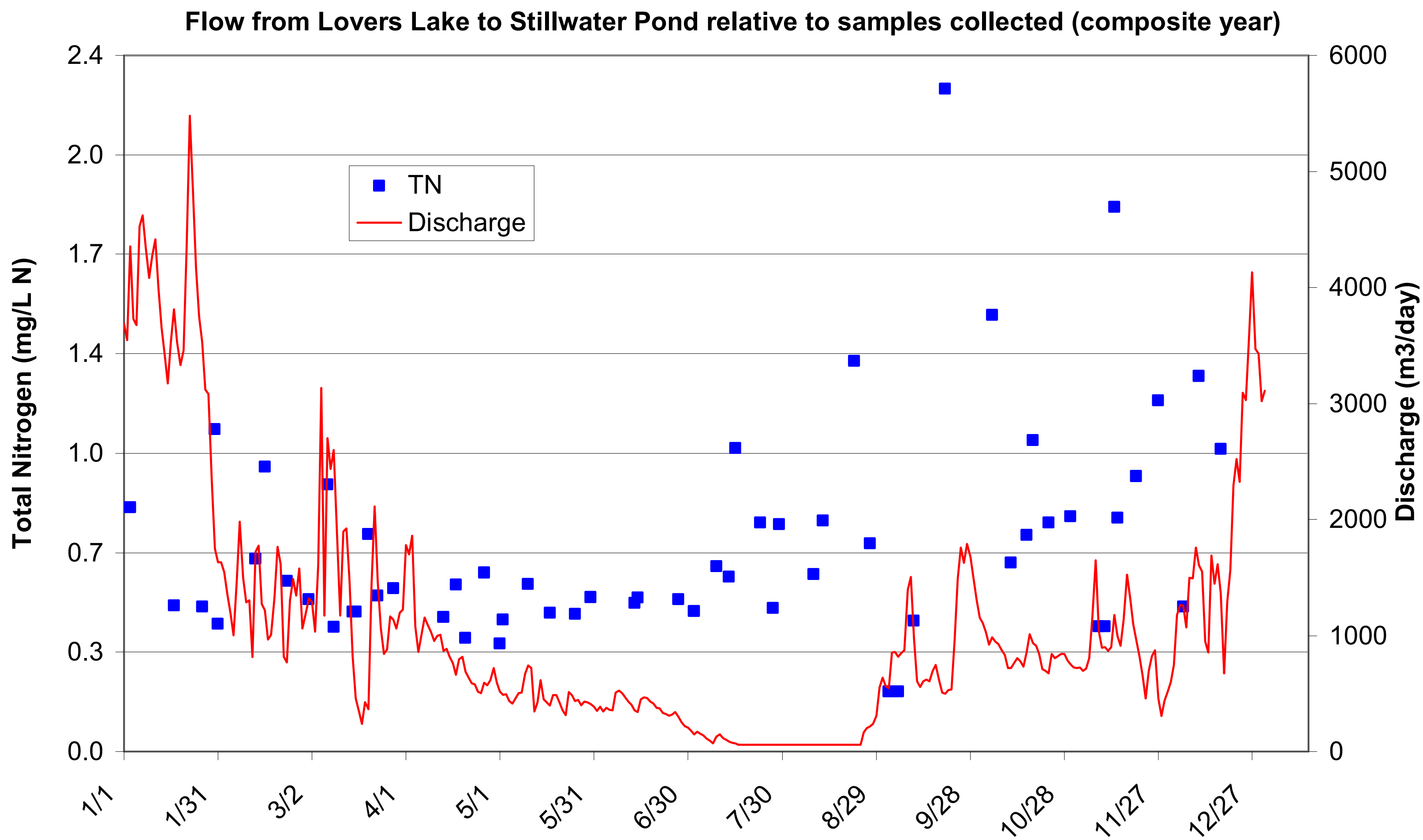


Figure IV-11. Annual composite developed from a stream gauge maintained in the outflow stream from Lovers Lake discharging to Stillwater Pond. Nutrient samples were collected approximately weekly and analyzed for inorganic and organic nitrogen species. These data were used to determine both annual flow and total nitrogen transport for determining nitrogen attenuation (see Table IV-5).

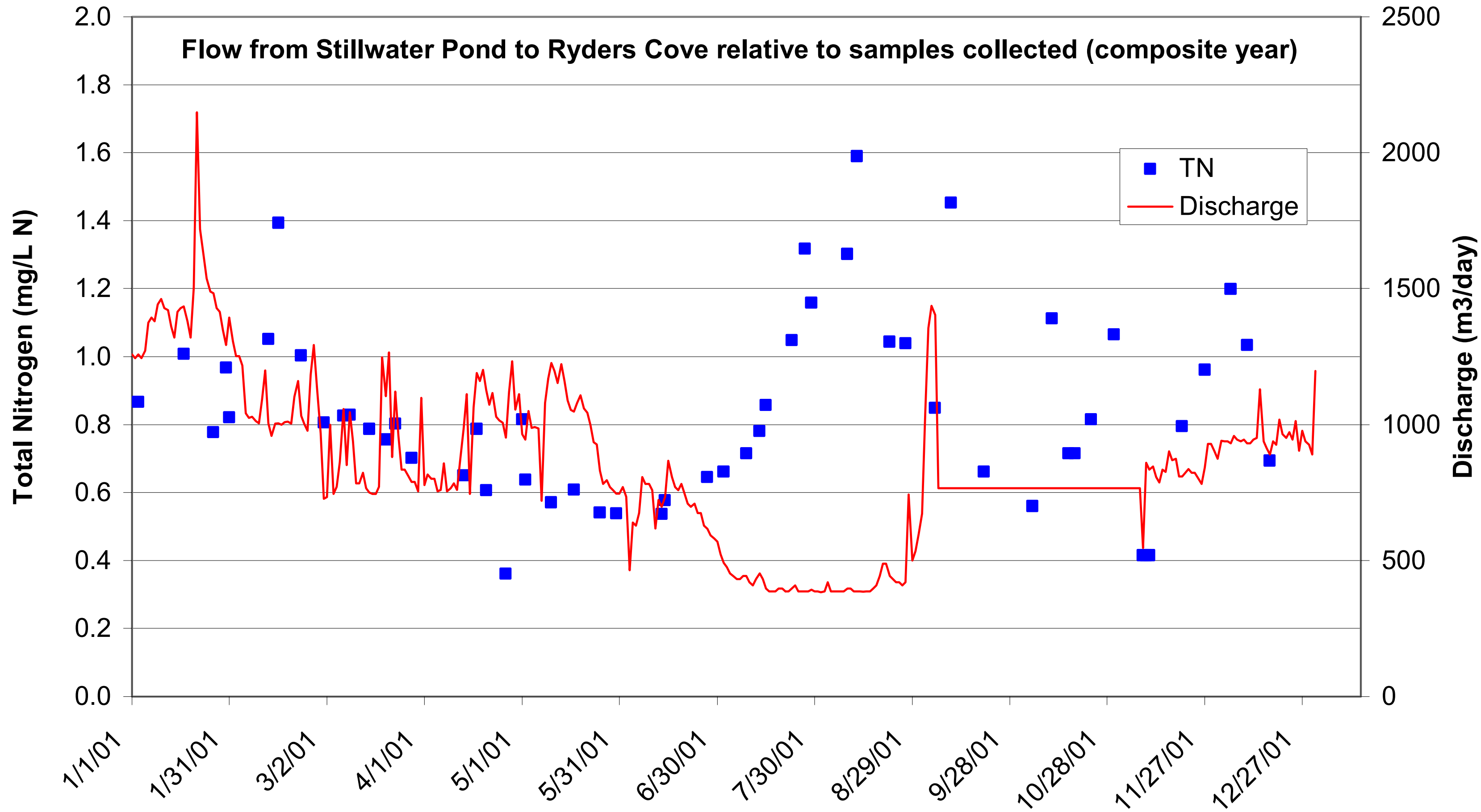


Figure IV-12. Annual composite developed from a stream gauge maintained in the outflow stream from Stillwater Pond discharging to Ryders Cove. Nutrient samples were collected approximately weekly and analyzed for inorganic and organic nitrogen species. These data were used to determine both annual flow and total nitrogen transport for determining nitrogen attenuation (see Table IV-5).

In contrast to Lovers Lake, the annual flow measured at the stream outflow from Stillwater Pond suggested that only a portion of the groundwater (and nitrogen) inflows from the watershed and Lovers Lake were exiting via the stream (34%). In fact less water was outflowing via Stillwater Pond stream (853 m^3 d^{-1}) than entering from Lovers Lake (1079 m^3 d^{-1}). In previous preliminary investigation at this site, there was concern that the lower than predicted flows from Stillwater Pond might result from an underestimate of the watershed area (Applied Coastal 2000). This does not appear to be the cause in the present case (even the Lovers Lake inflow is greater than Stillwater outflow). The most likely explanation for this observed water imbalance is that the elevation of the outflow weir from Stillwater Pond results in pond water outflow to the aquifer on the down-gradient shore, as in kettle ponds without stream outflows. In this case it is still possible to estimate nitrogen attenuation by Stillwater Pond. By correcting the nitrogen outflow relative to the proportion leaving via the stream and assuming that the outflowing groundwater has the same nitrogen concentration as the streamwater (conservative estimate), the total mass leaving the pond can be determined. This total discharging nitrogen mass when compared to the predicted watershed nitrogen inflow yields an attenuation of 14% for Stillwater Pond. If it is further assumed that lower groundwater levels are causing lower flows and the ratio from Lovers Lake (0.91) is used to adjust the predicted flow rate, then the calculated attenuation factor rises to 23%. These relatively low nitrogen attenuation rates may result from the relatively high nitrogen load to this system which enters from Lovers Lake, Schoolhouse Pond watershed and the adjacent Stillwater Pond watershed. The high nitrate levels in the outflowing water appear to support a lower attenuation rate for this pond. Given the uncertainties due to the hydrologic balance, the attenuation rate for this system should be considered to be a minimum.

IV.2.3 Freshwater Discharge and Attenuation of Watershed Nitrogen: Frost Fish Creek

Frost Fish Creek (above the Rt. 28 culverts) is a tidal basin with fringing salt marsh (see also Section V for hydrodynamics). Given its tidal flow, continuous stream gauging could not be conducted in the Frost Fish Creek discharge to the Bassing Harbor system. Instead, intensive discrete tidal flux analyses were conducted on four separate occasions (Summer 2002) in order to quantify freshwater inflow to Frost Fish Creek and nitrogen attenuation by this tributary system to Bassing Harbor.

Freshwater and tidal flows were measured over complete tidal cycles. Direct flow measurements were made at the weir near the mouth of Frost Fish Creek (Figure IV-10) combined with high frequency (hourly during ebb and flood, every half hour around the turn of each tide) water quality sampling for nutrients. The combination of both records allowed for the calculation of nitrogen load into and out of the embayment for each of the four tidal periods analyzed in July (1), August (2), and September (1) of 2002. Comparison of measured nitrogen loads resulting from the freshwater fraction of the Frost Fish Creek flow enabled the calculation of a nitrogen attenuation term applicable to the calculated watershed based nitrogen loads for the Frost Fish Creek sub-watershed.

Each of the tidal flux studies performed on Frost Fish Creek were completed over a complete tidal cycle, beginning approximately one hour prior to low tide and continuing through the high tide, ending approximately one hour past the time of the following low tide. The tidal flux studies were conducted with at least two days of no precipitation such that flow measurements, water quality sampling and subsequent nitrogen loading calculations would not be biased by storm related flows.

Table IV-5. Comparison of water flow and nitrogen discharges to Ryder Cove and from School House Pond, Lovers Lake and Stillwater Pond watershed through Stillwater Pond Stream. The "Stream" data is from previous SMAST studies with the Town of Chatham and the MEP stream gauging effort. Watershed data is based upon the MEP watershed modeling effort by USGS.

Stream Discharge Parameter	Stream flow to Ryder Cove	Steam flow into Stillwater Pond	Data Source
Total Days of Record [a]	361	470	(1)
Flow Characteristics:			
Stream Average Discharge (m3/d)	853	1079	(1)
Contributing Area Average Discharge (m3/d)	2488[b]	1185[c]	(2)
Proportion Discharge Stream vs. Contributing Area (%)	34%	91%	
Nitrogen Characteristics:			
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.263	0.127	(1)
Stream Average Total N Concentration (mg N/L)	0.851	0.732	(1)
Nitrate + Nitrite as Percent of Total N (%)	31%	17%	
Stream Average Nitrate + Nitrite Discharge (g/d)	207	192	(1)
Stream Average Total Nitrogen Discharge (g/d)	717	812	(1)
Contributing Area Average Total Nitrogen Discharge (g/d)	2465	1693	(2)
Proportion Total Nitrogen Stream vs. Contributing Area (%)	N/A	48%	
Attenuation (Total) of Nitrogen in Pond/Stream (%)	14%*	52%	

[a] from 11/8/00 to September 2002 (Ryder gage) and December 2002 (Stillwater Pond gage)
[b] flow and N load to Stillwater Pond include Lovers Lake Contributing Area, with correction for low flow using Lovers Lake Outflow %
[c] flow and N load to Lovers Lake represent only the Lovers Lake Contributing Area
* attenuation based upon expected nitrogen in measured volume discharge.
N/A = data not available
(1) MEP data, collected Amendment to present study
(2) Calculated from MEP watershed delineations to School House Pond, Lovers Lake and Stillwater Pond; the fractional flow path from each sub-watershed which contribute to Stillwater Stream Flow; and the annual recharge rate.

All four of the Frost Fish Creek tidal flux studies were conducted at the weir/culvert just up-gradient of Route 28 in Chatham. This culvert separates the main body of Frost Fish Creek from a small impoundment that receives Frost Fish Creek flows prior to final discharge to the Bassing Harbor embayment. Ebb and flood tide velocities were all measured at the same end of the culvert and generally taken concurrently with the water quality samples. In the instances when velocities were obtained at slightly different times than the water quality sample taken, a linear interpolation was utilized to match a flood or ebb tide velocity with the appropriate time of the water quality sample. Completing the linear interpolation on velocity for the complete tidal period yield a detailed record of flow out and in (ebb/flood) that related directly to changes in tidal stage (Figures IV-13A-D) The tidal flux volume results for Frost Fish Creek served the dual purpose of being a means to quantify attenuation of watershed based nitrogen loading to Frost Fish Creek as well as cross check for the RMA-2 hydrodynamic model. With the exception of the tidal study conducted on July 21, 2002, modeled and measured tidal flux volumes differed by only 2 and 6 percent.

As described above, each nutrient water quality sample was paired with a flow rate such that nitrogen and other constituent fluxes in Frost Fish Creek could be calculated for each of the tidal cycles studied. Tidal volume for each study was determined over the period from ebb slack to flood slack tide (Flood) and from flood slack to ebb slack (Ebb). In cases where tidal asymmetry resulted in a change in the water volume stored within the Frost Fish Creek basin (during a tidal cycle), the appropriate flood or ebb interval (time) was adjusted to ensure a zero change in storage volume within the basin, by keeping the measured tidal elevation at the end of a study equal to that at the start. Net tidal flux volume for the system was then determined by the difference in total volume inflow versus outflow over a tidal cycle, positive (+) indicating a net inflow into the system on the flood versus a negative (-) a net discharge from the system (Table IV-6). Determining freshwater inflow to a basin from differences in inflow/outflow at the tidal inlet is an acceptable approach in cases like Frost Fish Creek, where changes in storage can be controlled and where the freshwater outflow is a large fraction of the total outflow volume (Millham and Howes 1994). In the present study, freshwater outflow represented about one-third of the total ebb tide volume, a very large proportion compared to the larger estuarine systems of Chatham.

The measurements of freshwater discharge to Frost Fish Creek from its watershed ranged from 1258 m^3d^{-1} to 900 m^3d^{-1}, with an average (1097 m^3d^{-1}) close to that predicted (1274 m^3d^{-1}) from the groundwater flow model (Section III). Given that measurements were conducted during the summer period when flows are lower than the annual average, the measured and modeled freshwater flows are in excellent agreement. This agreement supports a straight-forward determination of nitrogen attenuation for this system.

Nitrogen mass on each inflowing and outgoing tide was calculated from the tidal sampling data by integrating over the flood and ebb tides. A net nitrogen outflow from Frost Fish Creek to lower Ryder Cove was observed in each event (Table IV-6). In fact, Frost Fish Creek was a net exporter of each of the major nitrogen related water quality constituents assayed. These exports result from the inflow and biological transformation of watershed derived nitrogen in Frost Fish Creek. Nitrogen attenuation was determined as the difference between the predicted watershed nitrogen input (Section IV.1) and the observed net loss of nitrogen to lower Ryder Cove. Comparing the observed mean net nitrogen tidal export of 1.82 kg N $tide^{-1}$ and the predicted watershed nitrogen load of 2.24 kg N $tide^{-1}$, natural attenuation of watershed derived nitrogen within Frost Fish Creek is 19%. This is a lower attenuation rate than the 40% observed in the Mashapaquit Creek Marsh in the West Falmouth Harbor System (Howes and Smith 1999). However, the Frost Fish Creek basin results in a dilution of inflowing groundwater nitrogen which can reduce the rate of denitrification of externally derived nitrate. In Mashapaquit Creek, groundwater flow during ebb tide was directly over creekbottom sediments, enhancing nitrogen removal by denitrification. The lower rate in Frost Fish Creek compared to Mashapaquit Creek is consistent with differences in factors related to denitrification in the 2 systems. In summary, the mass of nitrogen entering lower Ryder Cove from Frost Fish Creek is approximately 19 percent lower than the nitrogen load calculated from the sub-watershed land use analysis (which have been adjusted accordingly for development of management alternatives).

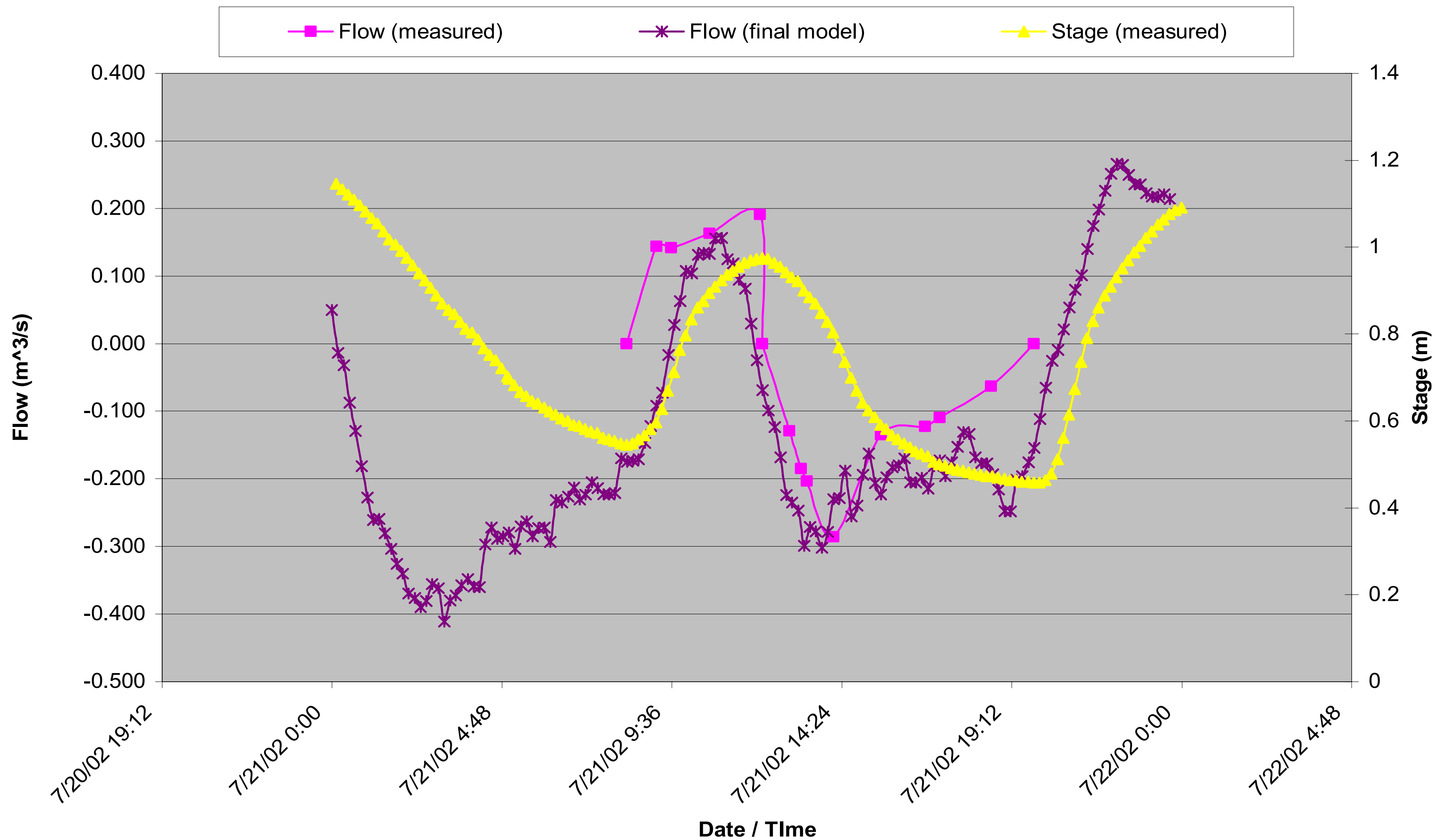


Figure IV-13a. Frost Fish Creek Tidal Study 1 (July 21, 2002). Comparison of measured and modeled tidal flow and measured tidal elevation.

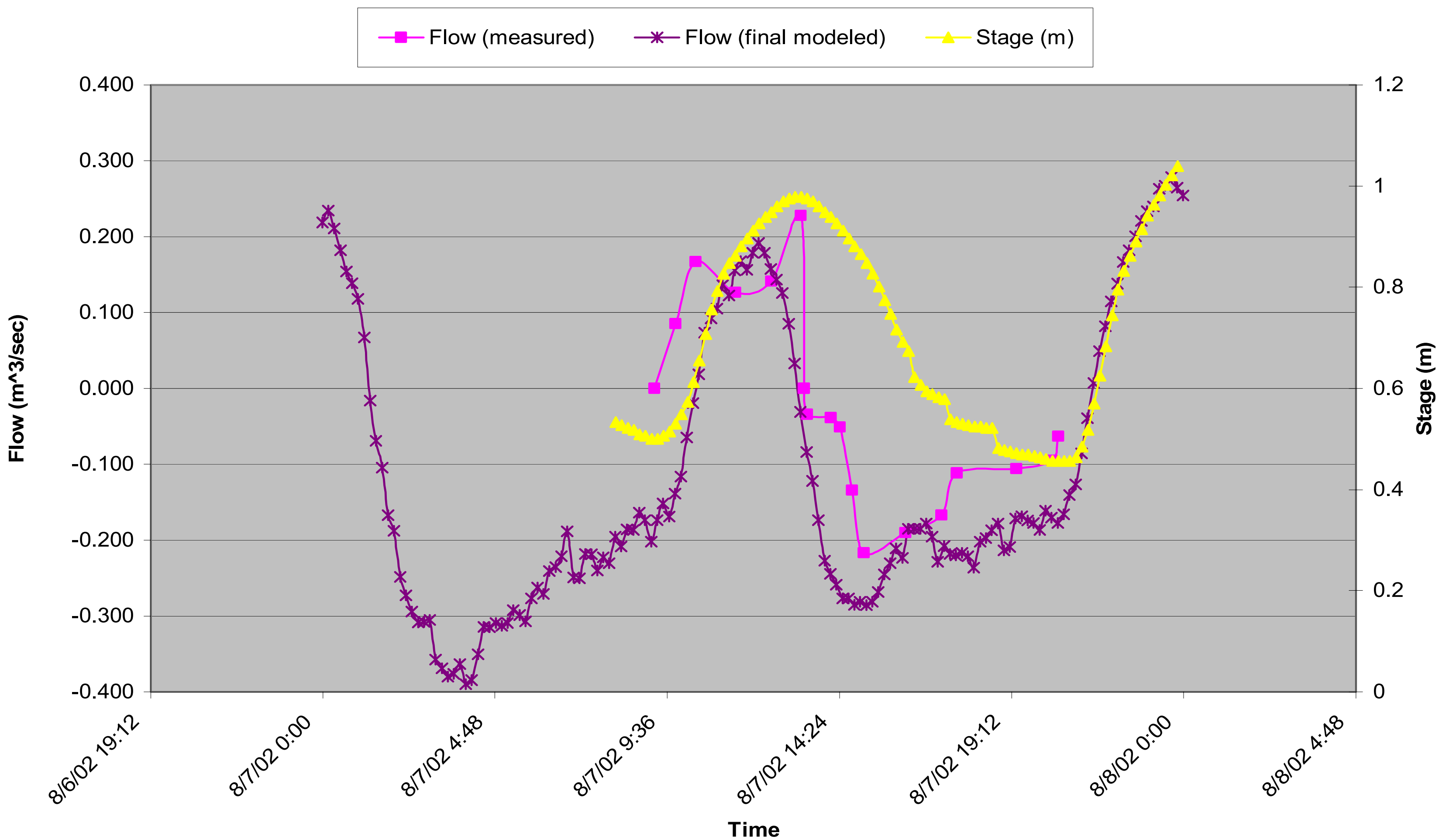


Figure IV-13b. Frost Fish Creek Tidal Study 2 (August 8, 2002). Comparison of measured and modeled tidal flow and measured tidal elevation.

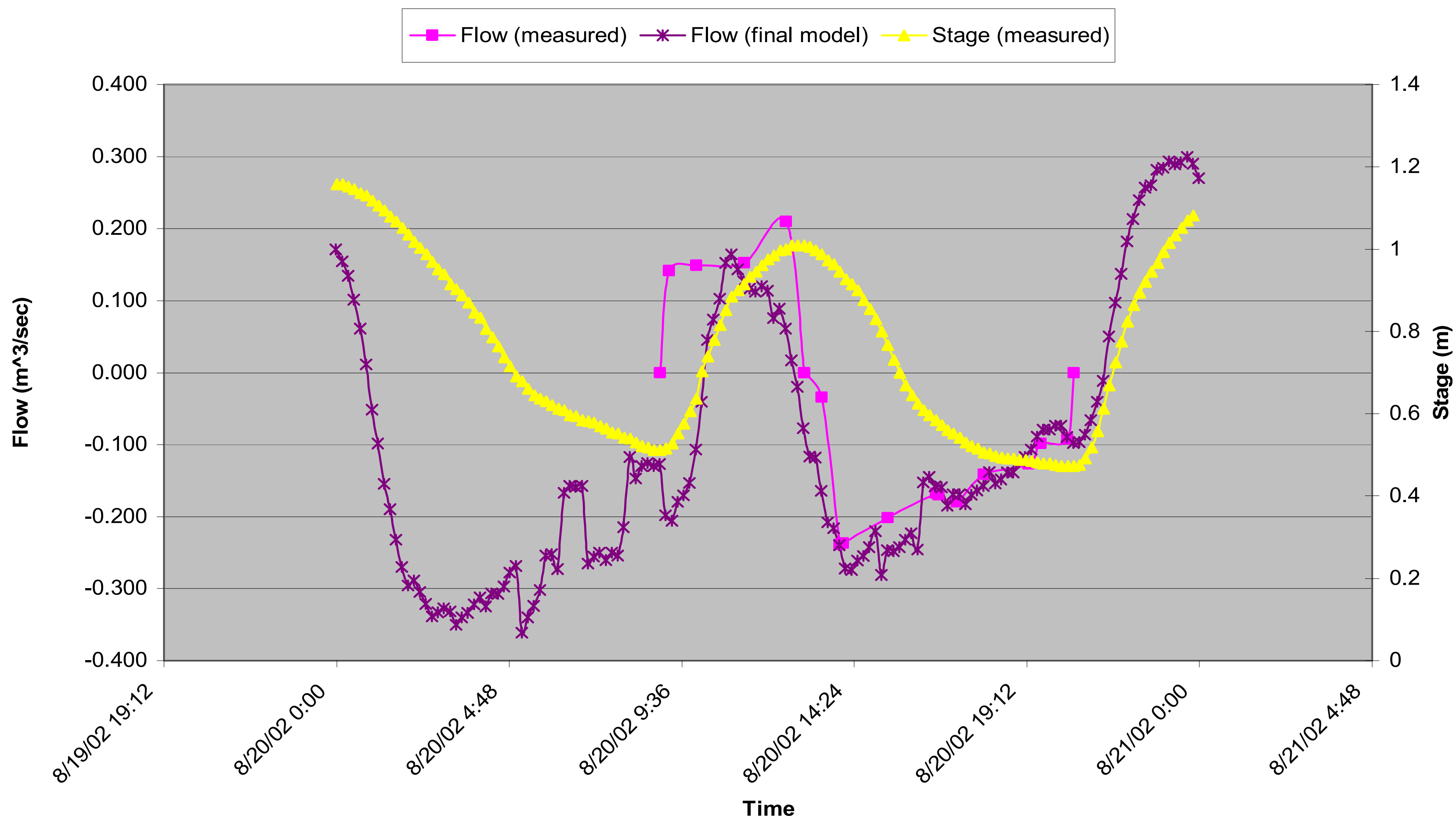


Figure IV-13c. Frost Fish Creek Tidal Study 3 (August 20, 2002). Comparison of measured and modeled tidal flow and measured tidal elevation.

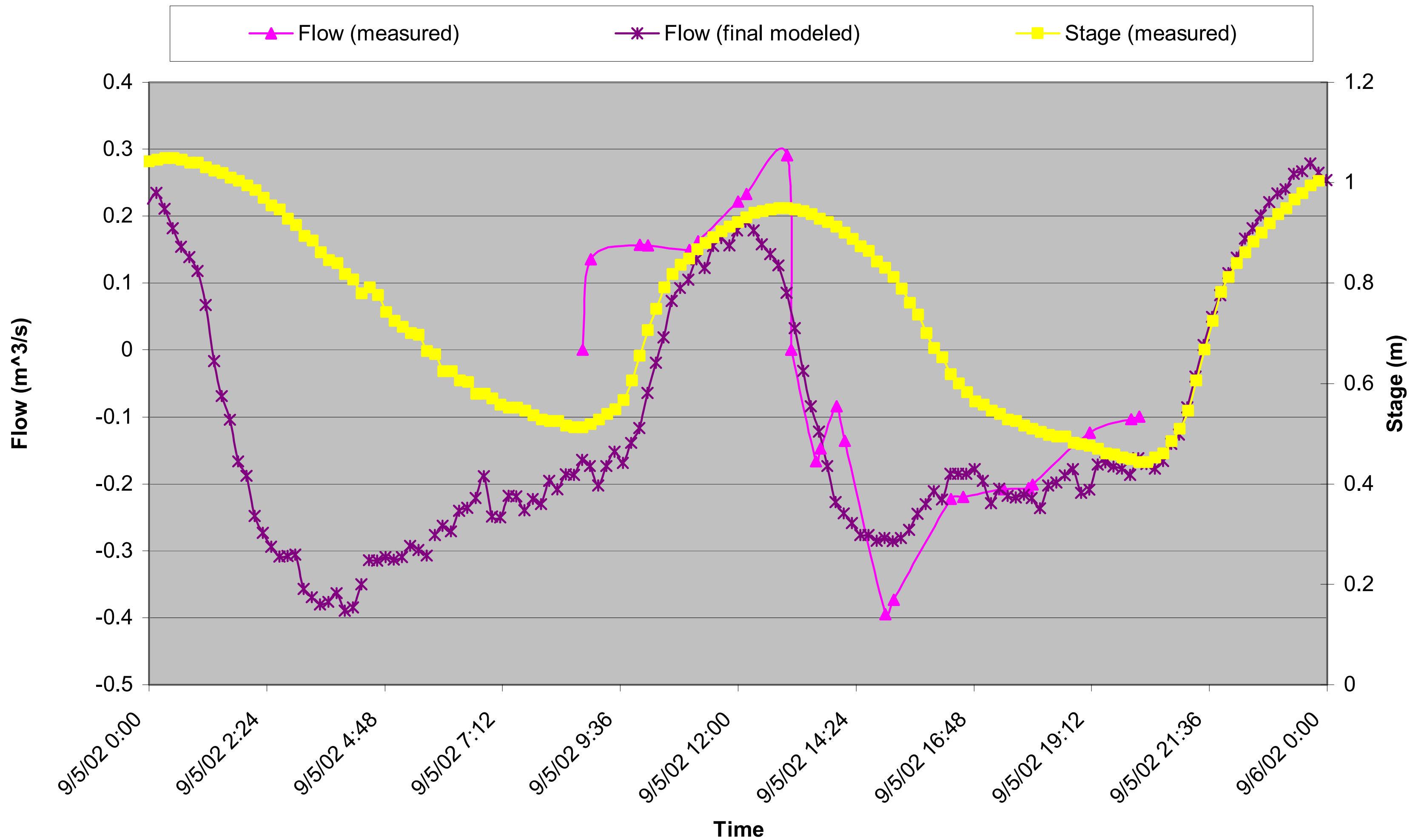


Figure IV-13d. Frost Fish Creek Tidal Study 4 (September 5, 2002). Comparison of measured and modeled tidal flow and measured tidal elevation.

Table IV-6. Measurement of nitrogen attenuation, flow and water quality constituents within Frost Fish Creek during summer 2002. The total freshwater discharge to Frost Fish Creek from the watershed as determined from the USGS groundwater model (Section III) was 1274 m^3 per day based upon the annual average, compared to the 1097 m^3 per day determined by the RMA-2 model (Section V) and the 1054 m^3 per day from the 4 Tidal Studies. Nitrogen attenuation is calculated as the difference in measured nitrogen mass in tidal outflow from Frost Fish Creek to Ryder Cove versus the nitrogen load entering from the watershed and within the inflowing tidal waters.

Study/Date	Tide	Tidal Flux RMA-2 Modeled m^3/day	Tidal Flux Measured m^3/day	NOX Kg N per tide	Total N kg N per tide	TON Kg N per tide	POC Kg C per tide	DIN Kg N per tide	Pigment g Pig per tide
Study 1	**Flood (+)**		1952	0.03	1.67	1.51	3.88	0.16	20.8
July 21, 2002	**Ebb (-)**		-2903	-0.88	-3.47	-2.51	-5.68	-0.96	-176.7
	Net Flux	**-1258**	**-951**	**-0.85**	**-1.81**	**-1.00**	**-1.80**	**-0.80**	**-155.9**
Study 2	**Flood (+)**		1999	0.04	2.57	2.47	7.88	0.10	44.1
August 7, 2002	**Ebb (-)**		-3222	-0.26	-5.16	-4.85	-15.93	-0.31	-92.7
	Net Flux	**-1155**	**-1223**	**-0.22**	**-2.59**	**-2.38**	**-8.05**	**-0.21**	**-48.6**
Study 3	**Flood (+)**		2128	0.04	1.97	1.88	6.46	0.09	52.6
August 20, 2002	**Ebb (-)**		-3019	-0.02	-2.99	-2.92	-10.32	-0.07	-93.0
	Net Flux	**-900**	**-891**	**0.02**	**-1.02**	**-1.04**	**-3.86**	**0.02**	**-40.4**
Study 4	**Flood (+)**		2756	0.18	5.86	3.39	6.64	2.47	45.1
September 5, 2002	**Ebb (-)**		-3906	-0.51	-7.71	-5.18	-11.97	-2.53	-105.9
	Net Flux	**-1075**	**-1150**	**-0.33**	**-1.85**	**-1.79**	**-5.34**	**-0.06**	**-60.7**
Mean Flux (N kg/tide)		**-1097**	**-1054**	**-0.34**	**-1.82**	**-1.55**	**-4.76**	**-0.26**	**-76.4**
S.E. (N kg/tide)		**75**	**79**	**0.18**	**0.32**	**0.33**	**1.32**	**0.19**	**26.8**
CV%		**-7%**	**-8%**	**-53%**	**-18%**	**-21%**	**-28%**	**-70%**	**-35%**
Total Land + Atmos. Inputs N kg/tide					**2.24**				
Attenuation (calculated)					**19%**				

IV.2.4 Confirmation of Watershed Nitrogen Discharge: Town-wide.

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. The data were collected by the Chatham Water Quality Laboratory at the sites shown in Figure IV-14. Water samples were collected primarily during the summer months from flowing surface waters. 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, 2.75 mg N/L (predicted) versus 1.6 - 3.1 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 1.95 and 1.40 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.

The apparent nitrogen attenuation within the Cockle Cove Creek system relative to predicted watershed nitrogen levels is likely due in part to stimulation of denitrification within this system. Measurements of nitrate uptake in Cockle Cove Creek made as part of the Sediment Nitrogen Regeneration Study (see below) indicated a large uptake by the Creek sediments. Additional data collection would have to be conducted in order to quantitatively determine the mass of nitrogen removed from the Creek System prior to discharge to Buck Creek. However, nitrogen attenuation by the tidal creek sediments is clearly demonstrated. Additional evaluation of Cockle Cove is relevant only to the nitrogen loading to the Bucks Creek System and macrophyte issues within the near shore region (Harding Beach area).

It appears that those embayment watersheds within Chatham that have significant surface water flows and water bodies have significant amounts of the watershed nitrogen load removed prior to discharge to the adjacent embayments.

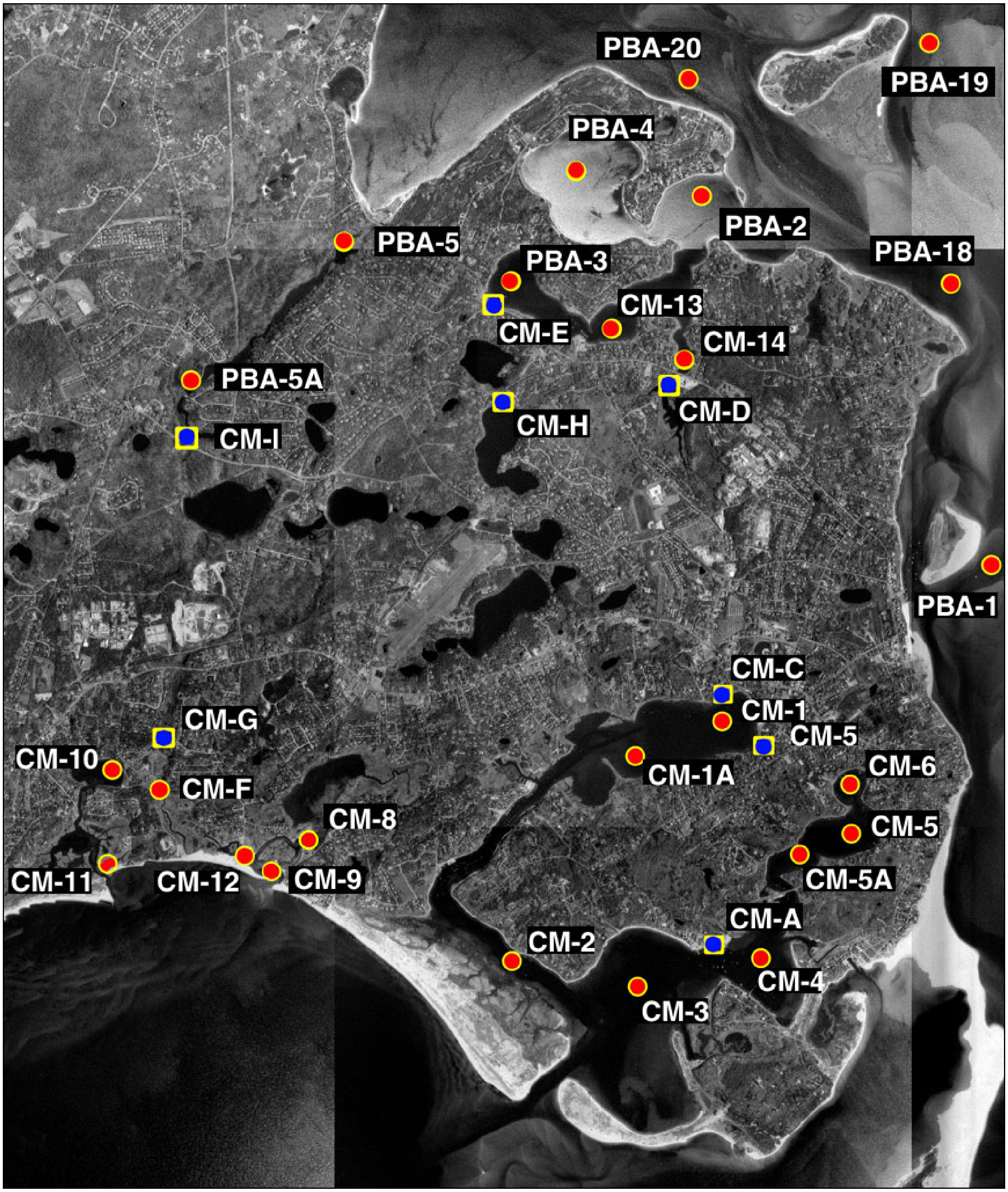


Figure IV-14. Map of freshwater discharge (blue squares) and estuarine (red circles) water quality monitoring stations. CM-E & H are the outflow from Stillwater Pond and Lovers Lake, respectively.

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 5 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 5 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 from 46 sites (Figure IV-15) were collected in late July 2000, with additional sampling of the Bassing Harbor Systems in 2001. 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.

Rates 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 5 embayments as follows: Stage Harbor System - 18 cores, Bassing Harbor System - 16 cores, Muddy Creek - 4 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 follow 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”.

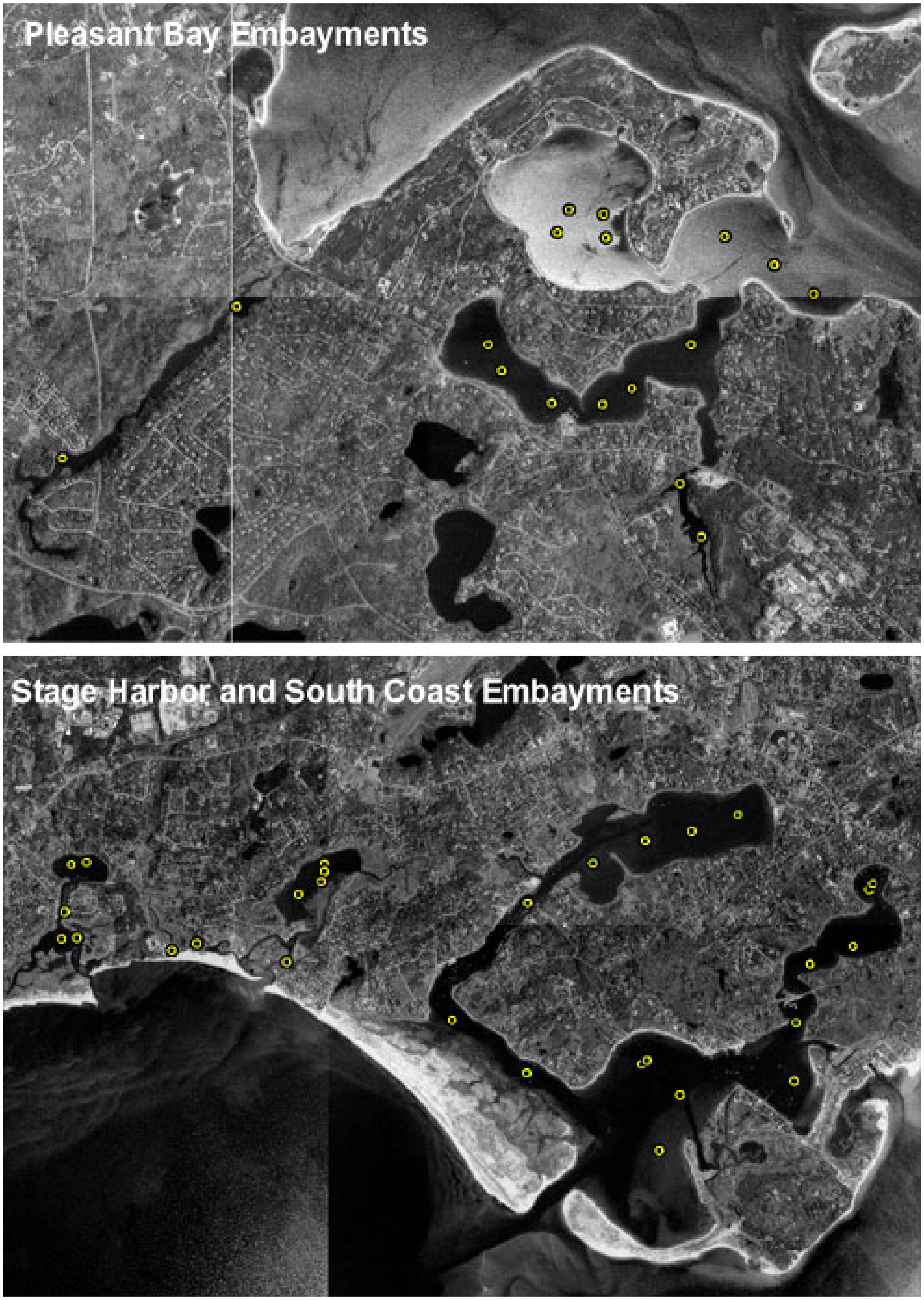


Figure IV-15. Chatham shoreline with locations of sediment core sampling stations shown as red filled circles. Some locations are sites of more than one sample. All sites were assayed in 2000 with Bassing Harbor having additional data collected in 2001.

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.

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-16).

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.

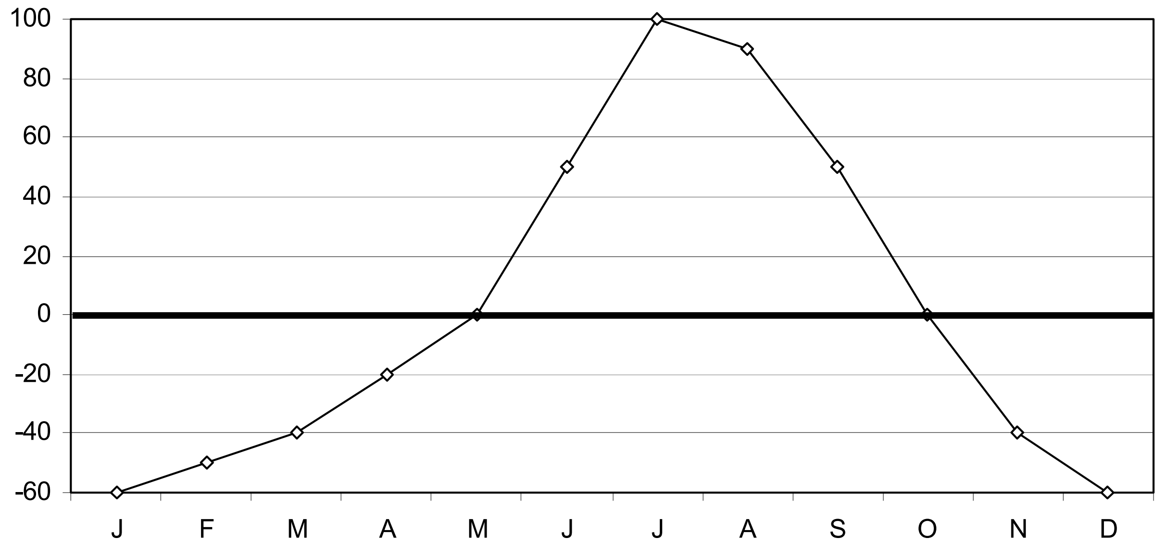


Figure IV-16. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

In order to obtain the net nitrogen balance of each embayments sediments, 48 cores were collected at 46 locations throughout the 5 embayments of Chatham (Figure IV-15). 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, by the Chatham Water Watchers and Pleasant Bay Alliance. 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-17). As depicted in figure IV-17, with the exception of Frost Fish Creek, 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 5 embayment systems used in the water quality modeling effort (Section VI) are presented in Table IV-7. There were concerns that the benthic regeneration rates measured in 2000 in the Bassing Harbor System were anomalously high (cf. Applied Coastal 2001), so the system was re-assayed in 2001. Evaluation of the 2000 and 2001 rates indicated that while the oxygen uptake rates were similar (<10% overall) in both years, the rates of nitrogen release in each of the sub-embayments were significantly higher in 2000 versus 2001. Additionally, the 2000 nitrogen release rates where higher than any observed rate in other systems. In contrast, the 2001 data shows nitrogen releases in line with other systems and oxygen uptake to nitrogen release ratios similar to other coastal systems (i.e. about 2 times the Redfield Ratio of 6.7 compared to 1.1 times in 2000). Only the 2001 data for the Bassing Harbor System was used. 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-7. Rates of net nitrogen return from sediments to overlying waters based on sub-embayment area coverage and core flux measurements.

Sub-embayment	Net N Efflux (kg/day)	Standard Error
Stage Harbor		
Oyster Pond	26.8	3.4
Oyster River	0.7	0.9
Stage Harbor	12.8	2.9
Mitchell River	-3.4	2.9
Mill Pond	3.7	1.6
Little Mill Pond	2.0	0.5
Sulphur Springs		
Sulphur Springs	-3.6	1.1
Bucks Creek	2.9	--
Cockle Cove Creek	-0.9	0.2
Taylors Pond		
Mill Creek	-0.3	0.2
Taylors Pond	1.7	1.1
Bassing Harbor		
Crows Pond	3.5	1.7
Ryder Cove	7.4	2.8
Frost Fish Creek	-0.2	0.2
Bassing Harbor	-0.1	0.6
Muddy Creek		
Muddy Creek –lower	-1.9	0.6
Muddy Creek - upper	4.7	0.1

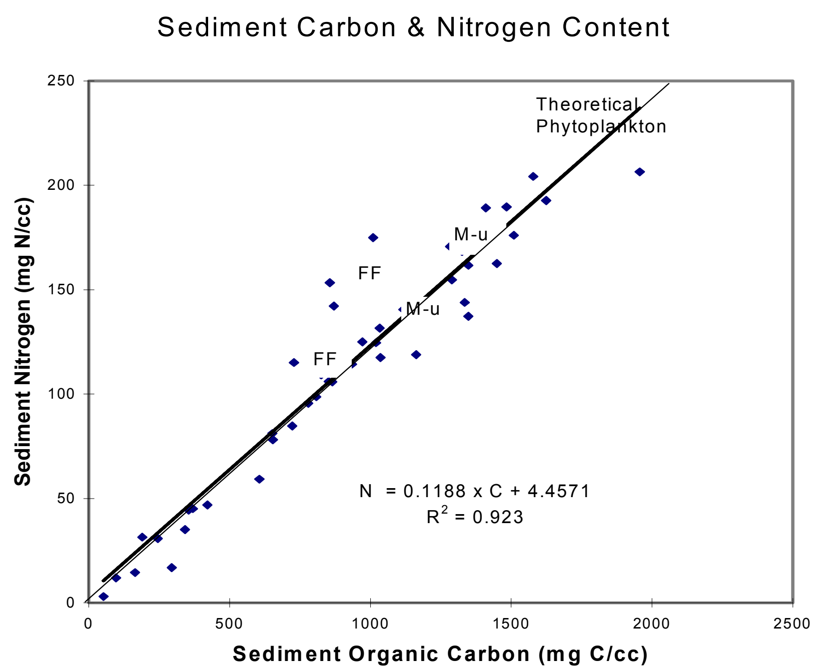


Figure IV-17. Sediment carbon vs. sediment nitrogen content for core samples taken from Chatham subembayments.