NITROGEN ASSESMENT FOR THE LAMPREY RIVER WATERSHED

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EXECUTIVE SUMMARY

1.0 METHODS TO ESTIMATE NITROGEN ATTENUATION

There has been much recent research on nitrogen attenuation in river networks as the issue of coastal N loading and eutrophication has caused scientists to look up into the coastal watersheds. This research has led to advancement of knowledge about the magnitude and mechanisms of riverine N attenuation. There have been several different approaches to estimating river N attenuation that are highlighted here: assessment of nutrient spiraling using in-stream N additions, measurement of N gas emissions, and geospatial modeling. These approaches can be used separately or in concert to provide a better estimate of N attenuation in river networks.

Studies of nutrient spiraling using experimental additions of N indicate that in-stream N uptake is significant, with considerable potential for permanent removal through denitrification. Most of these studies indicate that an increase in N loading to river networks in urbanizing or agricultural watersheds leads to a reduction in the capacity of these streams to remove a substantial portion of that N loading. Ideally, coastal managers would want to maximize denitrification in river networks while not overloading the river's ability to transform a high fraction of the N inputs. Very few *in situ* experimental approaches have occurred in the Great Bay and Piscataqua River watershed and there is a significant need to accurately assess N attenuation in the river networks.

Nitrogen loading in urbanizing watersheds also leads to increased river emissions of N_2 and N_2O through river, soil, and groundwater denitrification. This permanent removal can be beneficial to downstream surface waters such as estuaries, but can also have climatic impacts due to the greenhouse gas potential of N_2O. Measurements of N_2O concentrations are more common than N_2 and in the Lamprey watershed they indicate spatial variation is low, but in areas where there is potential for both high N_2O production and high stream-atmospheric exchange, then the variation and magnitude of flux is likely to increase. Recent technological advances are allowing researchers the opportunity to measure N_2 emissions directly from rivers, which can give us insight into hot spots of denitrification in river networks.

Applying Lamprey and Oyster River N flux models (Section 3.0) to upstream, tributary and downstream points and applying spatially referenced regression (SPARROW) models to reaches of interest allow for estimates of N loading and attenuation (or accumulation) using a geographical information system (GIS). New England SPARROW models over-predict N loading in Lamprey and Oyster basins and therefore may under-estimate N storage or removal mechanisms or over-estimate N inputs in the Great Bay watershed, but the models developed using sub-basins of the Lamprey and Oyster Rivers can accurately be applied to the entire Lamprey River watershed. These Lamprey and Oyster river models could be applied to the wider Great Bay watershed to assess upstream-downstream N attenuation (or accumulation).

These geospatial modeling efforts can inform and direct the use of the experimental approaches outlined above.

2.0 ATMOSPHERIC DEPOSITION OF TOTAL NITROGEN

Atmospheric deposition can account for a large portion of N inputs to watersheds. Total atmospheric deposition of N is the combination of both wet and dry deposition. Wet deposition is the portion of N dissolved in cloud droplets and deposited during precipitation events (e.g. rainfall and snowfall). Dry deposition is the amount of N that settles as aerosols, dust or other deposits on surfaces during periods of no precipitation. The National Atmospheric Deposition Program/National Trends Network (NADP/NTN) is the nation's major source for wet deposition data and has multiple sites throughout the US that are located away from urban areas and point sources of pollution. The Clean Air Status and Trends Network (CASTNET) is a national monitoring network managed by the U.S. EPA that provides data on ambient air quality and dry deposition. Total N deposition is estimated for CASTNET sites as the combination of wet deposition estimated by interpolating NADP/NTN data and measured dry deposition.

Wet and dry N deposition varies among the NADP/NTN and CASTNET sites throughout New England and varies over time. Because geographic factors such as elevation could not accurately explain this spatial and temporal variability, we recommend several approaches to estimating both dry and wet N deposition for Great Bay and the Piscataqua River watershed. The first approach to estimating wet deposition would be to assume that wet deposition rates measured at Thompson Farm (TF) in Durham, NH by NH WRRC staff are uniform across the entire watershed. If data prior to 2004 are needed, the average of two nearby NADP/NTN stations (MA08 and MA13) with similar wet deposition could be used by dividing inorganic wet deposition at these stations by 0.94 (average ratio of DIN:TDN wet deposition at TF) to estimate wet TDN deposition. The second approach to estimate wet deposition would be to assume that the average annual concentration of N in wet deposition measured at TF is consistent throughout the watershed, but that the annual precipitation amount varies based on watershed position. For different locations, the precipitation amount could be estimated from the nearest NCDC or CRN station with similar elevation or could be interpolated among NCDC and CRN stations based on area and elevation using GIS. Once the precipitation amount is determined, it would then be multiplied by the volume-weighted concentration at TF to estimate wet deposition. Another approach to estimate wet deposition across the Piscatagua River watershed would be to use the ClimCalc model (http://www.pnet.sr.unh.edu/climcalc/) developed by Ollinger et al. (2001). This model predicts average wet, dry and total deposition based on latitude, longitude, elevation, slope and aspect and was calibrated on data collected in the 1980's and early 1990's. Despite earlier development of this model, it predicted median deposition (2004-2009) at TF fairly well (under-estimated by 11%) and could be used to estimate wet deposition throughout the Great Bay and Piscataqua River watershed, but is not able to predict variability in deposition over time.

In addition to modeled estimates of dry deposition by ClimCalc (Ollinger et al. 2001), there are two approaches that could be used to estimate dry deposition once wet deposition is estimated. One approach to estimate dry deposition would be to apply the average ratio of dry to wet deposition at the closest CASTNET sites (ABT147 and WST109) during the period of interest to the estimated wet deposition in the Great Bay and Piscataqua River watershed. The other approach to estimating dry to wet deposition ratios for the Piscataqua River watershed would be to apply the ratio of dry to wet deposition predicted by ClimCalc to the estimated wet deposition. ClimCalc predicts a dry to wet deposition ratio of 0.58 for TF. From 2004-2009, median total N deposition (wet + dry) at TF was 7.30 kg N/ha/yr for calendar years and 7.55 kg N/ha/yr for water years (WY) when we use the 0.58 dry to wet deposition ratio predicted by ClimCalc.

3.0 LAMPREY AND OYSTER RIVER BASIN NITROGEN MODELS

Over the past 10 years, the NH WRRC has collected data on nitrogen concentrations in many streams in the Lamprey and Oyster basins. A total of 39 sites have been sampled at a monthly to weekly frequency and our existing stream data has been assembled to determine median annual fluxes of DIN (NO_3-N and NH_4-N), DON and TDN (DIN and DON) from the 39 basins during WY 2000-2009. Landscape characteristics (e.g. % land use/cover, human population density and % impervious surfaces) for the basins were determined using GIS. Median annual net nitrogen inputs, including food, fertilizer (both agricultural and non-agricultural), animal manure and wet and dry deposition, were determined for each basin. Both TDN outputs and N outputs excluding DON flux were compared to net N inputs to determine the fraction of N retention ((inputs-outputs)/inputs).

Various models to predict median N flux and N retention among the sub-basins were developed based on two different approaches. Simple regression analysis was used to examine the relationships between N fluxes and retention and net N inputs. Both simple regression analysis and backwards step wise regression analysis were used to examine the relationships between median DIN, DON and TDN flux and landscape characteristics that represent potential sources or sinks. A similar approach was taken with N retention.

Among Lamprey and Oyster sub-basins, median N inputs ranged from 8.05 to 24.9 kg N/ha/yr, median TDN flux ranged from 0.86 to 6.88 kg N/ha/yr and DIN flux accounted for 15 to 93% percent of TDN flux. Increased N inputs resulted in increased median TDN ($r^2 = 0.62$, $p < 0.01$) and DIN flux ($r^2 = 0.63$, $p < 0.01$), but median DON flux did not respond to net N inputs. The relationship between TDN flux and net N inputs is largely driven by DIN flux, which dominates TDN flux when net N inputs are high.

Landscape characteristics were strong predictors of DIN and TDN flux and significant, but weaker, predictors of DON flux. Human population density was the single best predictor of median DIN flux ($r^2 = 0.76$, $p<0.01$) and human population density, % impervious surface and %

agriculture were retained in the multiple regression model (R^2=0.86, p<0.01). Impervious surface was the single best predictor of median TDN flux (r^2 = 0.68, p<0.01) and % impervious, % agriculture and % wetlands were retained in the backwards multiple regression model (R^2= 0.78, p<0.01; Table 3.2). Percent wetland cover was a significant, but weak predictor of median DON flux (r^2 = 0.14, p<0.05) and % wetlands and % agriculture were retained in the multiple regression model (R^2= 0.28, p<0.01). These results show that DIN and DON respond to different factors in the landscape. DIN responds to human N inputs and DON weakly responds to natural features in the landscape. This has strong management implications for N reduction strategies in the Great Bay and Piscataqua River watershed and suggests that reductions of non-point source N inputs will not reduce DON flux in contributing rivers.

Watershed N retention was fairly high among the sub-basins (61 – 92 %), but declines with increased N inputs (r^2 = 0.30, p<0.01). There is a stronger decline in retention with increased N inputs when DON flux (which does not respond to N inputs) is excluded from outputs used in calculating N retention (r^2 = 0.64, p<0.01). Nitrogen retention excluding DON outputs ranged from 75 to 98% in most sub-basins. This decline in watershed retention (which represents both in-stream and terrestrial retention) with increased N inputs is analogous to the decrease in efficiency (V_f) of overall N uptake and denitrification that occurs in stream networks (Mulholland et al. 2008). In addition to in-stream losses, these watershed-wide retention rates may also be attributed to N storage in soils, vegetation and groundwater, or denitrification in riparian zones or elsewhere in the basin. We have seen elevated groundwater nitrate in the Lamprey watershed, suggesting that N is being stored, and this elevated groundwater could also be driving the long-term increase in Lamprey River nitrate that we have observed.

Landscape characteristics were significant predictors of N retention. Forest cover was the single best predictor of N retention (r^2 = 0.51, p<0.01) and % forest cover and % water were retained in the multiple regression model (R^2=0.58, p<0.01). Impervious surface cover was the single best predictor of N retention excluding DON flux (r^2 = 0.72, p<0.01) and % forest cover, % impervious surface and % water were retained in the multiple regression model (R^2=0.78, p<0.01). These results suggest that forests and streams, rivers, lakes, ponds and water pooled up behind dams are areas where N retention can occur. Impervious surfaces often reduce N retention by by-passing potential areas for terrestrial N retention and deliver N quickly to surface waters. Our results suggest that forests have a greater influence on N retention than surface water, but both are significant mechanisms for N retention.

These models that predict N flux and retention based on both N inputs and landscape characteristics should be directly applicable to other areas of the Great Bay and Piscataqua River watershed in NH. Because the land use/cover data assembled for the Lamprey and Oyster basins (NH LC2001) is limited to NH, these models may need to be adjusted or recalibrated using the Northeast 2006 Land Cover Analysis data (http://www.csc.noaa.gov/crs/lca/northeast.html) for application to areas of the watershed in Maine. Fertilizer and manure N input rates for other counties beyond Rockingham and Strafford County should also be assessed.

RECOMMENDATIONS BASED ON EXISTING INFORMATION

1.0 METHODS TO ESTIMATE NITROGEN ATTENUATION

Given that information on N attenuation in riverine networks under average conditions is needed within the next few months to estimate the portion of WWTF effluent discharged to 4th to 7th order rivers that is attenuated in the Great Bay Estuary watershed, we recommend that the average time specific nutrient uptake rates (k_t) from various SPARROW models be used. We have assembled data from Table 1.2 below and these average rates likely represent the maximum amount of uptake that may occur in Great Bay tributaries because these reaction rates, which are constant within each stream flow category, do not account for the decline in k_t that is associated with deeper streams and streams with higher nitrate concentrations and loads (Alexander et al 2009). However, these relationships between k_t and stream depth and concentration or load were developed on 1st-4th order streams and little information exists on the impact of biological (e.g. NO_3 and DOC availability for denitrification) and hydrological (e.g. stream depth, velocity) factors on k_t in larger streams (Alexander et al. 2009) although we hypothesize that these factors would have a similar impact in larger streams. Additionally, these reaction rates do not incorporate the potential importance of lakes and reservoirs at large scales (Harrison et al. 2009) and this may invalidate our assumption that these average in stream loss rates represent maximum values. Reservoir retention was significant in the Chesapeake SPARROW but not the other SPARROW models. Because the New England SPARROW model over-predicts N flux, we believe that in stream and reservoir retention may be more important than the New England model suggests (where k_t is only significant in streams where the average stream flow (Q) is < 2.83 m^3/s or 100 cfs). We could expect the minimum uptake for the entire reach to be between 0 and 0.78 d^{-1} for small streams only (Q < 100 cfs; Moore et al. 2004). Over the long-term (next several years), more research is needed to better constrain these estimates of in stream N loss and determine how much N is permanently removed from the watershed via denitrification.

Mean Stream flow Category	SPARROW Models Used	Average In Stream Loss Coefficient (k_t; d^{-1})
Mean Q $\leq$ 200 cfs	Chesapeake and New England	0.770
200 cfs < Mean Q $\leq$ 1000 cfs	Chesapeake, Mississippi and National	0.378
Mean Q > 1000 m^3/s	Chesapeake, Mississippi and National	0.103

2.0 ATMOSPHERIC DEPOSITION OF TOTAL NITROGEN

To estimate deposition throughout the Great Bay and Piscataqua River watershed based solely on readily available data, we recommend using measured wet deposition and estimated dry and total N deposition for TF in Durham, NH presented in Table 2.9. If deposition estimates are needed prior to 2004, we recommend taking the average wet inorganic N deposition measured at NADP stations MA08 and MA13, dividing by 0.94 (ratio of wet DIN:TDN deposition at TF) to estimate

wet TDN deposition and multiply the estimated wet TDN deposition by 0.58 (ratio of Dry:Wet deposition at TF predicted by ClimCalc) to estimate dry deposition. The sum of estimated wet TDN and dry N deposition would then be used to estimate total N deposition prior to 2004. For example, total N deposition for CY 2003 and WY 2003 would be 7.12 and 7.60 kg/ha/yr, respectively. If more funding becomes available, geospatial models should be used to estimate annual precipitation amounts throughout the Great Bay and Piscataqua River watershed. These precipitation amounts could then be used in conjunction with precipitation-weighted mean TDN concentrations at TF (or average estimated precipitation-weighted mean TDN concentrations at MA08 and MA13 for years prior to 2004) and ClimCalc ratios of dry:wet deposition to estimate spatial variability in total N deposition throughout the watershed.

3.0 LAMPREY AND OYSTER RIVER BASIN NITROGEN MODELS

Because spatial variability in DON flux does not relate to the spatial variability in non-point N inputs and is only weakly related to landscape features ($r^2 = 0.14$ to 0.28, $p<0.05$), we recommend that management efforts to reduce non-point sources of N in the Great Bay tributaries focus on reducing DIN flux which is strongly related to non-point N inputs ($r^2 = 0.73$, $p <0.01$ when sub-basins with suspected leaky sewer lines or illicit discharges are excluded) and landscape features ($r^2 = 0.76$ to $R^2 = 0.86$, $p<0.01$). The single best predictor of median DIN flux was human population density ($r^2 = 0.76$, $p<0.01$) and this relationship could be used to predict DIN flux throughout the Great Bay and Piscataqua River watershed (Table 3.3). If % impervious surface and % agriculture data (ideally from NH LC2001) are also available, they could be used in conjunction with human population density to increase the predictive power of DIN flux ($R^2=0.86$, $p<0.01$).

To estimate current N inputs, we recommend following a similar approach to the one presented in section 3.0 (Table 3.1) and recalibrating the fertilizer and manure N input rates to other counties throughout the Great Bay and Piscataqua River watershed. Once these inputs are determined, they can be used to estimate DIN flux (Fig. 3.3) and to calculate N retention. Since DON flux does not respond to non-point N inputs, we recommend excluding DON outputs from outputs used in N retention calculations. This exclusion is critical to estimate the extent to which N flux can be reduced through management activities. Watershed N retention rates excluding DON outputs (which range from 75 to 98% in most Lamprey and Oyster sub-basins) decline with increased N inputs ($r^2 = 0.64$, $p<0.01$; Fig. 3.15) and estimated N inputs can be used to estimate overall watershed N retention (i.e. both in-stream and terrestrial storage and denitrification). The single best predictor of N retention excluding DON flux was impervious surface ($r^2 = 0.72$, $p<0.01$) and this relationship could be used to predict N retention throughout the Great Bay and Piscataqua River watershed (Table 3.4). Note, however, that other predictors of N retention (excluding DON flux) are almost as effective (% forest cover, $r^2 = 0.68$ and human population density, $r^2 = 0.61$). If % forest cover and % water are also available (ideally from NH LC2001 data), they could be used in conjunction with % impervious surface cover to increase the predictive power of N retention excluding DON flux ($R^2=0.78$, $p<0.01$).

1.0. METHODS TO ESTIMATE NITROGEN ATTENUATION

Nitrogen uptake and transformations along flow paths are complex in both terrestrial and aquatic environments. One of the fundamental complexities stems from the fact that there are multiple forms of nitrogen, each of which behaves in a distinctly different manner and is subject to different biogeochemical transformations and thus is attenuated differently following its introduction into a watershed. From the standpoint of water quality management, the most important forms of nitrogen to consider are particulate nitrogen (PN), dissolved organic nitrogen (DON), ammonium (NH_4^+), nitrate (NO_3^-) and nitrite (NO_2^-). The combination of all these forms of nitrogen is termed Total N (TN). The combination of NH_4^+ and NO_3^- plus NO_2^- (often measured together) is termed dissolved inorganic N (DIN). The gaseous forms of nitrogen are also important in understanding water quality, as N_2O and N_2 are produced by denitrification. In oxic environments, and those with high anthropogenic N loading, NO_3^- typically dominates DIN. DON consists of various amino acids, urea and other organic compounds and is typically reported as a bulk measure of organic N. In this report, we will focus on the dissolved forms of nitrogen since total dissolved nitrogen (TDN) dominates total nitrogen in the Lamprey watershed (83-89% of total N in the Lamprey River and two tributaries for which we have extensive data. A similar situation exists in most streams and groundwater).

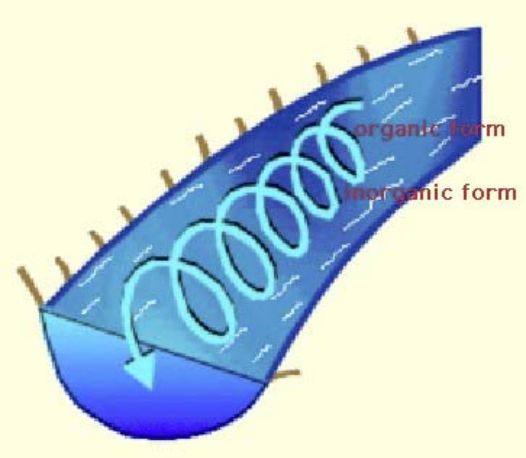

Fig 1.1. Conceptual figure of nutrient spiraling. (http://science.kennesaw.edu/~jdirnber/limno/LecStream/LecStreamEcologyPhysChem.html)

Attenuation of nitrogen as it moves through the landscape occurs when nitrogen is taken up by microbes or plants, stored as soil organic matter, or denitrified. Nitrogen attenuation is most easily studied in streams, where various work has shown that it is not passively transported downstream, but instead is typically assimilated by aquatic organisms and cycled between organic and inorganic forms as it is moves downstream (Webster 1975). This concept is referred to as nutrient spiraling (Fig. 1.1). Although a similar framework could be applied to terrestrial ecosystems, the literature is much less developed than it is for streams. Recent research on streams in the United States indicates that a significant portion (more than 50%) of the N load can be retained or transformed within hundreds of meters (Peterson et al. 2001). The various N transformations that can occur as N travels through the terrestrial and aquatic environment are shown in Fig 1.2. Of the many transformations that N can undergo only denitrification (the conversion of NO_3^- to nitrous oxide (N_2O) and nitrogen gas (N_2)) leads to permanent removal of nitrogen from streams or groundwater.

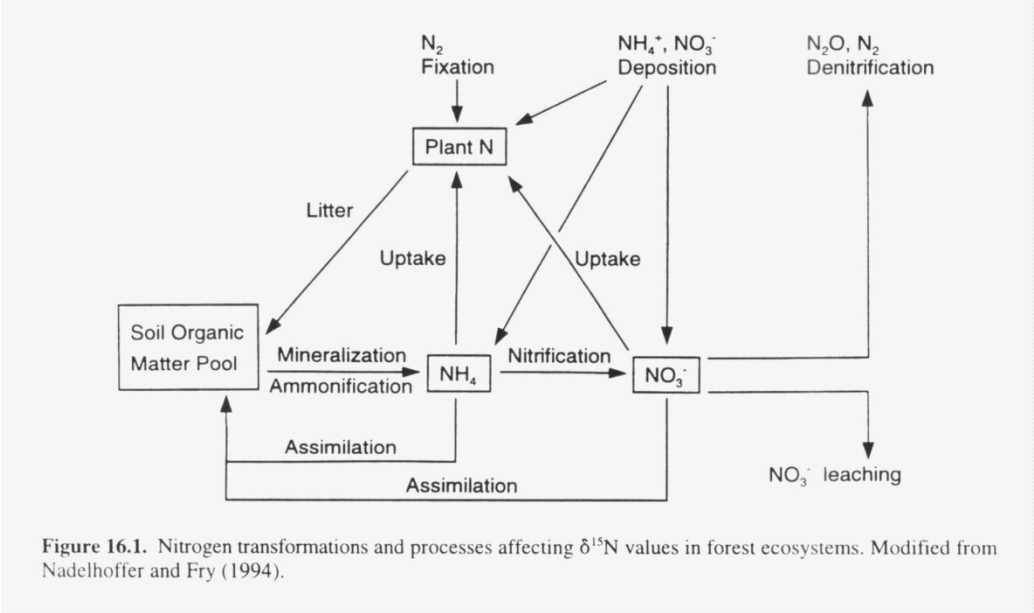

Figure 16.1. Nitrogen transformations and processes affecting $\delta^{15}N$ values in forest ecosystems. Modified from Nadelhoffer and Fry (1994).

Fig. 1.2. Nitrogen cycle in forests (from Kendall 1998).

1.1 NITROGEN ATTENUATION IN RIVERINE NETWORKS

Most research on nitrogen uptake and denitrification in riverine and stream networks has focused on the uptake or removal of experimentally added NO_3^- or NH_4^+ (Ensign & Doyle 2006). Other research has focused on measuring ambient N_2O, N_2 and atmospheric exchange rates as a way to estimate denitrification and the amount of N permanently removed. Lastly, modeling efforts that predict stream water N concentration and flux from landscape characteristics quantified using a geographical information system (GIS) can be useful in assessing N removal or accumulation along a reach. Here we will examine the various methods in each of these three broad categories which could be used to quantify N uptake and denitrification in river networks.

1.1.1 Lotic N Additions

Small stream *in situ* additions

The simultaneous uptake, release, and transport of nutrients in fluvial ecosystems has been termed "nutrient spiraling" (Webster 1975) and is one of the cornerstones of stream ecosystem ecology. The most popular and effective method for measuring nutrient spiraling in small streams is to conduct *in situ*, short-term experimental manipulations in which the nutrient of interest is added to the stream along with an inert tracer at a constant rate (Stream Solute Workshop 1990). Once stations downstream of the point of addition reach a constant tracer concentration (or steady state "plateau"), the stream segment is sampled at several stations along its length. The concentration of added nutrient can be corrected for groundwater dilution by the decrease in tracer concentration, as groundwater entering the stream is lower in tracer

concentration than the stream. After correcting for dilution by incoming groundwater, any decrease in nutrient concentration relative to the tracer is due to uptake in the stream.

The average distance that N or any added nutrient travels until it is taken up is termed the uptake length ($S_{W,}$ meters); V_f (uptake velocity, cm sec^{-1}) and U (uptake rate, mg m^{-2} hr^{-1}) are also valuable metrics for quantifying and comparing in-stream rates of N removal. There are two basic approaches to in-stream nutrient spiraling experiments: isotope tracers and bulk inorganic additions. Isotope approaches are preferred, since specific N uptake pathways and mechanisms can be measured and their rates determined at ambient concentrations. Isotopic tracer additions are especially valuable because they allow for the direct measurement of both denitrification end products, N_2 and N_2O. Fertilization experiments in which nutrient concentrations are elevated significantly over background concentrations can overestimate uptake length, especially at low ambient concentrations (Mulholland et al. 2002; Dodds et al. 2002). The drawback of isotopic approaches is cost, so methods for attaining ambient uptake rates with cheaper bulk inorganic nutrient additions have been developed (Payn et al. 2005).

The majority of the information on in-stream N dynamics with isotopic tracer additions comes from the two Lotic Intersite Nitrogen eXperiment (LINX) studies, in which staff of the UNH Water Quality Analysis Lab were active participants and authors. These experiments are summarized in Peterson et al. (2001), Webster et al. (2003), Mulholland et al. (2008), Mulholland et al. (2009), and Hall et al. (2009). In LINX I, Webster et al. (2003) conducted stream $^{15}NH_4$ additions in a variety of ecosystems and quantified uptake among the different stream compartments and trophic transfers. They found good correspondence between N assimilation and biological demand and determined that the factors that control the autotrophic/heterotrophic balance are what indirectly control N uptake. Overall the amount of N assimilated between streams in different biomes is fairly constant (within an order of magnitude) and this indicates metabolic compensation among the streams. One of the study streams was Bear Brook, a small heterotrophic stream with low nutrient concentrations and a forested canopy in the White Mountains of New Hampshire. NH_4 uptake (V_f = 0.119 mm s^{-1}) was similar to the median (0.154 mm s^{-1}) and to other streams of the same type reflecting consistency in N assimilation.

The LINX II study quantified N uptake (Hall et al. 2009) and removal (Mulholland et al. 2009) in streams of varying land use with $^{15}NO_3$ additions. This was the first cross system comparison of whole reach *in situ* denitrification. They found that NO_3 removal (denitrification) is a significant portion of total uptake (median = 16%) across streams, with high variation among streams and the highest rates in streams that receive substantial amounts of anthropogenic N loading. They also found that the efficiency (V_f) of both overall N uptake and denitrification declines with N loading, with important implications for river networks (Mulholland et al. 2008; Fig. 1.3). As human population density and overall land use change increases in a watershed, the ability of the river network to retain N decreases. The capacity of benthic organisms to take up N has been demonstrated to exhibit saturation in many small scale studies (Dodds et al. 2002;

Bernot et al. 2006) and even efficiency loss has been demonstrated more recently (O'Brien et al. 2007). In an urbanizing basin such as the Lamprey, where a large portion of N inputs is retained, this decrease in fractional N retention with increased nitrate inputs likely has significant consequences for Great Bay.

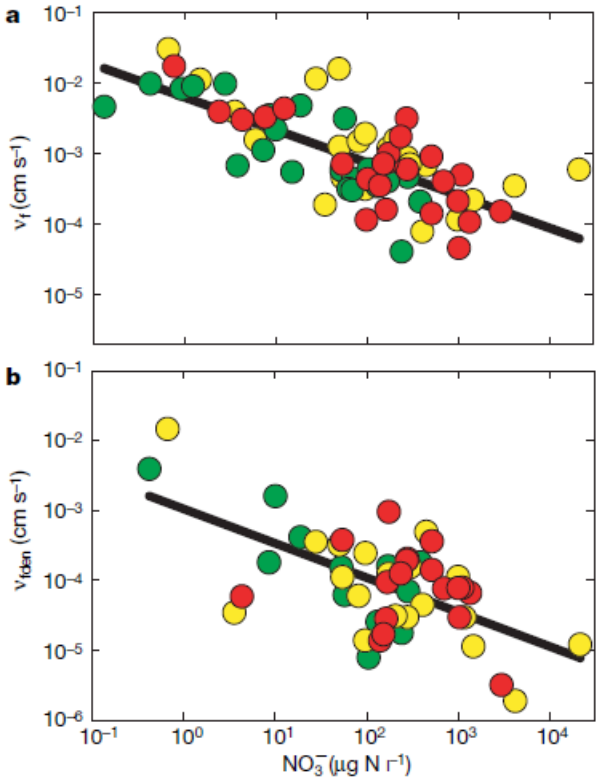

Figure 2 | Relationships between NO_3^- uptake velocity and concentration. **a**, Regression of total NO_3^- uptake velocity (v_f) on NO_3^- concentration (log $v_f = -0.462 \times \log[NO_3^-] - 2.206$, $r^2 = 0.532$, $P < 0.0001$). **b**, Regression of denitrification uptake velocity (v_{fden}) on NO_3^- concentration (log $v_{fden} = -0.493 \times \log[NO_3^-] - 2.975$, $r^2 = 0.355$, $P < 0.0001$).

Fig. 1.3. Relationships between NO_3-N uptake velocity and Concentration (from Mulholland et al. 2008).

One of the LINX II study sites, Plum Island Estuary, is located in a similar watershed and region (coastal New England) as the Lamprey River basin. Denitrification rates were high (V_{fden} = 0.067 mm s^{-1}), although total NO_3 uptake rates were low (1.29 x 10^{-4} cm s^{-1}) in comparison to other sites in the US (Mulholland et al. 2008). The low gradient with high particulate organic matter substrate and relatively high concentrations of NO_3 (mean = 584 ug N L^{-1}) led to these high denitrification rates. In similar reaches in the Lamprey River network, we can expect to see similar denitrification rates.

Large Reach Assessment

One of the other significant findings from LINX I was that small streams play a disproportionately large role in river networks in removing N, with headwater streams capable of removing more than half of inorganic N inputs during periods of high biological activity (Peterson et al. 2001). The literature is contradictory in this regard as many studies indicate small streams are the most important for N removal (Alexander et al. 2000; Peterson et al. 2001; Muholland et al. 2008), while others suggest that on balance, large rivers are more important (Schlesinger et al. 2006). What is known is that the relative importance of nutrient removal among different size rivers depends on how the comparisons are made (Wollheim et al. 2006). On equally sized segments, small order streams are more important, but when considering the entire stream's length, then proportional removal of upstream inputs are much greater in larger rivers due to increased water residence time. At any rate, few studies have empirically measured N uptake in larger rivers (> 4^{th} order) (Ensign & Doyle 2006), with ~90% of uptake measurements made in streams with discharge <200 L/s (Tank et al. 2008). This is both a critical research need that would help clear up the comparisons of river size and a management need, as evidenced by the large amount of N unaccounted for in a landscape like the Lamprey River basin.

Wastewater discharged from municipal facilities is predominantly directed into river reaches that are larger than what is economically and logistically feasible for study with the whole stream addition approach; therefore, two different approaches need to be considered at this larger scale. In the case of river reaches that receive wastewater treatment plant (WWTP) effluent, it may be possible to intensively study the effluent input similar to a constant rate injection experiment (Marti et al. 2004; Mereseburer et al. 2005). The relative contribution of the WWTP to Cl^- and nutrient loads is determined by measuring and calculating their flux in the effluent and in the river above and below the effluent discharge. Longitudinal variation in Cl^- is then used as the conservative tracer and nutrient uptake can be calculated.

In the past 50 years, WWTPs have been constructed to improve water quality overall, but decreased nutrient loads remains elusive due to economical restraints (Paul and Meyer 2001). Large, long-term N loading from WWTPs appear to stress stream communities and reduce their retention efficiency (Marti et al. 2004; Mereseburer et al. 2005). The effects of WWTP effluent on streams can be further exacerbated in watersheds with significant land use change that contributes non-point source inputs (Mereseburer et al. 2005). The ecological effect of WWTPs is even significant over long distances (kilometers) as indicated from the calculated uptake distances (Haggard et al. 2001; Marti et al. 2004). This has significant implications for receiving water bodies such as Great Bay.

In larger reaches that do not receive WWTP effluent (and where WWTP effluent tracing may not work or no access is granted to the effluent) the instantaneous (or pulse) addition approach can be used (Tank et al. 2008; Powers et al. 2009), which is the addition of a nutrient

with a conservative tracer in a single, rapid pulse and subsequent measurement of the decline downstream. The small stream, constant rate injection approach is economically and logistically impractical in larger systems, since nutrient fluxes are too high due to a combination of discharge and nutrient concentration. Recent hydrologic tracer research suggests that constant rate injections result in a more consistent loading of all storage zones along a reach and therefore have the potential for more reliable inter-site comparisons of stream solute transport (Wagner & Harvey 1997; Gooseff et al. 2008), but pulse additions have been shown to give comparable results to constant rate injections in recent nutrient uptake studies and demonstrate that they are adequate for many purposes (Triska et al. 2006; Powers et al. 2009). Additionally, the transport-based model (using non-linear regression OTIS-P) for analysis of nutrient spiraling developed by Runkel (2007) can be used for pulse injections and it incorporates changing conditions in time and space (with time series sampling), making them more versatile. This relatively new analysis combined with pulse additions also allows the investigator to measure stream nutrient uptake in situations not possible with the simple linear regression approach that is widely used (i.e. variable flows & chemistry, high discharge, and high water residence time).

The majority of pulse additions have been done using P as the nutrient of interest (Meals et al. 1999; Triska et al. 2006; Powers et al. 2009). Most of these studies have found significant P uptake in larger river reaches and at the same order of magnitude as headwater streams. Tank et al. (2008) performed a large river N pulse addition in the Upper Snake River, Wyoming, where they found that biotic N demand likely stays the same (or maybe even increases) with increasing river size. This was also found in a large river, constant rate injection for NH_4 in Alaska (Wollheim et al. 2001). Tank et al. (2008) also found that the demand for the different species of N changes with river size, as NH_4 is preferred over NO_3 in small streams, but there is no preference in larger rivers. This corroborates the findings of modeling efforts (Ensign & Doyle 2006; Wollheim et al. 2006). These types of empirical reach studies have not been applied to rivers where human driven ecological changes lead to higher N loads, including those in New England.

1.1.2 Measurements of Ambient N_2O, N_2 and atmospheric exchange

Measurements of the production of N_2O and N_2 from streams and rivers have the potential to provide useful information on rates of stream denitrification, but their use is not widespread due to a variety of technical issues associated with both measurement of N_2, and the partitioning of watershed and in-stream sources of any gas evasion that is measured from stream surfaces. Global atmospheric concentrations of N_2O have increased from approximately 270 ppbv in 1750 to 314 ppbv in 2000 (EPA 2007; IPCC 2007; Garnier et al. 2006; Bange 2000). This increase is due to both anthropogenic sources (direct emissions from agriculture, sewage treatment, and mobile and stationary combustion of fossil fuels) as well as indirect effects from increased use of reactive nitrogen since the development of the Haber-Bosch process (Peters et al. 2005; Galloway and Cowling 2002; Bouwman et al. 1995), and as part of the natural nitrogen cycle. The few studies that have evaluated the flux of N_2O from rivers and streams to the

atmosphere are limited both spatially and temporally as current methods are often labor-intensive and expensive in the case of N_2O (Garnier et al. 2006; Clough et al. 2006; McMahon and Dennehy 1998). However, freshwater streams have the potential to be significant contributors of N_2O to the global atmosphere, based on a global assessment of the importance of global surface waters in the carbon cycle (Cole et al. 2006). Dissolved N_2O can enter rivers directly through runoff or groundwater inputs, or it may be produced in the system via nitrification and denitrification (McMahon and Dennehy 1998). Denitrification occurs in conditions of low oxygen or at the interface between the anaerobic streambed and the river, while nitrification may occur within the water column (Clough et al. 2006).

Denitrification in streambed sediments is particularly important as it represents a permanent removal of NO_3^- from a stream in the form of N_2O and N_2 gases, and as such, has the potential to decrease the threat of eutrophication to coastal waters (Clough et al. 2006). Although this N removal is typically seen as beneficial to the ecological integrity of surface waters, it also has negative climate change implications because N_2O has an effective warming potential of approximately 310 times that of CO_2 and can persist in the atmosphere for up to 120 years (Granier et al. 2006; Vitousek et al. 1997). The release of N_2O from streams may indicate that incomplete denitrification is occurring, as NO_3^- is not being completely reduced to N_2, but is only being converted to N_2O. This has been shown to occur in areas of high nitrate and low pH (Ullah and Zinati 2006; Hefting et al. 2003; Firestone et al. 1980). Although NO_3^- is being removed from the stream regardless of which end-product of denitrification is released, N_2 is more desirable as it does not have the negative air quality effects of N_2O (Hefting et al. 2003).

Dissolved N_2O concentrations and flux from streams

One method commonly used to determine N_2O flux from streams is to measure the concentration of dissolved N_2O in stream water, calculate the atmospheric exchange coefficient (k), and use the N_2O concentration and exchange rate to calculate the exchange of N_2O with the atmosphere. Although this method does not allow for a determination of sources of N_2O (from groundwater inputs or in-stream production via denitrification), it does allow the magnitude of the contribution of a specific stream to be determined. In the field, water samples are collected in syringes submerged underwater to prevent contamination from atmospheric N_2O. The concentration of dissolved N_2O is determined by extracting gases from water samples through the addition of high purity helium in the lab. These samples are then injected into glass vials sealed with butyl-septa stoppers and analyzed on a gas chromatograph.

DiFranco (2009) measured the concentration of dissolved N_2O in 16 streams throughout the Lamprey River basin using the method described above. The concentrations found in this study (0.9 to 2.71 μg N_2O-N L^{-1}) were within the range of values found in the literature (Table 1.1). Calculation of the atmospheric exchange coefficient (k) is typically used with this method to determine an estimate of flux. Using the median exchange coefficient found for streams and small rivers in the literature (19.4 d^{-1}; Bernot et al. in press) the magnitude of net efflux from the

Lamprey River streams could be 1998 to 6016 μg N m^{-2} day^{-1}. In the DiFranco study, no direct measurements of exchange coefficient were possible, and thus percent saturation was used to understand the potential for degassing of N_2O from the Lamprey River basin (if percent saturation exceeded 100%, there is potential for loss of N_2O to the atmosphere). Overall, the Lamprey River basin was found to be a net source of N_2O to the atmosphere as most streams exceeded 100% saturation of N_2O on each date sampled (DiFranco 2009).

Estimating/Measuring Atmospheric Exchange

Stream reaeration is the rate at which gases are exchanged with the atmosphere, and is driven by physical parameters such as stream velocity, slope, and turbulence. Because streams are typically supersaturated with gases such as CO_2 (Cole & Caraco 2001a) and N_2O (Cole & Caraco 2001b), the tendency of a stream to approach gaseous equilibrium through gas exchange with the atmosphere results in a net gaseous efflux from the stream surface. The classic experimental approach to measuring the reaeration coefficient *k* is with in-stream addition of a volatile, non-reactive gaseous tracer and downstream measurement of its loss (Kilpatrick et al. 1989). In addition to being non-reactive, the tracer should not be found in incoming groundwater, nor should it be produced or consumed in the stream channel. The amount of the tracer gas lost is used to estimate *k*. Many gases have been used as aquatic tracers, but the most frequently used are hydrocarbons like propane (Rathburn et al. 1975), and inert gases such as SF_6 (Hibbs et al. 1998).

In addition to measuring exchange coefficient directly, predictive equations available in the engineering literature for use in waste load allocation studies can also be used to estimate reaeration coefficient (Kilpatrick et al. 1989). Physical measurements of the stream (energy dissipation: channel slope, water velocity, and depth) can be used to predict *k* (Tsivoglou & Neal 1976; Melching & Flores 1999). In comparisons between the experimental and modeling approaches (Young & Huryn 1999; Mulholland et al. 2001) it appears that in larger river reaches where reaeration rates are lower (< 50 day^{-1}) predictive equations work well, but in small streams (depth < 6cm) the energy dissipation models underestimate the actual reaeration rate. At the same time, addition and complete mixing of gas tracers to large river reaches is difficult. In their analysis of 371 individual reaeration measurements in USGS stream reaches, Melching & Flores (1999) found that stream reaeration varied considerably from about 0.1 to 100 day^{-1}. Therefore it is critical to water quality models that a reasonable value for *k* is determined. In a river network that spans many stream orders like the Lamprey and other tributaries to Great Bay, a combination of approaches might be warranted to estimate *k*.

Closed Chamber Method

The closed chamber method, which does not rely on measurement of *k*, is another method that has been used to measure the flux of N_2O from rivers. In this method, a closed acrylic or polypropylene container with a known volume and surface area is suspended over the river to

directly collect N_2O. A rubber septum at the top of the chambers allows for gas sample collection via syringes at regular intervals over a set period of time. These samples are then injected into glass vials sealed with butyl septa stoppers and analyzed in the laboratory on a gas chromatograph.

Clough et al. (2006) used the closed chamber method to estimate the flux of N_2O from a 12-km river in New Zealand. At each of the four sampling sites along the river, ten floating conical chambers were deployed approximately 1-m apart and allowed to drift with the current . They found a total average flux of 912-12,2024 $\mu gN_2O\text{-}Nm^{-2}day^{-1}$, with the river consistently oversaturated with dissolved N_2O (200-400%) (Table 1.1). McMahon and Dennehy (1998) used a similar method to determine the flux of N_2O from a 733-km reach of the South Platte River in Colorado. Chambers were suspended over the river using tripods at nine sampling sites (3-5 replicates per site). All sampling sites were oversaturated with N_2O, with some as high as 2500%, and flux was estimated to range from 90 to 32,600 $\mu g\ N\ m^{-2}d^{-1}$ (Table 1.1).

Table 1.1. Estimates of N_2O concentrations, percent saturation, and flux for multiple studies.

Site	Watershed Size (km^2)	Discharge (cfs)	Measurement Type	Dissolved N_2O concentrations	Percent Saturation	N_2O Flux	Study
Lamprey River Watershed, NH	479	0.27 – 2040.59	Dissolved Gas	0.9 -2.71 $\mu g\ N_2O\text{-}NL^{-1}$	45 – 705%	--	DiFranco Thesis (2009)
North Island, New Zealand	5- 48.9 (3 small watersheds)	0.03 – 3.85	Dissolved Gas	0.16 – 11.35 $\mu g\ N_2O\text{-}NL^{-1}$	--	5501.8 – 2.9 x 10^4 $\mu gNm^{-2}day^{-1}$	Wilcox and Sorrel (2008)
Indiana/ Illinois/ New Jersey	--	70.63- 953.50	Dissolved Gas	0.006 – 0.017 $\mu g\ N_2O\text{-}NL^{-1}$	102-209%	422.4 – 6.3 10^4 $\mu gNm^{-2}day^{-1}$	Laursen and Seitzinger (2004)
Hudson River, NY	33,500	12,360	Dissolved Gas	0.196 – 0.606 $\mu g\ N_2O\text{-}NL^{-1}$	125-385%	4,383.6 $\mu g\ Nday^{-1}$	Cole and Caraco (2001)
South Island, New Zealand	--	26.82	Closed Chamber	0.43-1.89 $\mu g\ N_2O\text{-}NL^{-1}$	~200-400%	912-12,024 $\mu gNm^{-2}day^{-1}$	Clough et al. (2006)
South Platte River, CO	63,000	2752.87	Closed Chamber	--	Up to 2500%	90-32,600 $\mu gNm^{-2}day^{-1}$	McMahon and Dennehy (1998)

Both the chamber and gas concentration methods have been used to determine the contribution of freshwater streams to the atmospheric concentration of N_2O. As shown in Table 1.1, concentrations, percent saturation, and flux of N_2O vary both between and within studies. These studies are often completed on large rivers with sampling stations dozens of kilometers apart, which may either over or underestimate the actual flux of N_2O due to variation in concentration and *k* values in different reaches.

Spatial differences in dissolved N_2O may be attributed to external inputs of N_2O from groundwater or overland flow, differences in denitrification rates, or differences in rates of atmospheric exchange. A study by Reay et al. (2002) showed strong small-scale spatial variation in flux and concentration of N_2O from agricultural drainage waters. This variation was thought to be caused by external inputs of dissolved N_2O from the surrounding agricultural fields. Within the Lamprey River basin, large and small scale spatial differences in the concentration and percent saturation of N_2O were not found (DiFranco 2009). However, it is possible that there are differences in the actual flux of N_2O due to differences in *k* values throughout a given reach of a stream. If this were the case, areas with a high production of N_2O and a high atmospheric exchange rate may have lower concentrations of N_2O than areas of low production and low atmospheric exchange. More research is needed to understand spatial variation of the flux of N_2O within a given reach.

N_2:Argon in the river

Membrane inlet mass spectrometry (MIMS) has made it possible to measure very small changes in dissolved gases (Kana et al. 1994), and has been applied to N removal in aquatic ecosystems with the measurement of the N_2:Argon in stream water. N_2 gas is affected by biological (mainly denitrification) and physical processes, while Ar is affected strictly by physical processes. Therefore an increase in the N_2:Ar would indicate production of N_2 through denitrification. This approach has the advantage that it provides a direct measure of denitrification without the benthic disturbance that other methods such as incubation of sediment cores would entail, and it does not require addition of stable isotopes. The shortcomings of MIMS are its insensitivity to shallow streams (due to gas exchange) and error due to other processes that alter N_2 concentrations (Bohlke et al. 2004). Regardless, this approach is still in its infancy (only a few studies to date) and the prospect for direct measurement of denitrification in a variety of aquatic ecosystems is "excellent" (Groffman et al. 2006). At this time it appears that this method is best suited for low gradient, large rivers where denitrification rates are high (McCutchan et al. 2003).

In the original N_2:Ar river study in high NO_3 streams in the (Laursen & Seitzinger 2002), the change in N_2:Ar was measured with distance and corrected for air-water gas exchange with addition of conservative gas tracers. They found that denitrification rates were similar to those in high NO_3, agriculturally influenced rivers from mass balance studies. Another approach is to measure N_2:Ar in a time-series and correct for air-water gas exchange and groundwater

discharge (McCutchan et al. 2003). This is very similar to (and uses the same calculations as) the open-channel O_2 technique of measuring ecosystem metabolism in rivers and may allow simultaneous measurement of denitrification and ecosystem metabolism in the same MIMS sample in the future. McCutchan et al. (2003) found denitrification rates comparable to mass balance studies in the same river and similar to the Laursen & Seitzinger (2002) study. We are not aware of similar studies that have been attempted anywhere in New England to date.

1.1.3. N attenuation or accumulation along a reach using models

Reach analysis between two sample points

Simple and multiple regression models developed for the Lamprey River and Oyster River basins (Section 3.0) could be employed to estimate nitrogen attenuation or accumulation along a river reach throughout the Great Bay watershed. Total dissolved nitrogen flux (kg N/ha/yr) from a sub-basin can be predicted by sub-basin % wetland cover, % agriculture and % impervious surface cover (R^2=0.80, p<0.01) or from % impervious surfaces (r^2=0.68, p<0.01) or human population density (r^2=0.67, p<0.01) alone. These relationships could be used to estimate TDN load (kg/yr; estimated flux (kg/ha/yr) x area (ha)) upstream, from tributaries along the river reach, for areas along the river reach that are not drained by a defined tributary and at the reach outlet (Fig 1.4). Attenuation or accumulation of nitrogen would be calculated as follows:

TDN attenuation or accumulation (kg/yr) = estimated TDN load at outlet – (estimated upstream TDN load + ∑ tributary estimated TDN load + estimated TDN load for riparian area not drained by defined tributary).

Another approach would be to estimate both DIN flux (from sub-basin % agriculture, % impervious surface and human population density, R^2 =0.86, p<0.01; or from human population density alone, r^2 =0.76, p<0.01) and DON flux (from sub-basin % wetland cover and % agriculture; R^2 = 0.28, p<0.01) for points along the reach and estimate respective DIN and DON loads. Attenuation or accumulation of DIN and DON would be estimated in a similar fashion to calculated TDN attenuation or accumulation above and provide another approach to estimate TDN attenuation or accumulation:

DIN attenuation or accumulation (kg/yr) = estimated DIN load at outlet – (estimated upstream DIN load + ∑ tributary estimated DIN load + estimated DIN load for riparian area not drained by defined tributary).

DON attenuation or accumulation (kg/yr) = estimated DON load at outlet – (estimated upstream D)N load + ∑ tributary estimated DON load + estimated DON load for riparian area not drained by defined tributary).

TDN attenuation or accumulation (kg/yr) = DIN attenuation or accumulation + DON attenuation or accumulation

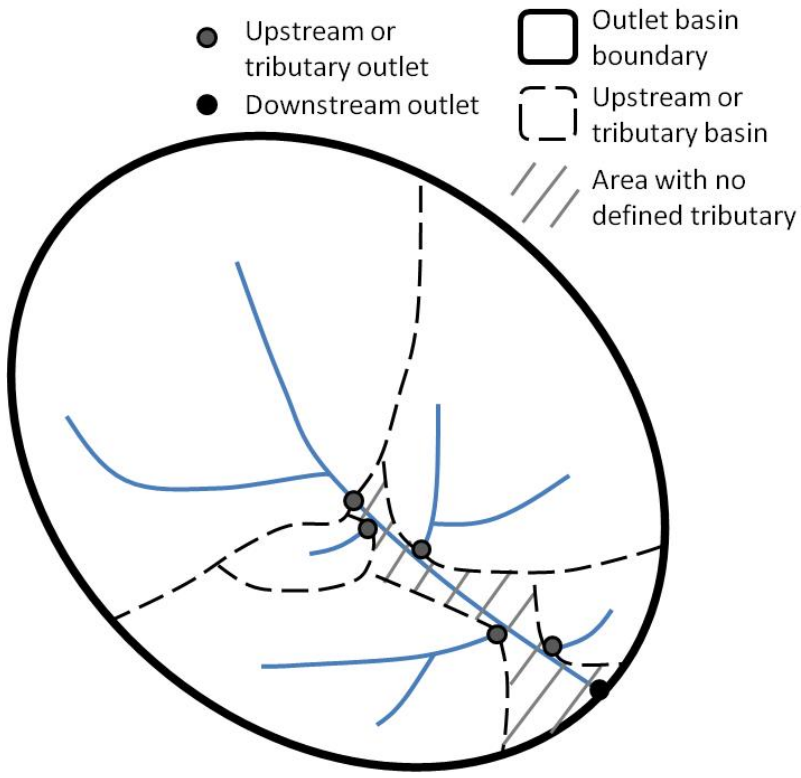

Fig. 1.4 Schematic of upstream, tributary and downstream outlets and respective basins.

Spatially Referenced Regression Models

Various spatially referenced regression models have been developed to date. Spatially Referenced Regressions on Watershed Attributes (SPARROW) models have been developed for the entire US (Smith et al. 1997), the Chesapeake Bay watershed (Preston and Brakebill 1999), the Mississippi watershed (Alexander et al. 2000), New England (Moore et al. 2004) and the New England SPARROW model has been refitted to coastal New Hampshire (Legere 2007) and will be referred to here as the preliminary coastal NH model. SPARROW is a spatially detailed non-linear statistical model that uses regression equations to relate total N loads (kg/yr) to nutrient sources and watershed characteristics. The model predicts total N load based on point and non-point sources of nitrogen, a land to water delivery term, an in stream loss coefficient (k_t, d^{-1}) and an error term. The preliminary coastal NH was developed by refitting the New England SPARROW model coefficients to coastal NH and included wetland cover as an additional source variable (Legere 2007). In the preliminary coastal NH model, wetland cover was defined by automated National Wetlands Inventory (NWI) maps that are more accurate than the 1992 National Land Cover Data (NLCD) used in New England SPARROW or the 2001 NH Land Cover Assessment data used for coastal NH (Legere 2007; Daley 2002).

The three different models that encompass the Great Bay watershed (the US, New England and preliminary coastal NH SPARROW) result in different land to water delivery terms and in-stream losses. In the national model, soil permeability, drainage density and stream loss were significant delivery variables (Table 1.2). In the New England SPARROW model, soil permeability and stream loss were the only delivery variables found to be significant and the stream loss coefficient (0.78/day) is only significant in streams with mean flows less than 2.83 m^3/s (100 cfs). In the coastal New Hampshire model, soil permeability was a significant delivery variable, but there was no in-stream loss for streams less than 2.83 m^3/s (100 cfs). The coastal NH model was calibrated on catchments containing only 1 point source and therefore, the coefficient for the source variable "point sources" coefficient in this model is potentially inaccurate. When the New England SPARROW is applied to the entire Lamprey basin (479 km^2, coastal NH) and sub-basins of the Lamprey, Oyster (coastal NH) and Ossipee (central NH) basins, the model over-predicts TN by 9 to 208 % (Fig. 1.5) and suggests that although the New England SPARROW is a better fit to New England, it is not an accurate model for coastal New Hampshire. The New England SPARROW was calibrated mainly on inland sites and does not accurately capture important features of coastal areas to accurately predict TN flux. This may be because the sources are over estimated or inaccurate, the land to water delivery term is over-estimated, the in stream loss coefficient is under-estimated or some combination of all of the above. The preliminary coastal NH model most accurately predicts TN flux in coastal NH (Legere 2007), but further refinement of the source variables and in-stream loss terms is necessary before it can reliably be applied to the Great Bay estuary. Population density and impervious surfaces are strong predictors of DIN and TDN flux in the Lamprey and Oyster basins (Section 3) and including these variables as source terms along with forests and water bodies which are significant areas of N retention (Section 3) would likely improve the predictive power of the coastal NH model.

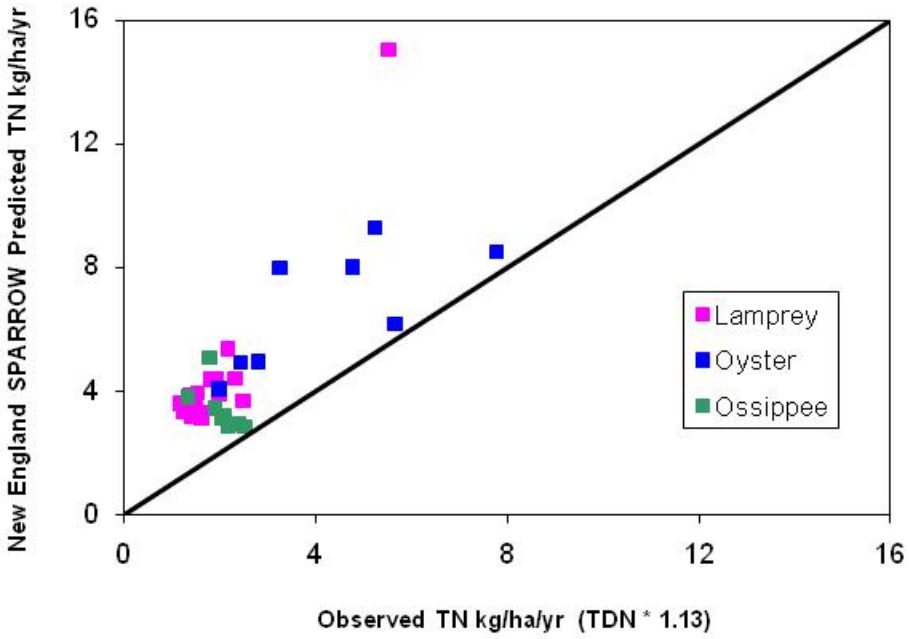

Fig. 1.5. New England SPARROW Predicted TN flux compared to observed TN flux from sub-basins within the Lamprey, Oyster and Ossipee basins in New Hampshire. For most sub-basins, TN was calculated as TDN x the median ratio of TN/TDN for the Lamprey basin and two of its sub-basins (WHB and NR).

Table 1.2. Loss coefficients for the National, Chesapeake, Mississippi, New England and preliminary coastal New Hampshire SPARROW models.

Model	Model Parameters	Coefficient Units	Parametric Coefficient
National SPARROW (Smith et al. 1997)	Land to water delivery ∝		
	Temperature (°F)	$°F^{-1}$	0.0196
	Soil permeability (cm/h)	h/cm	0.0442
	Stream density (km^2/km)	km^{-1}	0.0215
	In stream loss k_t		
	Mean Q $\leq$ 28.3 m^3/s	d^{-1}	0.3758
	28.3 m^3/s < Mean Q $\leq$ 283 m^3/s	d^{-1}	0.1233
	Mean Q > 283 m^3/s	d^{-1}	0.0406
Chesapeake SPARROW (Preston and Brakebill 1999)	Land to water delivery ∝		
	Soil permeability (in/h)	h/in	0.0754
	In stream loss k_t		
	Mean Q $\leq$ 5.66 m^3/s	d^{-1}	0.7595
	5.66 m^3/s < Mean Q $\leq$ 28.3 m^3/s	d^{-1}	0.3021
	Mean Q > 28.3 m^3/s	d^{-1}	0.0669
	Reservoir retention	d^{-1}	0.4145
Mississippi SPARROW (Alexander et al. 2000)	Land to water delivery ∝		
	Temperature (°F)	$°F^{-1}$	0.017
	Soil permeability (cm/h)	h/cm	0.036
	Stream density (km^2/km)	km^{-1}	0.043
	In stream loss k_t		
	Mean Q $\leq$ 28.3 m^3/s	d^{-1}	0.455
	28.3 m^3/s < Mean Q $\leq$ 283 m^3/s	d^{-1}	0.118
	283 m^3/s < Mean Q $\leq$ 850 m^3/s		0.051
	Mean Q > 850 m^3/s	d^{-1}	0.005
New England SPARROW (Moore et al. 2004)	Land to water delivery ∝		
	Natural log of soil permeability (cm/h)	h/cm	0.37
	In stream loss k_t		
	Mean Q $\leq$ 2.83 m^3/s	d^{-1}	0.78
Preliminary Coastal NH (Legere 2007)	Land to water delivery ∝		
	Natural log of soil permeability (cm/h)	h/cm	0.437
	In stream loss k_t		
	Mean Q $\leq$ 2.83 m^3/s	d^{-1}	0

1.2 NITROGEN ATTENUATION IN TERESTRIAL AND GROUNDWATER FLOWPATHS

Nitrogen attenuation and transport through terrestrial and groundwater flowpaths is rarely studied in totality from the specific source of N inputs (e.g. fertilizers, atmospheric deposition or septage) all the way through to discharge to a surface water body. Instead most studies have examined either N uptake in discrete compartments of the flow path or have determined the magnitude of total nitrogen inputs, outputs and retention at the watershed scale (Boyer et al. 2002; Howarth et al. 1996; Nixon et al. 1996; Williams et al. 2004), with fewer studies differentiating the actual sources of the exported nitrogen (Cole et al. 2006; Kroeger et al. 1999; Valiela 2000) or the specific retention rate for each input category (e.g. atmospheric, fertilizers, septage). Examining the loss of nitrogen through an entire flowpath is important as nitrogen attenuation processes such as denitrification may occur at each stage. When a reduction in the overall load of nitrogen to coastal waters or other water bodies is necessary, managers need to identify the contribution of the various sources of nitrogen to develop management plans. By estimating or directly measuring the contribution of each piece of the flowpath to N transformation and removal, it is may be possible to provide a more accurate estimate of total removal of nitrogen within the system (Cole et al. 2006; Valiela et al. 1997) and to identify the relative contribution of each N source (e.g. fertilizers, septage, atmospheric deposition) to N exported from the watershed.

1.2.1 Nitrogen Uptake in Forests and Soils

Forests and forest soils can be responsible for large amounts of terrestrial N retention. Tree species (Lovett et al. 2004; Hackl et al. 2004), stand age, land use history (Compton and Boone 2000), soil characteristics (Gunderson et al. 1998) and climate can all affect N cycling and uptake. Biomass N accumulation estimates were made by Goodale et al. (2002) based on the USDA's Forest Inventory and Analysis data and they estimate a net growth forest uptake rate of < 6.0 kg N/ha/yr for Rockingham County NH which is less than the 7.41 kg/ha/yr in wet deposition (median value for 2000-2009) received by the Lamprey basin. If we apply this rate to the 69.5% forest cover in the Lamprey basin we obtain a maximum forest uptake rate of 4.2 kg N/ha/yr for the basin. Studies where N is added to the soil surface at Harvard Forest (HF), MA have estimated N uptake rates for pine and hardwood stands. Pine and Hardwood control sites (8 kg N/ha/yr in deposition) had N uptake rates of 2.9 and 5.2 kg N/ha/yr respectively (Aber et al. 1998). Again, these rates are lower than the wet deposition received by the Lamprey basin and if we apply these rates to the forests in the entire basin, 2-3.6 kg N/ha/yr could be retained in forests and forest soils.

Soils can represent a large sink for terrestrial N and two-thirds or more of N in addition experiments can be incorporated into soil organic matter (Aber et al 1998). The mechanisms of this soil N retention are not fully understood and both biotic and abiotic mechanisms are plausible. Abiotic N retention averaged 19% of total N immobilization in semiarid grasslands

(Barrett et al. 2002). Other recent research suggests that abiotic N retention in a wide range of forest soils is non-existent and previous studies which show abiotic N retention in forests soils may have been misled due to an iron interference with NO_3 (Colman et al. 2007). ^{15}N tracer experiments at Harvard Forest show that added N is incorporated into the mineral soil within the first few years of addition. This mineral soil N then becomes a source of N for vegetation over a longer (8 year study) time period (Currie et al. 2004). In the Adirondacks (ADKs), NY a ^{15}N addition experiment showed that the mineral soil and forest floor were large sinks for added N over a 3 year period (Mitchell et al. 2001). Under ambient conditions (4.6-11.7 kg N/ha/yr of throughfall), these ADK sites retained from 3.6-4.7 kg N/ha/yr at 50 cm depth (Mitchell et al. 2001). In an old growth temperate forest in Chile, rates of N addition up to 160 kg N/ha/yr were predominately retained in soil organic matter (SOM) as identified by ^{15}N tracers and did not stimulate N leaching (Perakis et al 2005). Addition rates >160 kg N/ha/yr did stimulate N leaching and coarse roots and particulate matter were more important retainers of ^{15}N than SOM or any other measured N pool. Soil C:N ratios have been shown to influence N leaching where forest floor C:N ratios below 25 tend to promote nitrate leaching and full N retention occurs at C:N ratios above 30 (Gunderson et al. 1998). Dise et al. (1998) found that regardless of C:N ratio, at low levels of deposition (< 9 kg/ha/yr) nitrate leaching did not occur; at moderate to high deposition, nitrate leaching was related to soil C:N ratio.

1.2.2 Nitrogen attenuation in Groundwater flowpaths

Nitrogen inputs from atmospheric deposition, fertilizers, septic systems, organic horizons in wetlands can all contribute to N loading to surface water bodies if this added N is not retained or removed along the groundwater flow path. Wastewater from on-site septic systems can be a major source of nitrogen entering coastal waters through pathways such as groundwater flow and river discharge (Cole et al. 2007; Valiela et al. 1997). Estimating the actual removal of nitrogen along groundwater flow paths and the contribution of on-site septic systems to the total nitrogen load in receiving water bodies such as rivers and estuaries has become increasingly important with increased population density in coastal areas and continued reliance on septic systems for waste disposal (Valiela et al. 1997).

Using Models to Estimate Attenuation of Septic System Nitrogen

To estimate the total contribution of nitrogen from septic systems at the watershed scale, one first needs to evaluate the efficiency of a standard septic system and then determine the extent to which nitrogen is attenuated as it moves from the septic plume through the aquifer and discharges to a surface water body (or if there is even a hydrologic connection from the septic plume to the surface water body in question). The total input of N to septic systems depends on how many people are using septic systems for waste disposal. Once this is determined for the entire watershed, the typical amount of N removal in the septic tank, leach field and plume (Fig. 1.6) can be applied to the total N input to determine how much N reaches the aquifer. Actual measures of nitrogen removal in septic systems are not widely published and consequently an

estimate of 50% N loss is often used, but his number may over or underestimate the actual amount of nitrogen removed (Valiela et al. 1997; Frimpter et al. 1990). Nitrogen transport, transformations and removal through the aquifer depend largely on hydrologic flow paths, carbon availability, the microbial communities present and oxygen status. If nitrate, carbon, denitrifying bacteria are present and oxygen is depleted (even at micro-sites) nitrate can be removed through denitrification. While rates of N retention in septic tanks, leach fields and septic plumes will vary with septic system design and maintenance and specific site characteristics, nitrogen transport and attenuation through the aquifer may be the largest error term in modeling septic system inputs to surface waters since hydrologic flow paths from septic system to surface waters are often unknown.

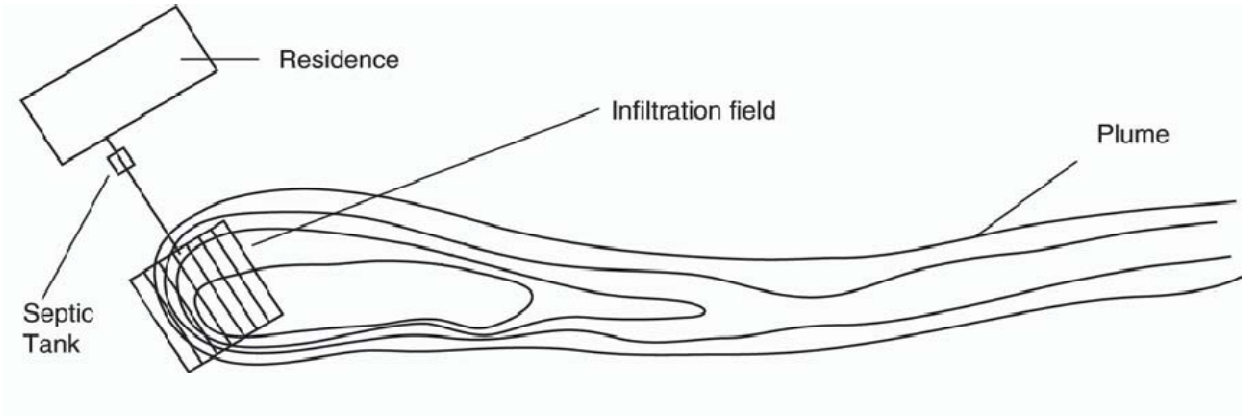

Fig. 1.6. Basic design of a septic system flowpath (from EPA, 2002)

Inputs to and Removal in the Septic Tank

The septic tank provides primary treatment of incoming wastewater, as most of the larger solids, oils, and greases are removed through sedimentation and flotation processes. The total input of nitrogen to a septic system ranges from 1.8 -7.4 kg N person^{-1} yr^{-1} (Table 1.3; Reay 2004; Valiela et al. 1997; Giblin and Gaines 1990). Overall, septic tanks are ineffective in removing nitrogen from wastewater and typically just convert organic nitrogen to ammonium (Gill et al. 2009) through a process known as mineralization. As such, most nitrogen within the tank is in the form of NH_4^+ and organic N (Beal et al. 2005: EPA 2002; Bunnell et al. 1999). Bunnell et al. (1999) sampled wastewater directly from 19 septic tanks located on the Atlantic Coastal Plain in New Jersey using peristaltic pumps. They found that, on average, the wastewater consisted of less than 1 mg/L of NO_3^-, and approximately 80 mg/L of NH_4^+ and 20 mg/L of organic N. Robertson and Cherry (1991) also noted that most of the nitrogen in the wastewater leaving the septic tanks in their study consisted of approximately 80% NH_4^+.

Reported removal rates within septic tanks are small, but have ranged from 1-20% (Table 1.3). Costa et al. (2002) found that approximately 1-3% of nitrogen entering a conventional septic tank was lost by the tank outlet. This study was conducted at the Massachusetts Alternative

Septic System Center, and may underestimate nitrogen removal within the tank, as the hydraulic loading rate in this study may be higher than in residential systems. Other studies have found slightly higher rates of attenuation. Valiela et al. (1997) found average losses within septic tanks to be about 5%, while other studies have found slightly higher rates of 10-20% (Reay 2004; Andreoli et al. 1979).

Removal in the Leach Field

Within the subsurface soil adsorption area, also known as the leach field, effluent leaving the septic tank undergoes physical, chemical and biological treatment. Most of the nitrogen entering the leach field is in the form of NH_3^+/NH_4^+ (~ 80%), which is then quickly transformed to NO_2^- and then NO_3^- by aerobic bacteria (a process known as nitrification) in the upper soil layers (Beal et al. 2005; EPA 2002). Most nitrogen leaving the leach field is in the form of nitrate (85-95%) (Costa et al. 2002). The NO_3^- can then be converted to N_2O and N_2 gases through the process of denitrification if the NO_3^- enters an anoxic zone with denitrifying bacteria and labile carbon. Denitrification is desirable as it represents a permanent loss of nitrogen from the system.

The actual amount of nitrogen removed within the leach field depends on characteristics such as the amount of soil moisture and organic matter present as well as the position of the leach field within the soil profile. Though raw sewage contains a significant amount of labile carbon, it has been shown to breakdown quickly in oxidized effluent once it has left the septic tank. As such, it may be unavailable in quantities needed to promote denitrification (Aravena and Robertson 1998).

Literature values on loss within the leach field range from approximately 15 – 35% (Table 1.3). An EPA report (2002) noted that Jenssen and Siegrist (1990) found that 20% of total nitrogen was removed in this part of the septic system, while Ricker et al. (1994) noted a loss of 15% in sandy soils and 25% in other soils. While Costa et al. (2002) noted a loss of approximately 20% within their septic system testing center, Valiela et al. (1997) found that on average, there was approximately a 35% loss of nitrogen within the leach field (Table 1.3).

Removal in the Effluent Plume and Estimates of Nitrogen Reaching Surface Water

The shape and movement of the effluent plumes in septic systems are influenced by climate, soil type, slope, geology, and hydrology of the watershed. Plumes can disperse broadly and deeply, and in the case of an elevated water table, and contaminate the groundwater before nitrogen has been attenuated. However, most effluent migrates in a long, narrow plume, which has been shown to reach up to 130 - 200 m in length in the upper soil layers before it reaches ground or surface water (Valiela et al. 1997; Robertson and Cherry, 1991).

Literature values for how much nitrogen is attenuated within the effluent plume and how much actually reaches nearby surface waters differed based on the length of the plume, distance

of septic system from a surface water body, and soil type. Valiela et al. (1997) estimated approximately 34% of the total nitrogen leaving the leach field is lost throughout the effluent plume (Table 1.4). A study by Robertson and Cherry (1991) on two septic systems serving single-family homes found that one system had a plume length of about 130 m and a plume width of about 10 m, while the other had a plume length of about 20 m before it entered a nearby river. While the site with the longer plume only lost approximately 50% of the nitrogen entering the plume by 130 m, almost complete nitrogen attenuation occurred within the last 2 m before it entered a nearby river at the site with the shorter plume. This loss is attributed to increased denitrification within the organic-rich riverbed sediments in this area. Beal et al. (2005) also noted the role of riparian zones in removing nitrogen from groundwater affected by wastewater in effluent plumes. Although not much nitrogen appears to be attenuated in the septic system itself, it appears that significant attenuation of septic system N can occur along the flow path from septic system to stream. Denitrification in riparian zones may play a particularly important role in this attenuation.

Table 1.3. Range from the literature of inputs or percent of nitrogen removed from incoming wastewater at each stage along the septic system flowpath.

Part of System Flowpath	Range	Source	Notes
Input	**1.8 – 7.4 kg N person^{-1}yr^{-1}**	Reay, 2004; Valiela et al. 1997; Giblin and Gaines, 1990	Must also include estimate of the number of houses and people per household
Septic Tank	**1-20% removal**	Reay, 2004; Costa et al. 2002; Valiela et al. 1997; Andreoli et al. 1979	May depend on loading rate of tank
Leach Field	**15-35% removal**	Costa et al. 2002; Valiela et al. 1997; Ricker et al. 1994; Jenssen and Siegrist, 1990	Depends on amount of carbon present for denitrification; soil type
Effluent Plume	**34-50% removal**	Valiela et al. 1997; Robertson and Cherry, 1991	Depends on soil type; length of plume

One example of a watershed model that determines nitrogen sources and delivery to an estuary is the Nitrogen Loading Model (NLM) developed by Valiela et al. (1997) for the Waquoit Watershed in Cape Cod, Massachusetts. The NLM determines the sources of nitrogen using land cover data such as the number of houses, occupancy rates, and the percentage of agricultural lands within the watershed and calculates nitrogen load to coastal waters by estimating loss terms along transport flowpaths. This model is most appropriate for watersheds with rural to suburban land cover where most of the nitrogen entering the estuary is from groundwater and the soils are very sandy. Valiela et al. (1997) estimated that wastewater from on-site septic systems was the largest source (approximately 48%) of the nitrogen load to

Waquoit Bay. Septic system load was estimated based on the population relying on septic systems for waste disposal and a human release rate of 4.8 kg N person^{-1} yr^{-1}. Losses of nitrogen through the septic tank, leach field, plume and aquifer were estimated using values presented in Table 1.4. This model estimates that a total of 86% of the nitrogen that enters a septic system is lost throughout the entire flow path (Table 1.4). One should note that the aquifer loss term in this model was determined by comparing the nitrogen concentration in groundwater near the water table under forested land parcels and N concentrations in groundwater about to enter the estuary. This approach does not consider the hydrologic connectivity of samples taken at the two locations and may under- or over-estimate actual N removal in the aquifer. Valiela et al. (1997) recognize that this is a rough guess and that direct measurements of nitrogen losses in aquifers are necessary. The next largest source of N to Waquoit Bay was atmospheric deposition (approximately 30%) and fertilizers accounted for approximately 15%. In a later study, this model was shown to have a significant linear relationship to field measured data of nitrate concentrations and $\delta^{15}N$ values indicated that the model can be used to estimate the amount of nitrogen entering coastal waters from wastewater (Valiela et al. 2000).

Table 1.4 Average concentration and percentage losses of wastewater nitrogen through a septic system (from Valiela et al. 1997).

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

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

To date, we have no data from the Lamprey or Oyster basins that quantifies actual N removal in septic tanks, leach fields, plumes and in aquifers. However, we do have nitrogen concentration data from two riparian zones in sub-urban basins (e.g. Fig. 1.7) where the only method for waste disposal is via septic systems. Upslope nitrogen concentrations are high where NO_3-N can exceed 4 mg L^{-1} (Fig. 1.8) and there is a decline in NO_3-N along the shallow groundwater flow path (Fig. 1.9). However, chloride concentrations also decline along the flow path indicating that the decline in concentration may result from dilution instead of NO_3^- uptake or denitrification, but further field experiments are needed to determine the ultimate fate of N in

this riparian zone. In the other sub-urban riparian zone Traer (2007) found that even though patterns of nitrogen concentration in ambient riparian groundwater suggested that denitrification might be occurring, field experiments showed that substantial N loss did not occur even with large amounts of added nitrate and dissolved organic carbon. Even though denitrification was not documented in this riparian zone, other studies have clearly shown that NO_3 concentrations are reduced as water moves though the riparian zone (Jordan et al. 1993; Simmons et al. 1992; Osborne and Kovacic 1993; McDowell et al 1992) and that > 90% NO_3^- removal is possible (Peterjohn & Correll 1984; Jordan et al 1993). Hydrologic flow paths, carbon availability, and soil conditions are important regulators of N flux through riparian zones (McDowell et al. 1992 and 1996, McClain et al.1994, Hedin et al. 1998, Burt and Pinay 2005). Sites that are most effective in NO_3 removal have permeable surface soils underlain by a shallow impermeable layer that produces shallow subsurface flow (Hill 1996). These are most common in low, flat areas of the landscape (Rassam 2005).

Despite the presumed importance of riparian denitrification at the watershed scale, empirical evidence documenting the importance of riparian denitrification at the watershed scale is weak (Smith et al. 2008). Rassam et al. 2008 attempted to mathematically conceptualize riparian NO_3 removal at the watershed scale and estimated that the riparian zone could remove up to 20% of the NO_3 load in South East Queensland, Australia. However, they admit that there is large uncertainty in the model. Others (Martin et al 1999; Hill 1996) indicate that the relative importance of the processes responsible for uptake (denitrification vs. plant uptake) are unknown, spatial variability in denitrification is high, and that removal is not sufficiently linked to groundwater flow paths in most studies. Despite the uncertainty in riparian removal, our data from the Lamprey basin (Fig. 1.8) clearly show elevated groundwater concentrations (in both deep and shallow groundwater) relative to stream water and suggest that septic systems are a likely source for this elevated groundwater NO_3^-. The shallow groundwater is undoubtedly connected to surface waters and evidence suggests most water in the Lamprey River has passed through near-surface or surficial flow paths such as riparian zones and wetlands prior to entering the channel. Isotopic composition of river water (mean δD -48.2 to -47.1‰, SD = 4.5) in both the headwaters and at the mouth at baseflow never approaches that of regional groundwater from bedrock wells (-58.4‰, SD=0.49) (Frades 2008 and McDowell unpublished). However, it is unclear whether deep groundwater pools are hydrologically connected to shallow groundwater and surficial flow paths.

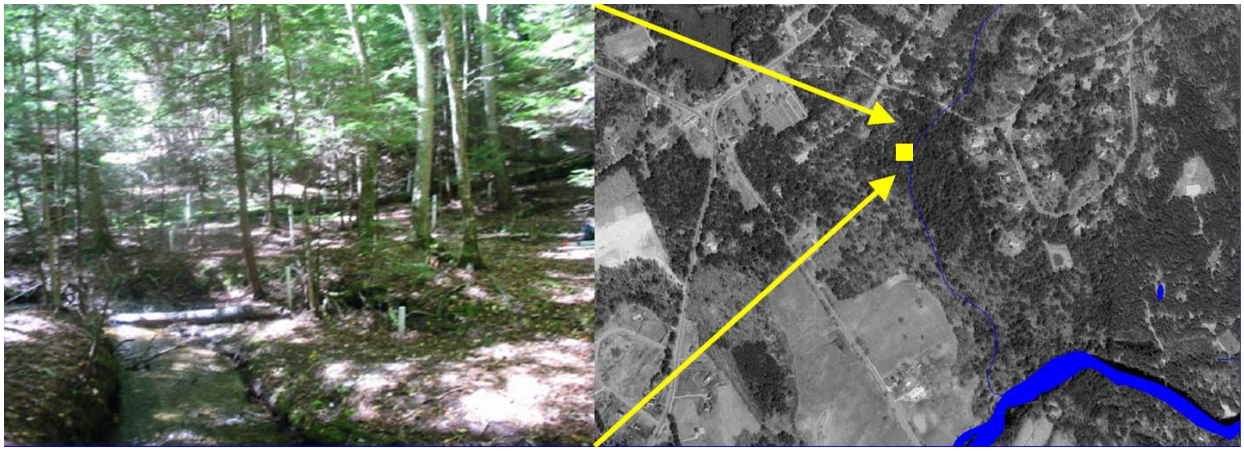

Fig. 1.7. Picture (left) of a riparian well field located in a sub-urban basin (aerial photo on right).

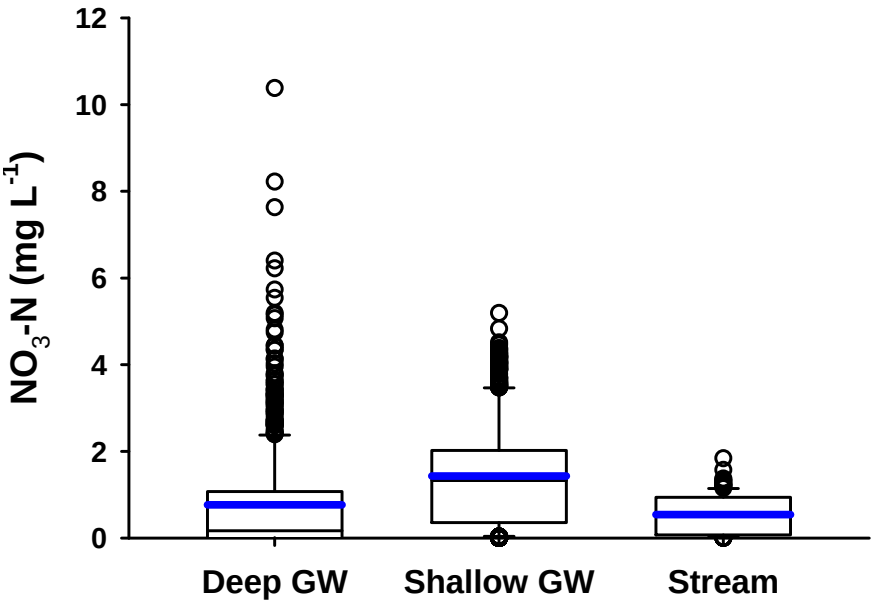

Fig. 1.8. Nitrate concentrations in deep groundwater (GW) sampled from private wells in the Lamprey basin, shallow GW sampled from riparian wells in two sub-urban basins and stream water that drains the two sub-urban basins as well as other sub-basins throughout the Lamprey basin where deep GW was sampled.

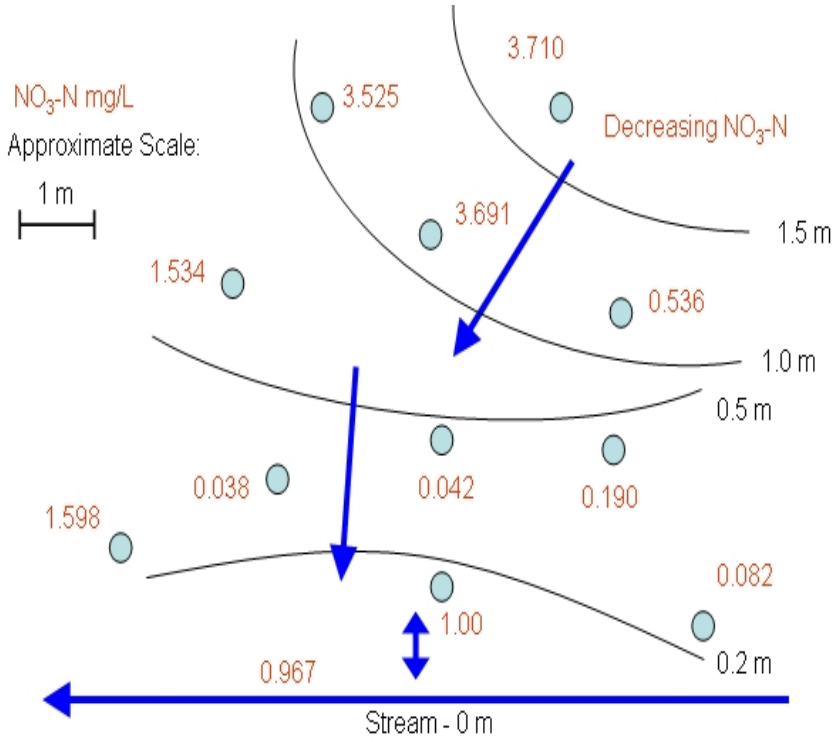

Fig. 1.9. Schematic of riparian well field located in one of our sub-urban basins where septic systems are the sole method for wastewater treatment. Mean monthly NO_3-N (mg L^{-1}) from 1/06 to 2/07 (in red) show a decline in nitrate along the riparian flow path (blue arrow).

Using Stable Isotopes to identify sources of N

Another method for determining the contribution of nitrate (the form of nitrogen that most consistently responds to increased N inputs; Task 3) from atmospheric deposition, fertilizers or human waste and manure is through the use of stable isotopes. By measuring the concentration of nitrate and the $\delta^{15}N$ values in streams and in groundwater at recharge zones, the relative importance of nitrate from various sources can be determined. Nitrate originating from manure has been shown to have a unique isotopic signature with $\delta^{15}N$ values ranging from +3 to +22 ‰, with nitrate from septic systems and sewage having a slightly narrower range of +7 to +20 ‰ (Table 1.5). Nitrate derived from other sources such as atmospheric deposition and fertilizers has lower $\delta^{15}N$ values (Table 1.5; Cole et al. 2006; Jin et al. 2004; Mayer et al. 2002; Valiela et al. 2000; Aravena and Robertson 1998).

Several studies to date have made use of this stable isotope method. Cole et al. (2006) found $\delta^{15}N$ values ranging from -6 to +10 ‰ in groundwater entering freshwater ponds and estuaries in Cape Cod, Massachusetts suggesting natural and anthropogenic sources at different locations. They also found a significant positive relationship ($r^2 = 0.46$, $p < 0.001$) between nitrate concentration and $\delta^{15}N$ values, with higher values in more developed sub-watersheds suggesting a larger influence for human waste. Jin et al. (2004) measured nitrate concentrations

and $\delta^{15}N$ values from 21 public and domestic wells in residential and agricultural areas in and around Hangzhou City, China. Groundwater in this area has been shown to exceed nitrate values of 10 mg NO_3-N L^{-1}. Values of $\delta^{15}N$ were found to range from +8 to over +35 ‰. These $\delta^{15}N$ values combined with knowledge of the predominant land use in the area indicated that the main source of nitrate in the groundwater in agricultural areas was manure and in residential areas was human wastewater (Jin et al. 2004).

Though $\delta^{15}N$ values have been used in some studies to identify sources of nitrate to a waterbody (Cole et al. 2006, Jin et al. 2004), the overlap of $\delta^{15}N$ values from different sources may lead to some error in identification of the most important nitrate sources. However, by pairing the use of stable nitrate isotopes with stable oxygen isotopes, nitrate sources can be more positively identified. Mayer et al. (2002) measured the $\delta^{15}N$ values and $\delta^{18}O$ values of riverine nitrate at the outlet of 16 watersheds throughout New England and were able to classify common sources of nitrate by the unique isotopic nitrogen and oxygen signatures (Fig. 1.10). They found that these isotopic signatures were related to land use. Nitrate in the more forested watersheds was thought to be derived from soil nitrification processes, as they had lower $\delta^{15}N$ values (~ +5) and higher $\delta^{18}O$ values (+16.7 to +18.5), while watersheds with more agricultural and urban land uses tended to have higher $\delta^{15}N$ values (+6 to +9) and lower $\delta^{18}O$ values (~13) thought to be derived from a mix of sewage and manure.

Table 1.5. Range of $\delta^{15}N$ and $\delta^{18}O$ values from different sources of nitrate.

Source of Nitrate	**$\delta^{15}N$ values (‰)**	**$\delta^{18}O$ values (‰)**
Atmospheric Deposition	-4 to +6 (Cole et al. 2006) -10.5 to +2 (Jin et al. 2004) -10 to +8 (Mayer et al. 2002) +2 to +8 (Valiela et al. 2000)	+25 to +70 (Mayer et al. 2002)
Soil Organic Nitrogen	-3 to +5 (Mayer et al. 2002)	
Fertilizers	-4 to +4 (Cole et al. 2006) 0 ± 3 (Mayer et al. 2002) -3 to +3 (Valiela et al. 2000) -5 to +5 (Aravena and Robertson, 1998)	+22 ± 3 (Mayer et al. 2002)
Manure	+10 to +22 (Jin et al. 2004) +7 to +20 (Mayer et al. 2002) +3 to +5 (Valiela et al. 2000) +8 to +21 (Aravena and Robertson 1998)	< +15 (Mayer et al. 2002)
Sewage	+10 to +17 (Jin et al. 2004) +7 to +20 (Mayer et al. 2002)	< +15 (Mayer et al. 2002)
Septic Systems	+10 to +20 (Cole et al. 2006) +7 to +15 (Jin et al. 2004) +10 to +20 (Valiela et al. 2000)	< +15 (Mayer et al. 2002)

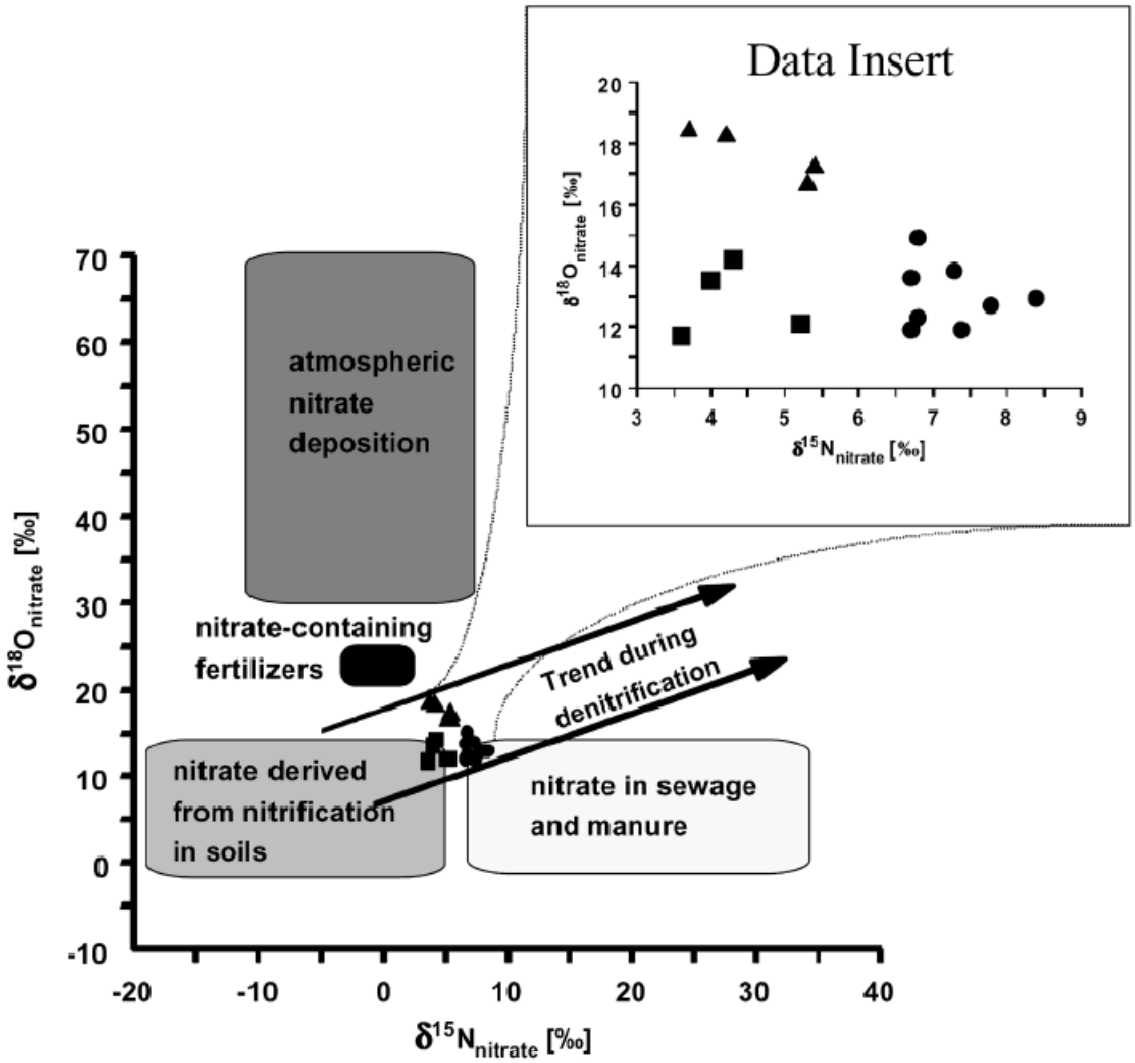

Fig. 1.10 Mean $\delta^{15}N$ values and $\delta^{18}O$ values of riverine nitrate from 16 watersheds in New England (taken from Mayer et al. 2002).

We have preliminary $\delta^{15}N$ and $\delta^{18}O$ values for nitrate in different water types in the Lamprey basin (Fig. 1.11). Values of $\delta^{15}N$ in the urban stream water and main stem of the Lamprey River ranged from +8 to +11 ‰ and suggest the source of nitrate is either human waste or manure. In the Lamprey River, both sources are plausible given that developed areas are inter-dispersed with agriculture. In the urban basin, the most likely source of nitrate is human waste despite the fact that the basin is entirely sewered. Agriculture covers less than 3% of the basin whereas urban areas cover 27% and illicit discharges have been historical problems in this basin. Field observations suggest that illicit discharges or leaky sewer pipes may still be an issue. The suburban stream water ranged in $\delta^{15}N$ from -4 to +14 ‰ and the shallow suburban

groundwater ranged from -6 to +17 ‰ suggesting that a combination of nitrified NH4 from fertilizers and nitrate from human or animal waste are plausible. One clear result from this isotopic data is that none of the streams or groundwater reflects unaltered deposition (Fig. 1.11).

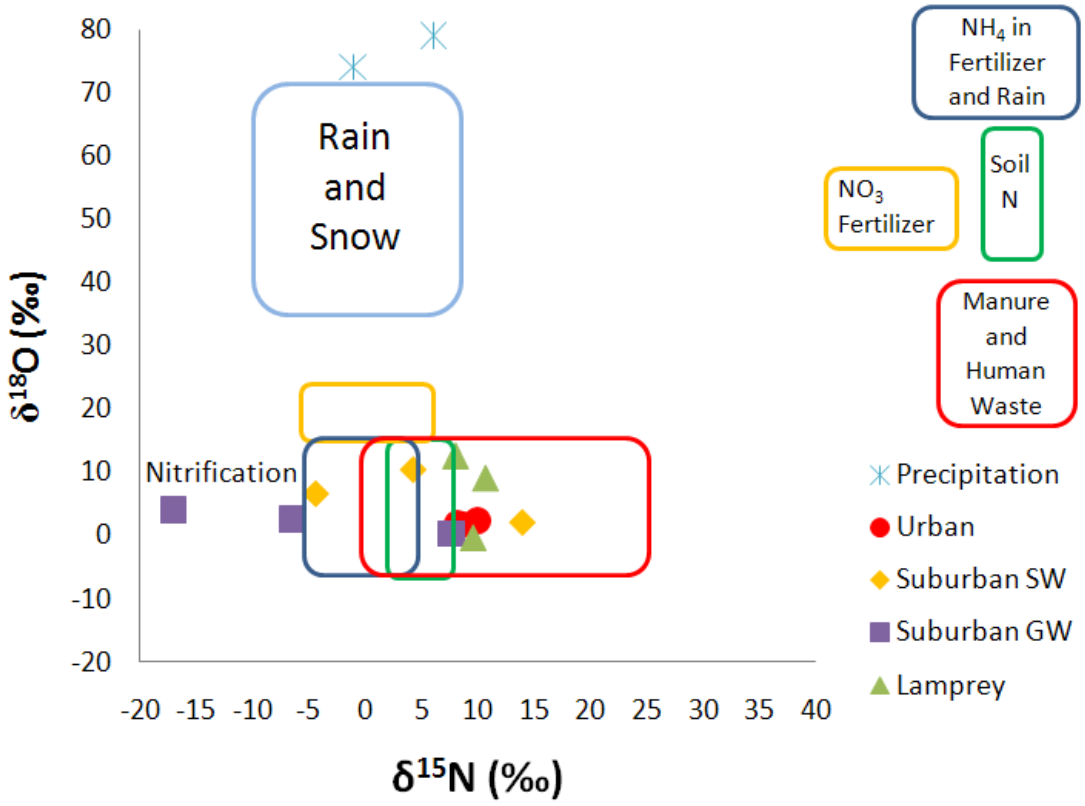

Fig. 1.11. Nitrate δ ^{15}N and δ ^{18}O values of samples collected from different water types in the Lamprey basin on three different sample dates. Source boxes are approximated from Kendall (1998). Water types are precipitation, stream water from an urban basin (completely sewered), sub-urban stream water (SW; from a basin served by septic systems), sub-urban groundwater (GW) collected from shallow riparian wells and from the main stem of the Lamprey River in Durham, NH.

2.0 ATMOSPHERIC DEPOSITION OF TOTAL NITROGEN

Over the last century, increases in emissions of nitrogen to the atmosphere (as a by-product of fossil fuel combustion) have led to increases in atmospheric concentrations of nitrogen as well as significant increases in atmospheric deposition (Holland et al. 2005; Vitousek et al. 1997). Total atmospheric deposition of N is the combination of both wet and dry deposition. Wet deposition is the portion of N dissolved in cloud droplets and deposited during rainfall and snowfall events (or other forms of precipitation such as hail). Dry deposition is the amount of N that settles as aerosols, dust or other deposits on surfaces during periods of no precipitation. Dry deposition of nitrogen also includes direct gaseous uptake or absorption of various N-containing gases (e.g. HNO_3, NO_y) that interact with surfaces such as vegetation. Atmospheric deposition can account for a large portion of N inputs to watersheds (Castro et al. 2003; Lovett and Rueth 1999; Ollinger et al. 1993) and even though wet deposition has received the most attention, dry deposition can also be an important component of watershed nitrogen budgets (Ollinger et al. 1993).

2.1 MEASUREMENTS OF WET DEPOSITION OF NITROGEN

Wet deposition can be directly measured through the collection and analysis of precipitation. Wet deposition collectors are typically automated samplers with a lid that closes during dry periods. Many measurements of precipitation chemistry are made with bulk deposition collectors, which are typically funnel collectors attached to a collecting bottle. Bulk deposition represents the sum of wet deposition plus some fraction of dry deposition. Bulk precipitation chemistry is thus typically higher in concentration and flux than wet-only precipitation (e.g. McDowell et al. 1990). Deposition (kg/ha) is typically calculated as the precipitation-weighted mean concentration (mg/L) in a given collector multiplied by the best available estimate of precipitation amount (mm) over the period of interest.

The National Atmospheric Deposition Program/National Trends Network (NADP/NTN) is the nation's major source for wet deposition data and provides a long-term record of precipitation chemistry. This program has multiple sites throughout the US that are located away from urban areas and point sources of pollution. There is one site located in NH (Hubbard Brook in Grafton County) and several are located in neighboring states (eight sites in ME, three sites in MA, and two sites in VT). The NADP/NTN provides precipitation-weighted concentrations of NO_3^- and NH_4^+ and wet deposition of inorganic N. Data are available for download directly from the NADP website (http://nadp.sws.uiuc.edu/data/ntndata.aspx) in both calendar (Jan 1 to December 31) and water year (October 1 to September 30) formats, though there is a delay of a few months to a year. Data are currently available from 1978 (for some sites) through 2008 and we have provided data for the 8 NADP/NTN sites closest to Great Bay (Fig. 2.1; Tables 2.1 and 2.2). There has been a slight decrease in wet deposition of inorganic N (Fig. 2.2) and a more noticeable decrease in the concentration of inorganic N in precipitation (Fig. 2.3) at these sites over the last twenty years. The decrease in concentration is largely due to emission reduction

regulations set forth under the Clean Air Act Amendments. Inorganic N deposition has ranged from approximately 2 - 8 kg N ha^{-1} yr^{-1} and precipitation-weighted inorganic N concentrations have ranged from approximately 0.2 to 0.7 mg L^{-1} among the sites (Fig. 2.2 and 2.3).

To quantify N deposition inputs on a more local level (specifically for the Lamprey River basin, NH Water Resource Research Center (WRRC) staff (including authors of this report) have collected precipitation samples since November of 2003 at the Thompson Farm (TF) AIRMAP (http://airmap.unh.edu/) site located in Durham, NH. Wet-only precipitation samples are collected on a weekly to storm event basis and analyzed for TDN, NO_3-N, NH_4-N and DON in the Water Quality Analysis Laboratory (WQAL) at UNH. Most samples (Nov 2003 to September 2007) were collected with an Aerochem Metrics wet-dry collector (Fig. 2.4). More recent samples (since October 2007) were collected using an N-CON wet-only collector (Fig. 2.5). Annual (both calendar year and water year) precipitation-weighted concentrations of TDN, DIN and DON (Table 2.3) are multiplied by the annual precipitation amount recorded by the Climate Reference Network (CRN; http://www.ncdc.noaa.gov/crn/) station co-located at TF (CRN station Durham 2 SSW; Fig. 2.6) to calculate annual wet deposition. From 2004 to 2009, annual wet TDN deposition at TF ranged from 4.1 to 6.6 kg N ha^{-1} yr^{-1} (Table 2.4) and was within the range reported for the closest NADP/NTN stations (Fig. 2.1; Table 2.2).

Fig. 2.1. Map of the 8 NADP/NTN (wet inorganic N deposition since 1978) closest to the Great Bay watershed and the 5 CASTNET (wet, dry and total inorganic N deposition since 1991), stations located in New England. Our site (TF) where we have measured wet deposition of inorganic and organic N since 2004 is also shown.

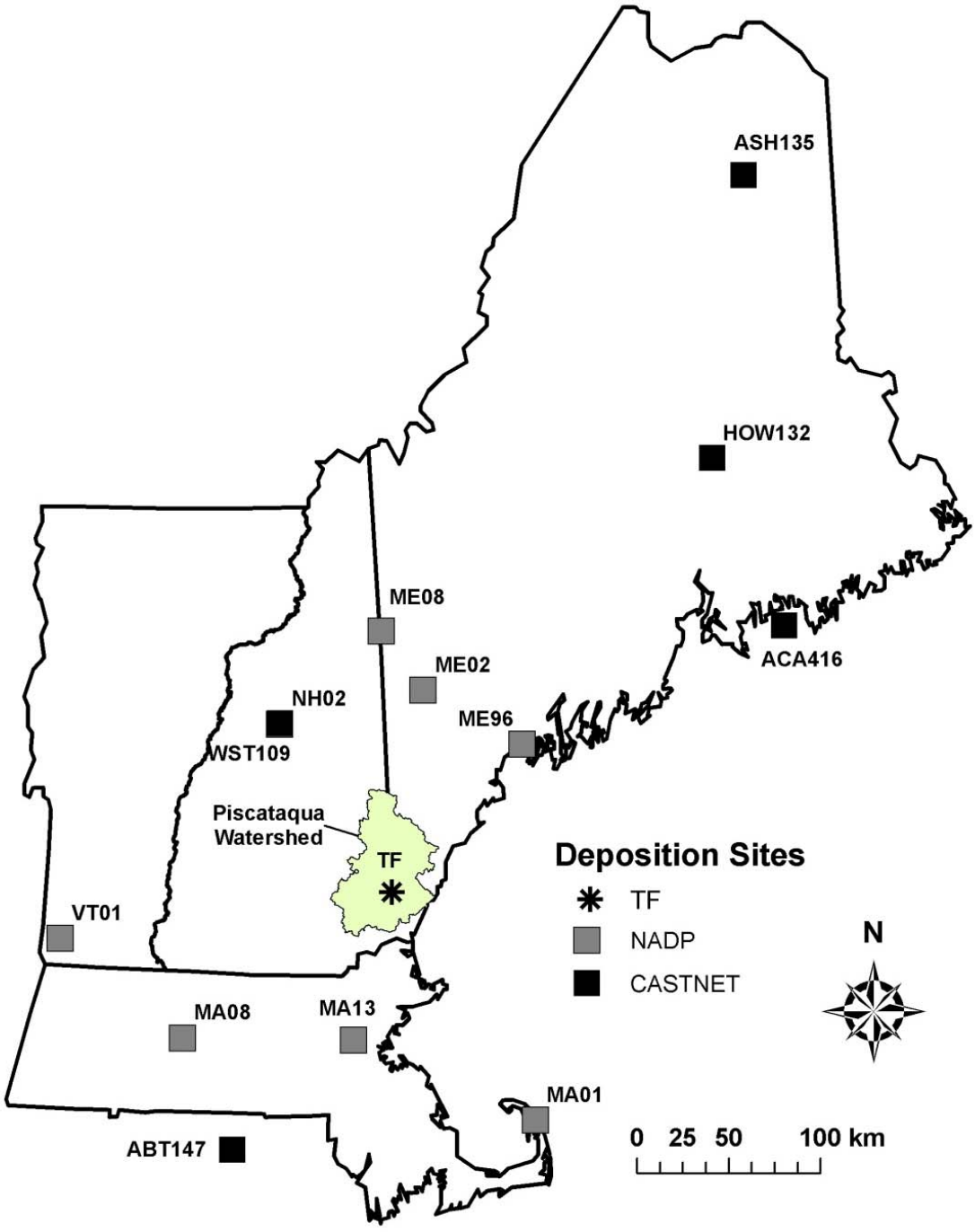

Table 2.1: Annual Precipitation-Weighted Mean Inorganic Nitrogen Concentration (mg N/L) for Calendar Years (CY) and Water Years (WY) at the 8 NADP/NTN sites in New England closest to Great Bay.

Site ID	**MA01**		**MA08**		**MA13**		**ME02**		**ME08**		**ME96**		**NH02**		**VT01**	
Site Name	North Atlantic Coastal		Quabbin Reservoir		East		Bridgton		Gilead		Casco Bay-Wolfe's Neck Farm		Hubbard Brook		Bennington	
County	Barnstable		Franklin		Middlesex		Cumberland		Oxford		Cumberland		Grafton		Bennington	
State	MA		MA		MA		ME		ME		ME		NH		VT	
Elev (m)	41		306		18		222		212		15		250		305	
Lat	41.9758		42.3925		42.3839		44.1075		44.4003		43.8325		43.9433		42.8761	
Long	-70.0247		-72.3444		-71.2147		-70.7289		-71.0098		-70.0645		-71.7029		-73.1633	
Year	**CY**	**WY**	**CY**	**WY**	**CY**	**WY**	**CY**	**WY**	**CY**	**WY**	**CY**	**WY**	**CY**	**WY**	**CY**	**WY**
1979													0.39	0.41		
1980													0.54	0.51		
1981							0.39	0.39					0.46	0.50	0.64	0.71
1982	0.35	0.34	0.45	0.42	0.37	0.34	0.44	0.39					0.49	0.42	0.49	0.50
1983	0.22	0.25	0.50	0.62	0.26	0.35	0.23	0.26					0.40	0.49	0.44	0.51
1984	0.25	0.24	0.55	0.50	0.29	0.25	0.32	0.27					0.42	0.39	0.54	0.49
1985	0.36	0.36	0.60	0.68	0.47	0.44	0.42	0.44					0.47	0.42	0.63	0.52
1986	0.35	0.40	0.44	0.49	0.38	0.49	0.31	0.34					0.38	0.42	0.46	0.57
1987	0.33	0.16	0.38	0.33	0.32	0.25	0.23	0.18					0.38	0.34	0.52	0.55
1988	0.32	0.41	0.36	0.41	0.35	0.43	0.28	0.31					0.47	0.45	0.55	0.55
1989	0.34	0.29	0.48	0.43	0.40	0.34	0.34	0.31					0.53	0.52	0.59	0.52
1990	0.31	0.34	0.38	0.49	0.31	0.36	0.37	0.45					0.37	0.45	0.48	0.56
1991	0.40	0.39	0.38	0.33	0.34	0.33	0.31	0.29					0.43	0.37	0.58	0.53
1992	0.33	0.31	0.56	0.54	0.46	0.40	0.39	0.36					0.44	0.45	0.61	0.56
1993	0.27	0.30	0.44	0.43	0.37	0.41	0.35	0.33					0.42	0.44	0.49	0.53
1994	0.30	0.30	0.54	0.57	0.41	0.44	0.40	0.42					0.49	0.46	0.57	0.56
1995	0.40	0.44	0.46	0.49	0.35	0.36	0.30	0.36					0.49	0.57	0.63	0.68
1996	0.29	0.29	0.41	0.39	0.31	0.37	0.41	0.35					0.39	0.40	0.49	0.47
1997	0.41	0.38	0.47	0.45	0.48	0.35	0.37	0.39					0.49	0.42	0.53	0.52
1998	0.23	0.22	0.39	0.37	0.29	0.28	0.29	0.29			0.27	0.23	0.42	0.39	0.55	0.52
1999	0.26	0.30	0.31	0.30	0.32	0.33	0.30	0.29	0.20		0.29	0.32	0.35	0.38	0.35	0.37
2000	0.34	0.30	0.44	0.44	0.43	0.44	0.35	0.39	0.28	0.29	0.28	0.34	0.41	0.46	0.52	0.51
2001	0.30	0.30	0.51	0.46	0.35	0.34	0.34	0.28	0.32	0.25	0.31	0.23	0.40	0.33	0.44	0.43
2002	0.32	0.36	0.40	0.48	0.41	0.48	0.34	0.38	0.31	0.34	0.30	0.36	0.40	0.44	0.55	0.59
2003	0.39	0.33	0.37	0.39	0.27	0.29	0.27	0.30	0.27	0.28	0.23	0.25	0.29	0.32	0.40	0.44
2004	0.25	0.29	0.36	0.33	0.34	0.30	0.24	0.25	0.21	0.23	0.37	0.32	0.35	0.35	0.45	0.42
2005	0.21	0.22	0.29	0.38	0.35	0.42	0.22	0.26	0.20	0.22	0.21	0.28	0.27	0.32	0.33	0.43
2006	0.23	0.21	0.32	0.27	0.34	0.31	0.22	0.20	0.21	0.20	0.23	0.19	0.25	0.23	0.36	0.29
2007	0.24	0.26	0.48	0.39	0.30	0.28	0.32	0.24	0.23	0.21	0.31	0.30	0.37	0.32	0.45	0.45
2008	0.20	0.20	0.31	0.35	0.32	0.35	0.25	0.30	0.16	0.18	0.21	0.21	0.26	0.30	0.33	0.38
Min	0.20	0.16	0.29	0.27	0.26	0.25	0.22	0.18	0.16	0.18	0.21	0.19	0.25	0.23	0.33	0.29
Max	0.41	0.44	0.60	0.68	0.48	0.49	0.44	0.45	0.32	0.34	0.37	0.36	0.54	0.57	0.64	0.71

Table 2.2: Annual Inorganic Nitrogen Wet Deposition (kg/ha/yr) for Calendar Years (CY) and Water Years (WY) at the 8 NADP/NTN sites in New England closest to Great Bay.

Site ID	**MA01**		**MA08**		**MA13**		**ME02**		**ME08**		**ME96**		**NH02**		**VT01**	
Site Name	North Atlantic Coastal		Quabbin Reservoir		East		Bridgton		Gilead		Casco Bay-Wolfe's Neck Farm		Hubbard Brook		Bennington	
County	Barnstable		Franklin		Middlesex		Cumberland		Oxford		Cumberland		Grafton		Bennington	
State	MA		MA		MA		ME		ME		ME		NH		VT	
Elev (m)	41		306		18		222		212		15		250		305	
Lat	41.9758		42.3925		42.3839		44.1075		44.4003		43.8325		43.9433		42.8761	
Long	-70.0247		-72.3444		-71.2147		-70.7289		-71.0098		-70.0645		-71.7029		-73.1633	
Year	**CY**	**WY**	**CY**	**WY**	**CY**	**WY**	**CY**	**WY**	**CY**	**WY**	**CY**	**WY**	**CY**	**WY**	**CY**	**WY**
1979													4.76	4.85		
1980													5.03	4.84		
1981							4.19	3.87					6.92	7.05	4.69	3.32
1982	4.67	3.80	4.65	3.33	3.54	2.59	3.67	3.80					4.96	4.78	4.24	4.62
1983	3.57	3.77	7.48	8.06	3.52	3.83	3.07	2.73					5.19	5.45	5.27	5.49
1984	3.33	3.90	7.28	7.65	3.50	3.70	3.52	3.66					5.25	5.62	6.20	6.00
1985	4.38	4.14	6.55	7.10	4.30	3.79	3.21	3.20					4.73	4.09	4.93	4.19
1986	4.21	3.99	5.03	5.33	4.10	4.58	3.08	3.46					4.71	5.41	4.94	5.80
1987	1.89	1.18	3.74	3.58	3.30	3.06	1.99	1.50					4.22	3.73	4.77	4.88
1988	3.11	3.52	3.43	3.84	3.22	3.75	2.57	2.77					4.35	4.26	5.73	6.05
1989	3.70	3.37	6.49	5.52	4.43	3.72	3.44	3.17					6.79	6.21	6.24	5.73
1990	3.08	3.41	5.45	6.22	4.13	4.28	4.04	4.19					5.74	6.61	5.97	6.42
1991	4.09	3.82	5.29	5.14	3.50	3.80	3.35	3.38					5.14	4.72	7.00	6.35
1992	3.48	3.39	5.89	5.97	4.74	3.90	3.50	3.55					4.98	5.45	5.13	5.21
1993	3.47	3.20	5.13	4.56	3.36	3.69	3.23	2.71					4.49	4.35	4.96	5.14
1994	4.11	4.77	7.42	8.32	4.52	5.01	3.78	4.30					5.22	5.55	5.52	6.10
1995	4.41	4.00	5.24	4.54	2.91	2.78	2.78	2.60					5.66	5.19	6.14	5.74
1996	4.31	4.34	6.79	6.67	4.37	4.50	5.46	4.52					5.97	5.99	6.43	6.21
1997	4.48	4.53	4.90	5.70	4.22	4.15	4.04	5.02					5.23	5.39	4.59	4.92
1998	2.95	3.25	4.54	4.27	4.01	4.09	3.24	3.20			3.49	2.65	5.22	4.72	5.60	5.37
1999	2.71	2.51	3.94	3.91	3.20	3.21	3.26	3.33			3.49	3.46	4.25	4.63	4.32	4.35
2000	4.58	3.96	5.79	5.95	4.67	4.52	3.83	3.90	3.27	3.22	3.41	3.75	5.08	5.40	6.51	6.70
2001	2.87	3.60	4.88	4.77	3.22	3.68	2.36	2.29	2.11	2.02	2.59	2.35	3.60	3.26	3.67	3.74
2002	4.45	3.91	4.38	4.41	4.62	4.18	3.36	3.45	3.10	3.12	3.30	3.59	4.62	4.80	7.19	6.78
2003	4.36	4.06	5.17	5.38	3.30	3.67	3.06	2.81	3.29	2.68	2.47	2.37	4.10	3.78	4.87	4.84
2004	2.77	3.10	4.15	3.94	3.92	3.71	2.41	2.82	2.15	2.78	3.72	3.67	4.26	4.77	4.90	5.38
2005	2.80	2.76	4.60	4.66	4.41	4.34	3.23	3.06	2.88	2.65	3.42	3.33	4.26	4.24	4.18	4.31
2006	3.02	2.94	4.48	4.74	4.56	4.62	2.87	2.85	2.81	2.92	3.47	3.30	3.44	3.37	4.20	4.05
2007	2.73	2.92	5.47	4.32	2.94	2.98	3.38	2.71	2.69	2.72	3.36	3.51	4.68	4.12	5.42	5.11
2008	2.63	2.69	6.08	6.86	4.65	4.87	3.80	4.61	2.27	2.31	3.10	3.10	4.08	4.79	4.45	4.93
Min	1.89	1.18	3.43	3.33	2.91	2.59	1.99	1.50	2.11	2.02	2.47	2.35	3.44	3.26	3.67	3.32
Max	4.67	4.77	7.48	8.32	4.74	5.01	5.46	5.02	3.29	3.22	3.72	3.75	6.92	7.05	7.19	6.78

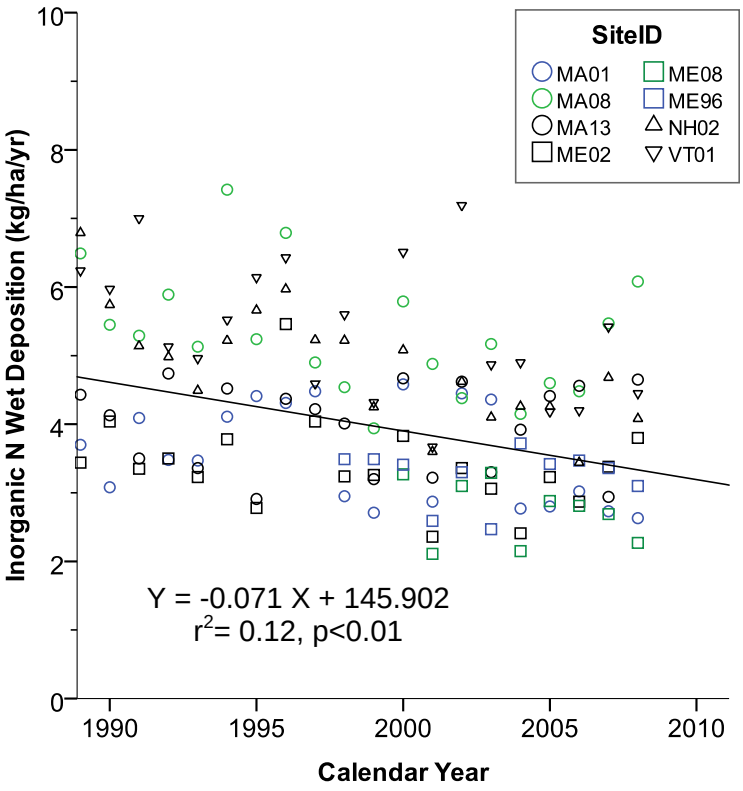

Fig. 2.2. Annual (calendar year) inorganic N wet deposition at 8 NADP/NTN sites in New England closest to Great Bay since 1990. Regression line shows decreasing trend over time among the 8 NADP/NTN sites.

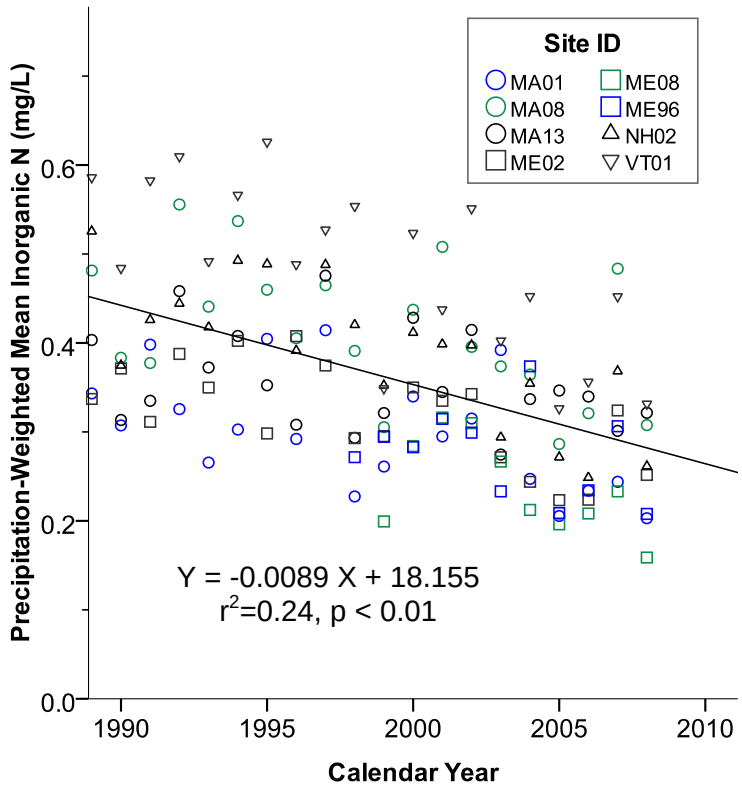

Fig. 2.3. Annual (Calendar Years) Precipitation-Weighted Mean Inorganic Nitrogen Concentration (mg N/L) at the 8 NADP/NTN sites in New England closest to Great Bay since 1990. Regression line shows decreasing trend over time among the 8 NADP/NTN sites.

Fig. 2.4. Picture of Aerochem Metrics wet-dry collector.

Fig. 2.5. Picture of N-CON wet-only collector.

Fig. 2.6. Picture of the Climate Reference Network (CRN) station (Durham 2 SSW; left side of picture) and the AIRMAP tower used to measure air quality (right side of picture) at Thompson Farm in Durham, NH.

Table 2.3. Annual precipitation-weighted concentrations of total dissolved nitrogen (TDN), dissolved inorganic nitrogen (DIN) and dissolved organic nitrogen (DON) at Thompson Farm (TF) in Durham, NH). Both calendar years (CY) and water years (WY) are reported.

Year	Collector ID	N Samples		TDN (mg/L)		DIN (mg/L)		DON (mg/L)		% DIN to TDN	
		CY	WY	CY	WY	CY	WY	CY	WY	CY	WY
2004	ppt1Thomp	63	59	0.365	0.352	0.336	0.326	0.029	0.026	92%	93%
2005	ppt1Thomp	87	77	0.324	0.384	0.309	0.365	0.015	0.020	95%	95%
2006	ppt1Thomp	87	87	0.289	0.278	0.270	0.259	0.019	0.019	94%	93%
2007	ppt1Thomp	89	81	0.358	0.311	0.338	0.298	0.020	0.013	94%	96%
2008	ppt2Thomp	93	105	0.401	0.413	0.371	0.383	0.029	0.030	93%	93%
2009	TF2	62	64	0.336	0.339	0.325	0.327	0.011	0.012	97%	96%
Minimum		62	59	0.289	0.278	0.270	0.259	0.011	0.012	92%	93%
Maximum		93	105	0.401	0.413	0.371	0.383	0.029	0.030	97%	96%

Table 2.4. Annual wet deposition of total dissolved nitrogen (TDN), dissolved inorganic nitrogen (DIN) and dissolved organic nitrogen (DON) in kg N ha^{-1} yr^{-1} at Thompson Farm (TF) in Durham, NH). Both calendar years (CY) and water years (WY) are reported.

Year	Collector ID	N Samples		TDN (kg/ha/yr)		DIN (kg/ha/yr)		DON (kg/ha/yr)		% DIN to TDN	
		CY	WY	CY	WY	CY	WY	CY	WY	CY	WY
2004	ppt1Thomp	63	59	4.23	4.23	3.89	3.92	0.34	0.31	92%	93%
2005	ppt1Thomp	87	77	5.35	5.02	5.10	4.76	0.25	0.26	95%	95%
2006	ppt1Thomp	87	87	4.79	5.06	4.48	4.71	0.31	0.35	94%	93%
2007	ppt1Thomp	89	81	4.29	4.08	4.05	3.91	0.24	0.17	94%	96%
2008	ppt2Thomp	93	105	6.58	6.65	6.10	6.16	0.48	0.48	93%	93%
2009	TF2	62	64	4.44	4.54	4.29	4.38	0.15	0.16	97%	96%
Minimum		62	59	4.23	4.08	3.89	3.91	0.15	0.16	92%	93%
Maximum		93	105	6.58	6.65	6.10	6.16	0.48	0.48	97%	96%

2.2 MEASUREMENTS OF DRY AND TOTAL DEPOSITION OF NITROGEN

While dry deposition is difficult to measure, it can be calculated based on the measured air concentration of a chemical species and the deposition velocity of that species. Atmospheric concentrations are often measured through the use of filter packs or in some cases real time continuous measurements are made or discrete air samples are collected in cans. The Aerochem Metrics wet-dry collector does have a "dry" collector which is a bucket exposed to the atmosphere during dry periods. However, dry deposition is not deposited on plastic as it is on vegetation or pavement and therefore measuring atmospheric concentrations and modeling deposition velocity is the preferred method to directly quantify dry deposition.

The Clean Air Status and Trends Network (CASTNET), previously known as the National Dry Deposition Network (NDDN), is a monitoring network managed by the U.S. EPA that provides data on ambient air quality and dry deposition data at the national level, with five sites in New England (one site in Woodstock NH, three sites in ME and one site in CT) where urban influences are minimal. CASTNET provides hourly concentrations of NOy gases, weekly ambient concentrations of HNO_3 gas and NO_3^- and NH_4^+ concentrations of particles collected by filter packs. These data along with deposition velocities modeled by CASTNET are used to calculate annual dry deposition of nitrogen at CASTNET sites. Annual wet deposition of N at CASTNET sites is interpolated from nearby NADP/NTN sites and summed with dry deposition to estimate total N deposition. Data is currently available for download directly from the CASTET site (http://www.epa.gov/castnet/) from 1991 (at some sites) to January 2010. At the New England CASTNET stations, total inorganic N deposition has ranged from 2.5 to 8.1 kg ha^{-1} yr^{-1} (Table 2.5, Fig. 2.7) with the highest values recorded in CT. Dry deposition comprised less than half of the total inorganic N deposition at these sites (Fig. 2.8) and ranged from 4 % (Woodstock, NH) to 41 % (Abington, CT) of the total inorganic N deposition.

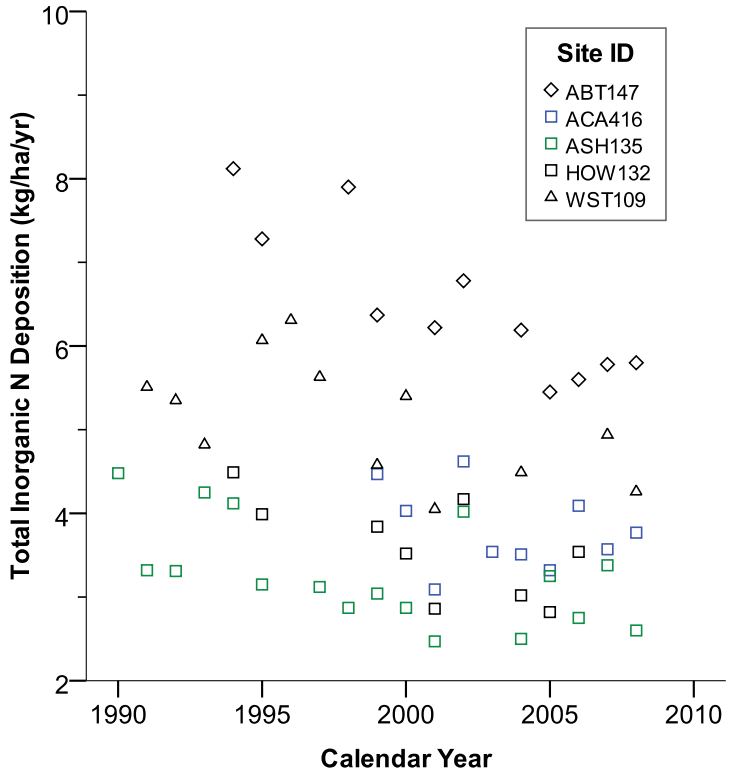

Fig. 2.7. Annual (available for calendar year only) total (wet and dry) inorganic nitrogen deposition (kg ha^{-1} yr^{-1}) at the 5 CASTNET sites in New England.

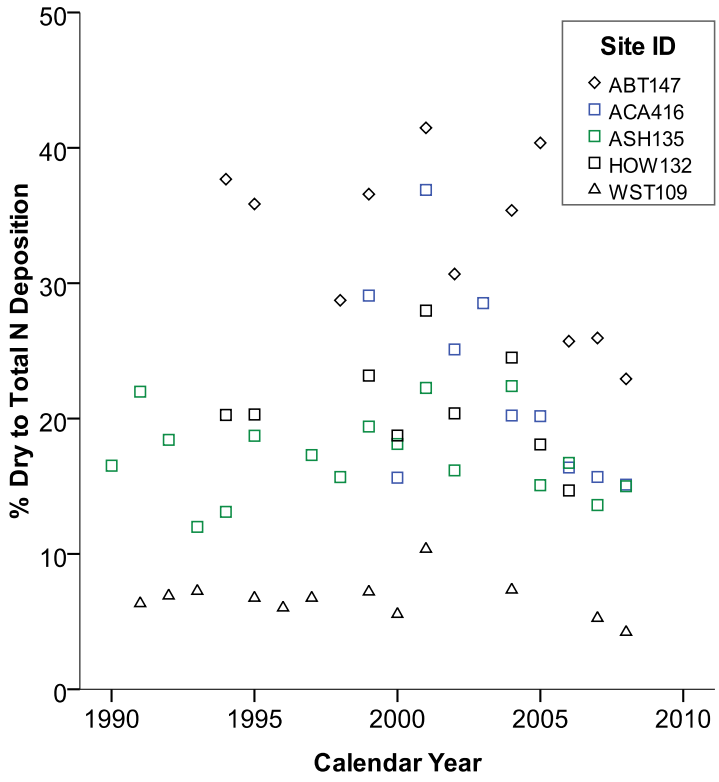

Fig. 2.8. Annual (available for calendar year only) percent of dry inorganic N deposition to total inorganic N deposition at the 5 CASTNET sites in New England.

Table 2.5. Annual (available for calendar year only) Wet, Dry, Total and Percentage of Dry to Total (% Dry) Inorganic Nitrogen Deposition at the 5 CASTNET sites in New England.

Site ID	**ABT147**				**ACA416**				**ASH135**				**HOW132**				**WST109**			
Name	Abington				Acadia NP				Ashland				Howland				Woodstock			
County	Windham				Hancock				Aroostook				Penobscot				Grafton			
State	CT				ME				ME				ME				NH			
Elev. (m)	209				158				235				69				258			
Latitutde	41.8402				44.3769				46.6041				45.2158				43.945			
Longitude	-72.01				-68.2608				-68.4135				-68.7082				-71.7008			
YEAR	**Wet**	**Dry**	**Total**	**% Dry**	**Wet**	**Dry**	**Total**	**% Dry**	**Wet**	**Dry**	**Total**	**% Dry**	**Wet**	**Dry**	**Total**	**% Dry**	**Wet**	**Dry**	**Total**	**% Dry**
1989									2.76								6.76			
1990									3.75	0.74	4.48	17%					5.80			
1991									2.59	0.73	3.32	22%					5.15	0.35	5.51	6%
1992									2.70	0.61	3.31	18%					4.98	0.37	5.35	7%
1993									3.74	0.51	4.25	12%	3.62				4.47	0.35	4.82	7%
1994	5.06	3.06	8.12	38%					3.58	0.54	4.12	13%	3.58	0.91	4.49	20%	5.21			
1995	4.66	2.61	7.28	36%					2.56	0.59	3.15	19%	3.18	0.81	3.99	20%	5.67	0.41	6.07	7%
1996	6.25								2.81				3.40				5.92	0.38	6.31	6%
1997	4.88								2.58	0.54	3.12	17%	3.47				5.25	0.38	5.63	7%
1998	5.63	2.27	7.90	29%					2.42	0.45	2.87	16%	3.14				5.20			
1999	4.05	2.33	6.37	37%	3.17	1.30	4.47	29%	2.45	0.59	3.04	19%	2.95	0.89	3.84	23%	4.25	0.33	4.58	7%
2000	4.79				3.40	0.63	4.03	16%	2.35	0.52	2.87	18%	2.87	0.66	3.52	19%	5.10	0.30	5.40	6%
2001	3.64	2.58	6.22	41%	1.95	1.14	3.09	37%	1.92	0.55	2.47	22%	2.06	0.80	2.86	28%	3.63	0.42	4.05	10%
2002	4.70	2.08	6.78	31%	3.46	1.16	4.62	25%	3.37	0.65	4.02	16%	3.32	0.85	4.17	20%	4.58			
2003	4.00				2.53	1.01	3.54	29%	2.85				2.96				4.13			
2004	4.00	2.19	6.19	35%	2.79	0.71	3.51	20%	1.94	0.56	2.50	22%	2.28	0.74	3.02	25%	4.16	0.33	4.49	7%
2005	3.25	2.20	5.45	40%	2.65	0.67	3.32	20%	2.77	0.49	3.25	15%	2.30	0.51	2.82	18%	4.21			
2006	4.16	1.44	5.60	26%	3.42	0.67	4.09	16%	2.28	0.46	2.75	17%	3.01	0.52	3.54	15%	3.45			
2007	4.28	1.50	5.78	26%	3.02	0.56	3.57	16%	2.92	0.46	3.38	14%	2.86				4.68	0.26	4.94	5%
2008	4.47	1.33	5.80	23%	3.20	0.57	3.77	15%	2.21	0.39	2.60	15%	2.73				4.08	0.18	4.26	4%
min	3.25	1.33	5.45	23%	1.95	0.56	3.09	15%	1.92	0.39	2.47	12%	2.06	0.51	2.82	15%	3.45	0.18	4.05	4%
max	6.25	3.06	8.12	41%	3.46	1.30	4.62	37%	3.75	0.74	4.48	22%	3.62	0.91	4.49	28%	6.76	0.42	6.31	10%

2.3 MODELING ESTIMATES OF WET AND DRY DEPOSTION OF NITROGEN

Lovett and Lindberg (1993) observed that in the eastern US, total atmospheric deposition (wet + dry) is approximately twice the measured wet deposition. This estimate that dry deposition is approximately equal to wet deposition has historically been used to estimate total deposition because wet deposition is more widely measured than dry deposition. However, these estimates have been shown to overestimate actual dry deposition (e.g. Boyer et al. 2002). Efforts to accurately estimate dry deposition through the use of regression modeling have been the focus of much recent scientific study.

Several studies have extrapolated total atmospheric nitrogen deposition rates measured at a single location to larger spatial scales using geographic approaches and regression models (Golden and Boyer 2009; Grimm and Lynch 2000 and 2004; Ollinger et al. 1993; Table 2.6 and 2.7). A combination of precipitation volume (obtained from local and national precipitation monitoring stations) and precipitation chemistry data (obtained from the NADP/NTN) is used to estimate wet deposition for various areas in the eastern US (Table 2.6). Atmospheric concentrations of a chemical species (obtained from CASTNET) and the deposition velocity of that species (obtained from literature values or calculated from meteorological conditions at the site) are used to estimate dry deposition. Linear regression models using CASTNET data and latitude and longitude have been used to model regional patterns in dry deposition (Ollinger et al. 1993).

Table 2.6: Estimates of wet and total (wet + dry) inorganic N Deposition (kg N ha^{-1} yr^{-1}) from models in the literature.

Study	Study Area	Chemical Species	Type of Deposition	Estimate of Deposition
Ollinger et al. 1993	Northeastern U.S.	Total Inorganic Nitrogen	Wet	3.0 to 6.6 (at low elevation ~300m) ≥ 30 (elevation >1200m)
Ollinger et al. 1993	Northeastern U.S.	Total Inorganic Nitrogen	Total Atmospheric (wet + dry)	5.0 to 11.5
Lovett and Rueth, 1999	Northeastern U.S.	Total Inorganic Nitrogen	Total Atmospheric Deposition (wet + dry)	4.2 (western ME) 6.8 (Hubbard Brook, NH) 7.1-11.1 (NY) 8.9 (Bennington, VT)
Castro et al. 2003	Atlantic and Gulf Coasts	Total Inorganic Nitrogen	Total Atmospheric (wet + dry)	Average of 4.1 to 11.7 4.5-10.3 (Northeast) 5.6 (Great Bay, NH) 4.5 (Casco Bay, ME) 7.0-10.5 (Mid-Atlantic 4.9-6.6 (Southeast) 4.7 – 11.7 (Eastern Gulf Coast) 4.1-8.9 (Western Gulf Coast)
Grimm and Lynch, 2004	Eastern U.S.	Average Nitrate	Wet	4.1 (Northeast) 2.3 (Southeast)
Holland et al. 2005	Continental U.S.	Average Nitrate	Wet	Average of 1.64 (Northeast) Maximum of >5 (Northeast)
Golden and Boyer, 2009	New York	Total Inorganic Nitrogen	Wet	4.7 to 10.5

Table 2.7: Estimates of Total Dry Deposition from the Literature.

Study	Study Area	Chemical Species	Deposition Velocity (cm s^{-1})	Total Dry Deposition (kg N ha^{-1} yr^{-1})	Percent of Total Dry Deposition
Holland et al. 2005	Continental U.S.	HNO_3^+ vapor NO_2 vapor NO_3^- aerosols NH_4^+ aerosols	1.36 0.34 0.35 0.08	Average of 1.3 (HNO_3^+ vapor) 0.18 (NO_3^-) 0.34 (NH_4^+)	
Lovett and Rueth, 1999	Northeastern U.S.	HNO_3^+ vapor NO_3^- aerosols NH_4^+ aerosols	2.14 cm s^{-1} 0.12 cm s^{-1} 0.12 cm s^{-1}	~ 1to ~4	
Ollinger et al. 1993	Northeastern U.S.	HNO_3^+ vapor NO_3^- aerosols NH_4^+ aerosols	1.3 cm s^{-1} 0.13 cm s^{-1} 0.13 cm s^{-1}	2.2 to 3.6	71-78% 2-12% 17-20%

2.3.1 Models for New England and New York

Ollinger et al. (1993) developed a model to identify spatial patterns in total atmospheric deposition of inorganic N throughout the northeastern US (Table 2.6 and 2.7). For wet deposition, precipitation data from over 300 stations were combined with modeled spatial patterns of inorganic N concentrations in wet deposition to estimate large scale spatial patterns of wet deposition. Inorganic N concentrations were obtained from approximately 30 NADP/NTN sites throughout the northeast. Ollinger et al. (1993) found a distinct west to east gradient for wet deposition, with wet deposition of inorganic N (NO_3-N + NH_4-N) at low elevations ranging from 3.0 kg ha^{-1} yr^{-1} in Maine to 6.6 kg ha^{-1} yr^{-1} in western New York (Fig. 2.9). Wet inorganic N deposition for elevations above 1200 m (in the Adirondacks) reached as high as 30 kg ha^{-1} yr^{-1}.

To estimate dry deposition, Ollinger et al. (1993) used atmospheric concentrations of a chemical species combined with estimates of deposition velocities for each species to model spatial patterns of dry deposition in the northeastern U.S. Atmospheric concentrations of commonly dry-deposited species (HNO_3^+ vapor and NO_3^- and NH_4^+ aerosols) were predominantly obtained from the CASTNET (formerly the NDDN). Ollinger et al. (1993) found that total dry nitrogen deposition showed a decreasing trend from south to north. Other studies (Holland et al. 2005; Boyer et al. 2002) have employed the methods of Ollinger et al. (1993), but have used updated deposition velocities from Lovett and Rueth (1999; Table 2.7).

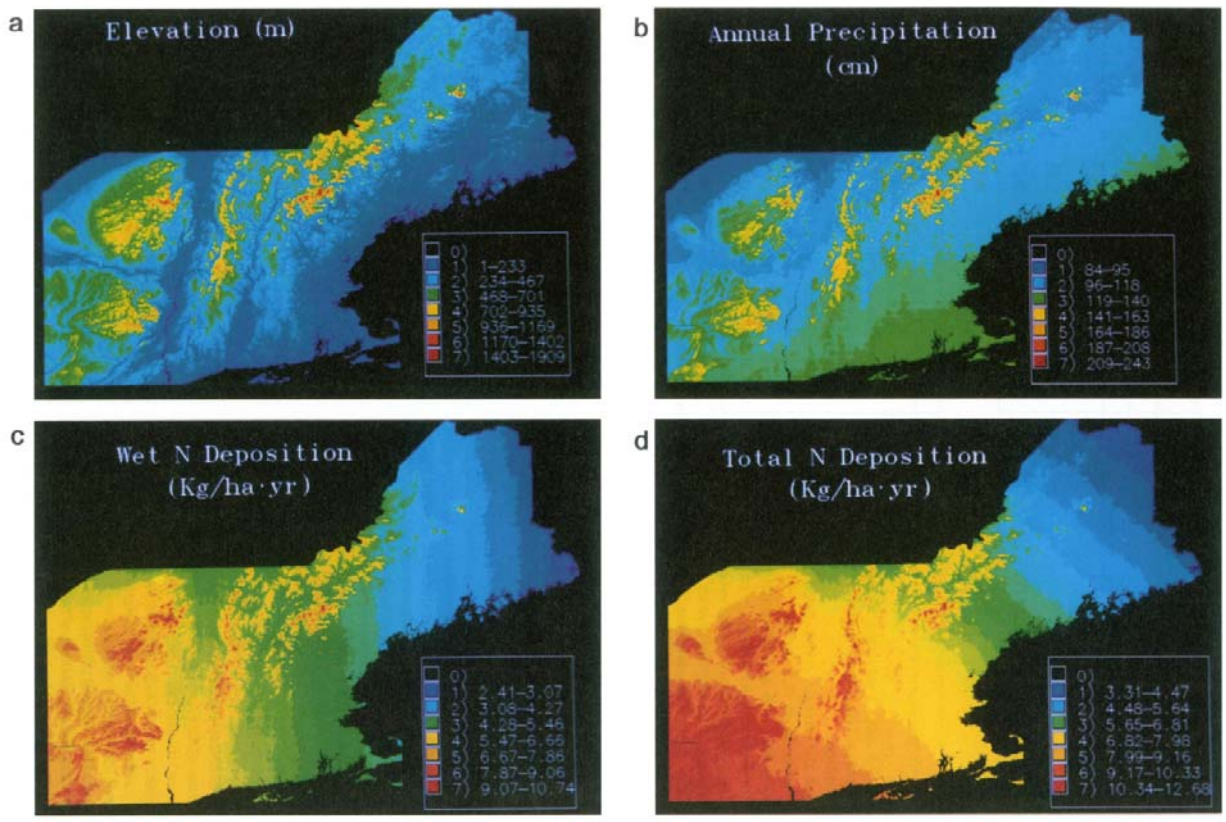

Fig. 2.9. Regional gradient of wet and total inorganic N deposition for New England and New York (from Ollinger et al. 1993).

From the work described above (Ollinger et al. 1993), and other sources (Jenkins et al. 1999; Ollinger et al. 1997; Ollinger et al. 1995; Aber et al. 1995), Ollinger et al. (2001) were able to create ClimCalc (http://www.pnet.sr.unh.edu/climcalc/), an online model to estimate climate and atmospheric deposition for New England and New York. This model inputs basic geographic and topographic variables (latitude, longitude, elevation, slope, and aspect) for a specific location in the region to estimate precipitation, solar radiation, and minimum and maximum daily temperatures as well as wet, dry, and total deposition for sulfur and inorganic nitrogen compounds. A regional wet deposition gradient was developed using data from NADP/NTN, with a trend of decreasing inorganic N deposition from west to east, primarily due to industrial areas in the western part of this region (Ollinger et al. 2001). From this gradient and estimated precipitation (based on data from local weather stations), wet deposition for a specific location can be estimated. A regional dry deposition gradient was also developed, using particle and gas concentration data from CASTNET and deposition velocities, with a trend of decreasing deposition from south to north, primarily due to urban areas in the southern part of this region. Overall, a regional gradient for total (wet + dry) deposition was found to decrease from the southwest (western New York and Pennsylvania) to the northeast (Maine) (Fig. 2.10).

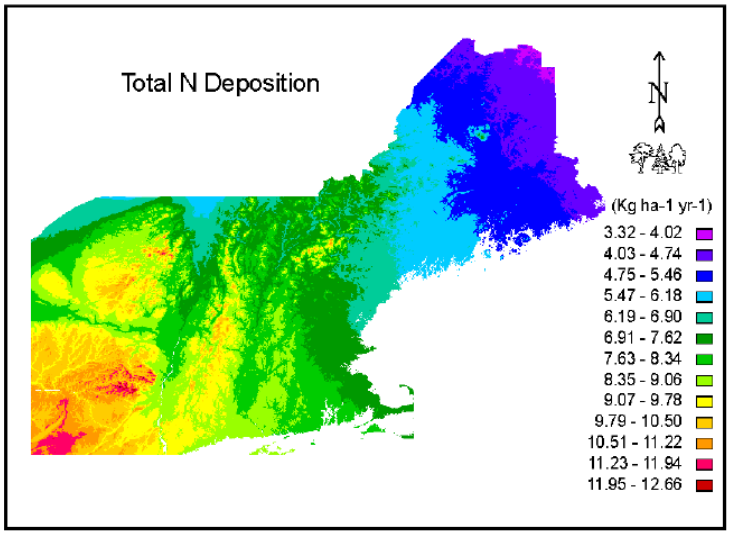

Fig. 2.10: Regional gradient of total (wet + dry) inorganic N deposition for New England and New York (from Ollinger et al. 2001).

Golden and Boyer (2009) used a model similar to Ollinger et al. 1993 to determine wet deposition of total inorganic nitrogen throughout New York (4.7 to 10.5 $kgha^{-1}yr^{-1}$). They found a distinct seasonal pattern of deposition, with total inorganic nitrogen deposition highest in the spring (~2.4 kg ha^{-1}), and lowest in the winter (~1.4 kg ha^{-1}). They also noted that nitrogen deposition is spatially distributed according to elevation and location (i.e. longitude; closeness to nitrogen sources and waterbodies). Longitude was the greatest predictor of deposition, with a clear trend of increased ion concentration in precipitation from east to west, similar to spatial patterns found by Ollinger et al. 1993. The actual influence of elevation is unclear, as some high elevation sites had low total inorganic nitrogen deposition values (5.7 kg ha^{-1} yr^{-1} in the Adirondacks), while others had higher values (8.6 $kgha^{-1}yr^{-1}$ in the Catskills).

2.3.2 Other Models

Castro et al. (2003) quantified the input from total atmospheric (wet + dry) inorganic nitrogen deposition to 34 estuaries on the Atlantic and Gulf Coasts of the U.S. including Great Bay, NH and Casco Bay, ME. In this study, total atmospheric inorganic N deposition to both land and surface water within each watershed were estimated using data from NADP/NTN and CASTNET. Deposition to surface water was estimated by multiplying the total surface water area in the watershed by the wet and dry deposition rates of inorganic nitrogen. Overall, total N deposition ranged from 4.1 to 11.7 kg ha^{-1} yr^{-1} for the entire region (Table 2.6). The southeast US had lower estimates of deposition (4.9 to 6.6. kg ha^{-1} yr^{-1}) than the Northeast and Mid-Atlantic U.S. (7 to 10 kg ha^{-1} yr^{-1}) though Great Bay, NH and Casco Bay, ME had estimates slightly lower than the rest of the Northeast (5.6 kg ha^{-1} yr^{-1} and 4.5 kg ha^{-1} yr^{-1} respectively; Table 2.6). Castro et al. (2003) further estimated that the total contribution made by atmospheric

deposition to the nitrogen inputs to the estuary was 22.3% of the total nitrogen input in Great Bay, NH, and 43.2% in Casco Bay, ME.

Grimm and Lynch (2004) developed a model to estimate wet deposition in the Eastern U.S. using daily precipitation data from NOAA stations, precipitation chemistry data from NADP/NTN, and topographic variables (such as slope, elevation, and aspect). The inclusion of topographic variables into their model allowed for a higher resolution for spatial patterns of wet deposition to be developed, as precipitation volume will vary with these variables. In an earlier study by Grimm and Lynch (1991), it was found that interpolating wet deposition estimates from NADP/NTN sites to unmonitored areas resulted in an average percent error for annual wet nitrate deposition of 13%.

2.3.3 Direct Deposition to Surface Waters

Jordan and Talbot (2000) assessed the direct wet and dry atmospheric deposition to the surface waters of the Gulf of Maine (measured at New Castle, NH) between 1994 and 1997. A filter pack system with a rain shield was used to obtain daily bulk aerosol and nitric acid samples, and precipitation samples were collected using a wet-only collector. Dry deposition was calculated using separate models for nitric acid deposition and deposition from aerosol particles. Wet deposition was calculated by multiplying the concentrations of nitrogen by the rainfall amount. The amount of deposition from fog was also determined.

Daily dry deposition was found to be highly variable, ranging from ~ 14 to 2016 μg N m^{-2} d^{-2} (median 224 μg $Nm^{-2}d^{-2}$). Wet deposition ranged from ~ 42 to 59,696 μg $Nm^{-2}d^{-2}$ (median 2996 μg $Nm^{-2}d^{-2}$). Overall, wet deposition (5.43 to 7.17 kg N/ha/yr) comprised ~ 80 to 90% of the total atmospheric deposition (6.35 to 8.41 kg N/ha/yr) for most of the study period (1994-1997). Deposition from fog water was also calculated by collecting samples with an active sampler following a design by Global Geochemistry Corporation that uses a fan to pull the fog through mesh strands. As the fog condenses on the strands, it collects in a sampler at the bottom of the instrument. Fog deposition was found to contribute as much nitrogen to the Gulf as heavy precipitation (up to 7005 μg $Nm^{-2}d^{-2}$) on some events (Jordan and Talbot, 2000).

Jordan and Talbot (2000) compared their directly measured deposition rates to those found at the two closest NADP/NTN sites (ME98 - Mt. Desert Island, ME and MA01 – Cape Cod, MA). They found that values measured at New Castle, NH exceeded those reported from the NADP/NTN sites by 69% and 42% respectively though precipitation amount was similar for all three sites. This may have occurred because of differences in sampling protocol; Jordan and Talbot (2000) collected samples daily while the NADP/NTN sites are collected as weekly bulk samples. New Castle, NH may also receive urban inputs which are not as significant at the NADP sites.

2.4 ESTIMATING TOTAL NITROGEN DEPOSITION IN THE GREAT BAY AND PISCATAQUA RIVER WATERSHED

To estimate total (wet + dry) nitrogen deposition throughout the Great Bay watershed, several approaches could be taken. All approaches would involve separate estimates of wet and dry deposition which could be summed to estimate total deposition. The simplest approach to estimate wet N deposition would be to assume rates measured at TF are constant throughout the Piscataqua River watershed. The NH WRRC has directly measured wet deposition at TF since Nov. 2003 and intends to continue to monitor wet deposition at this site as long as funding is available. The NH WRRC also collects data on DON in wet deposition which is rarely monitored at other sites and is not reported at NADP and CASTNET stations. To date TF deposition data indicate that DON is a small component of wet deposition (4 to 8%, Table 2.4), however, given that deposition is often the largest input to forested watersheds (Castro et al. 2003; Lovett and Rueth 1999; Ollinger et al. 1993), it may be an important input to quantify in watersheds where other N inputs from human activities and agriculture are minor. If annual wet deposition estimates are needed prior to 2004, they can be estimated by the average of wet deposition at the MA08 and MA13 NADP stations. These two NADP stations reported values that were closest to TF deposition from 2004-2008 (Fig. 2.11). The average inorganic N deposition at these two stations could be divided by 0.94 (the average ratio of DIN:TDN wet deposition at TF) to estimate annual wet TDN deposition prior to 2004. Since CASNET wet deposition is interpolated from NADP deposition, we did not compare TF wet deposition to modeled wet deposition at CASTNET stations which are located further away.

The second approach to estimate wet deposition would be to assume that the average annual concentration of N in wet deposition is consistent throughout the Piscataqua River watershed, but that the annual precipitation amount varies based on watershed position. In 2004, NH WRRC staff carried out an intensive precipitation sampling campaign to assess the spatial variability in wet deposition throughout the Lamprey River basin (479 km^2). Weekly precipitation samples were collected using funnel and bottle collectors (during non-winter months) and open buckets (during winter months) at 5 stations in addition to TF. Sites ranged in elevation from 29 to 236 MASL. Results showed that TDN concentrations did not vary throughout the basin (Fig. 2.12) with the exception of the station in Lee, NH. This station was located at a dairy farm and we suspect volatilized animal excrement and fertilizers were re-deposited locally and the elevated TDN concentrations are not representative of broader conditions. Concentrations of TDN were slightly lower at TF than at the 5 additional sites. This was expected, however, as the TF farm sampler measured wet deposition, while the others measured bulk deposition. We conclude that assuming concentrations of N in wet deposition are consistent throughout the Lamprey basin and the larger Piscataqua River watershed is reasonable.

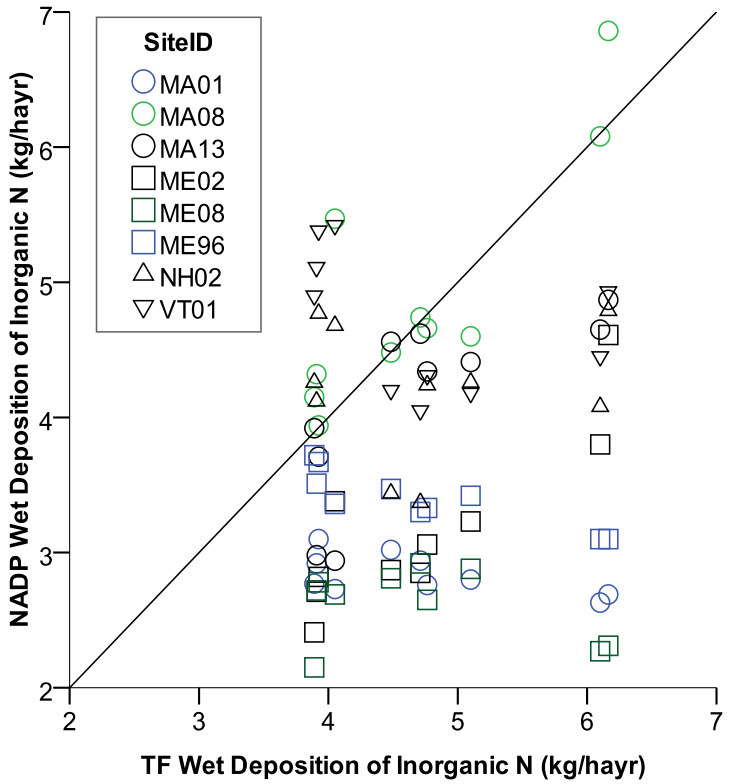

Fig. 2.11. Wet inorganic N deposition at the 8 NADP stations closest to the Piscataqua River watershed compared to wet inorganic N deposition measured at Thompson Farm (TF) in Durham, NH (TF) over from 2004 – 2008 (both calendar and water years are included). The 1:1 line is shown.

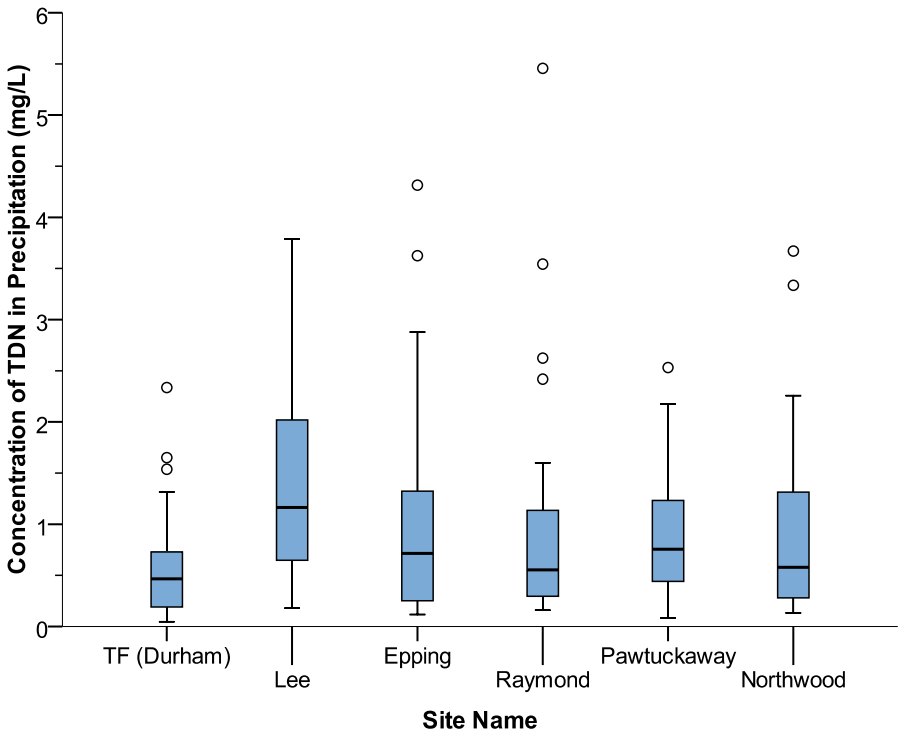

Fig. 2.12. Concentration of total dissolved nitrogen (TDN) in weekly precipitation samples collected throughout the Lamprey basin during 2004. Horizontal lines represent median values, boxes represent 25th to 75th percentiles and whiskers represent 10th to 90th percentiles.

Although concentrations of N in wet deposition are consistent throughout the Lamprey watershed within a year, the amount of annual precipitation varies throughout the Lamprey basin (by approximately +/- 30 % of the average in 2004) and the greater Piscataqua River watershed. We assembled data from the NCDC (http://www.ncdc.noaa.gov/oa/ncdc.html) and CRN (http://www.ncdc.noaa.gov/crn/) stations located within the Piscataqua River watershed or within 25 km from its border (Fig. 2.13; Table 2.8) and included data from 1980-2009 (Appendix A; see accompanying excel file for historical data from 1894-1979, Appendix B). Annual precipitation varied spatially throughout the Piscataqua watershed (by approximately +/- 10 to 30% of the average in any given year) and has ranged from 625 to ~1900 mm/yr over the last 30 years (Fig. 2.14). Even though precipitation is generally related to elevation throughout New England (Ollinger et al. 1993), elevation was not a strong predictor of annual (Fig. 2.15) or median (Fig. (2.16) precipitation among sites within and near the Piscataqua River watershed from 2000-2009. To estimate precipitation amount at a location, one could assume it is similar to the closest NCDC or CRN station with similar elevation. Another approach would be to interpolate (using a geographical information system) precipitation values for the period of interest over the Piscataqua River watershed based on area and elevation (e.g. Boyer et al. 2002). Once the precipitation amount for the period of interest is estimated, this amount would then be multiplied by the volume-weighted concentration at TF over the same time period to estimate deposition across the watershed.

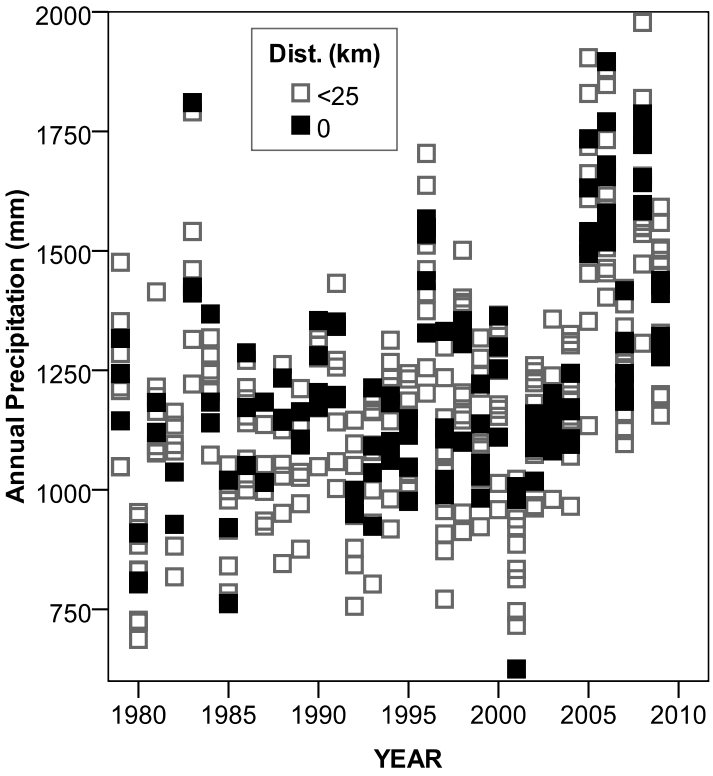

Fig. 2.14. Annual (calendar year) precipitation at NCDC and CRN stations in the Piscataqua River watershed (0 km) and within 25 km of its border.

Fig. 2.13. Map of the CRN (hourly precipitation data since November 2002) and NCDC (daily and monthly precipitation data since 1894 at some locations) stations located within (0 km) or closest to the Piscataqua River watershed (< 25 km from watershed border).

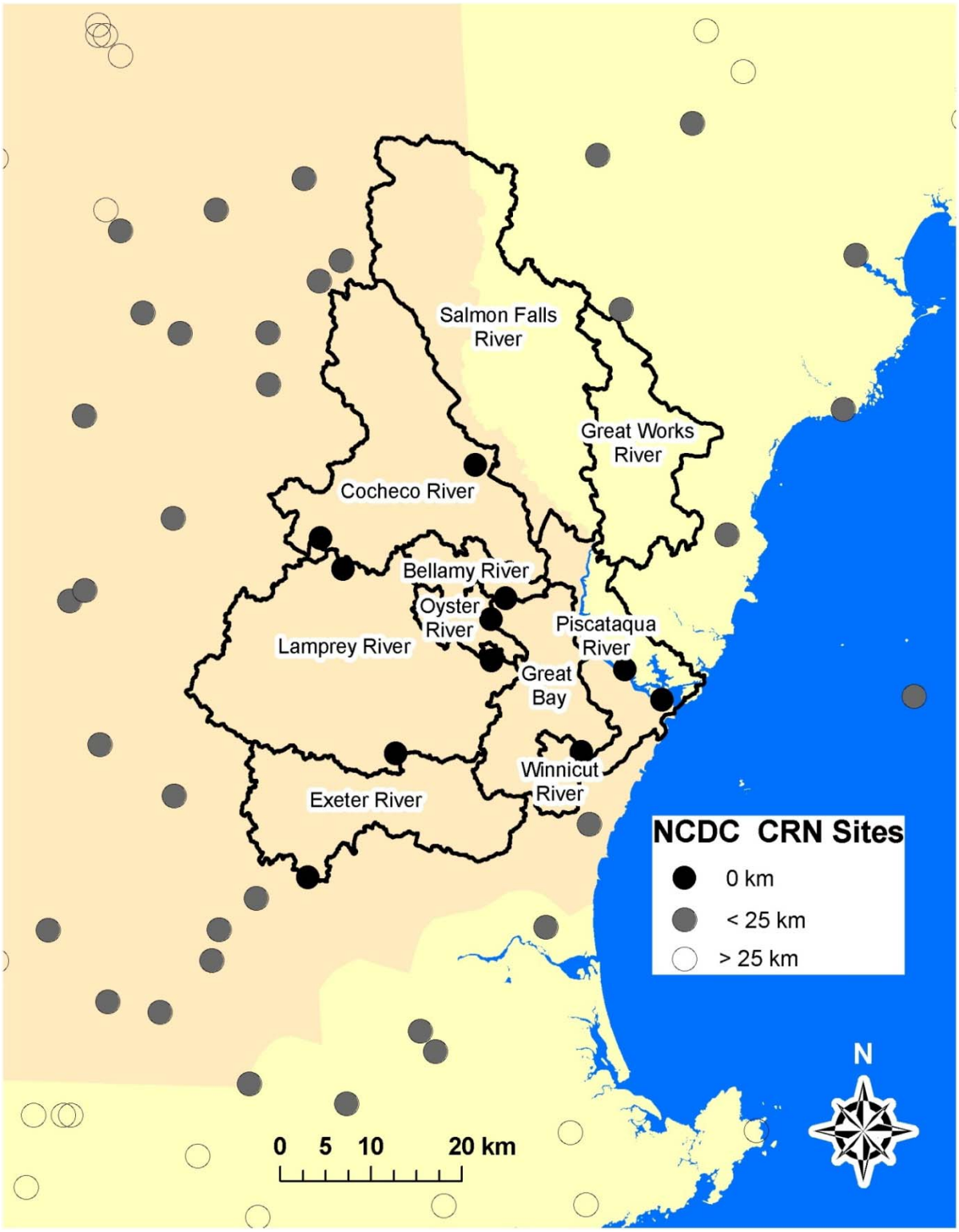

Table 2.8. Metadata for NCDC and CRN stations within (0 km) or near (< 25 km) the Piscataqua River watershed.

Source	Station Name	County	State	Latitude	Longitude	Elev. (m)	Dist. (km)
CRN	Durham 2 SSW	STRAFFORD	NH	43.11	-70.95	19.2	0
CRN	Durham 2N	STRAFFORD	NH	43.17	-70.93	36.3	0
NCDC	ELIOT	YORK	ME	43.10	-70.77	6.1	0
NCDC	DURHAM	STRAFFORD	NH	43.15	-70.95	24.4	0
NCDC	EPPING	ROCKINGHAM	NH	43.02	-71.08	48.8	0
NCDC	GREENLAND	ROCKINGHAM	NH	43.02	-70.83	25.9	0
NCDC	NORTHWOOD	ROCKINGHAM	NH	43.20	-71.15	164.9	0
NCDC	NORTHWOOD CTR	ROCKINGHAM	NH	43.23	-71.18	241.1	0
NCDC	PORTSMOUTH	ROCKINGHAM	NH	43.07	-70.72	18.0	0
NCDC	ROCHESTER	STRAFFORD	NH	43.30	-70.97	70.1	0
NCDC	WEST HAMPSTEAD	ROCKINGHAM	NH	42.90	-71.20	91.4	0
NCDC	DRACUT	MIDDLESEX	MA	42.70	-71.28	82.0	≤ 25
NCDC	GROVELAND	ESSEX	MA	42.73	-71.03	10.1	≤ 25
NCDC	HAVERHILL	ESSEX	MA	42.75	-71.05	6.1	≤ 25
NCDC	LAWRENCE	ESSEX	MA	42.68	-71.15	15.2	≤ 25
NCDC	NEWBURYPORT 4 NNW	ESSEX	MA	42.85	-70.88	25.9	≤ 25
NCDC	CAPE NEDDICK	YORK	ME	43.23	-70.63	39.6	≤ 25
NCDC	HOLLIS	YORK	ME	43.63	-70.67	96.0	≤ 25
NCDC	KENNEBUNKPORT	YORK	ME	43.35	-70.47	6.1	≤ 25
NCDC	SACO	YORK	ME	43.50	-70.45	24.1	≤ 25
NCDC	SANFORD 2 NNW	YORK	ME	43.45	-70.77	85.3	≤ 25
NCDC	WATERBORO	YORK	ME	43.60	-70.80	147.8	≤ 25
NCDC	ALTON	BELKNAP	NH	43.43	-71.25	219.5	≤ 25
NCDC	BARNSTEAD 5N	BELKNAP	NH	43.38	-71.25	211.8	≤ 25
NCDC	BOW GARVINS FALLS	MERRIMACK	NH	43.17	-71.52	66.1	≤ 25
NCDC	CONCORD MUNI AP	MERRIMACK	NH	43.18	-71.50	105.5	≤ 25
NCDC	DERRY	ROCKINGHAM	NH	42.85	-71.32	91.4	≤ 25
NCDC	EAST DERRY	ROCKINGHAM	NH	42.88	-71.27	164.6	≤ 25
NCDC	GILMANTON	BELKNAP	NH	43.45	-71.42	313.9	≤ 25
NCDC	GILMANTON 2 E	BELKNAP	NH	43.43	-71.37	244.1	≤ 25
NCDC	HUDSON 1 SSE	HILLSBOROUGH	NH	42.77	-71.40	56.4	≤ 25
NCDC	LAKEPORT 2	BELKNAP	NH	43.53	-71.45	152.4	≤ 25
NCDC	MANCHESTER	HILLSBOROUGH	NH	43.03	-71.48	64.0	≤ 25
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	42.98	-71.38	77.1	≤ 25
NCDC	MERRIMACK	HILLSBOROUGH	NH	42.85	-71.55	73.2	≤ 25
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.78	-71.47	42.7	≤ 25
NCDC	NEW CASTLE		NH	43.07	-70.38	3.0	≤ 25
NCDC	NEW DURHAM 3 NNW	STRAFFORD	NH	43.48	-71.18	195.1	≤ 25
NCDC	NEW DURHAM 4 N	STRAFFORD	NH	43.50	-71.15	201.2	≤ 25
NCDC	NORTH CHICHESTER	MERRIMACK	NH	43.25	-71.38	110.0	≤ 25
NCDC	NORTH HAMPTON	ROCKINGHAM	NH	42.95	-70.82	18.3	≤ 25
NCDC	SALISBURY	MERRIMACK	NH	43.35	-71.50	167.6	≤ 25
NCDC	WEST ALTON	BELKNAP	NH	43.55	-71.32	156.1	≤ 25
NCDC	WINDHAM 3 NW	ROCKINGHAM	NH	42.82	-71.33	67.1	≤ 25
NCDC	WOLFEBORO FALLS	CARROLL	NH	43.58	-71.20	161.8	≤ 25

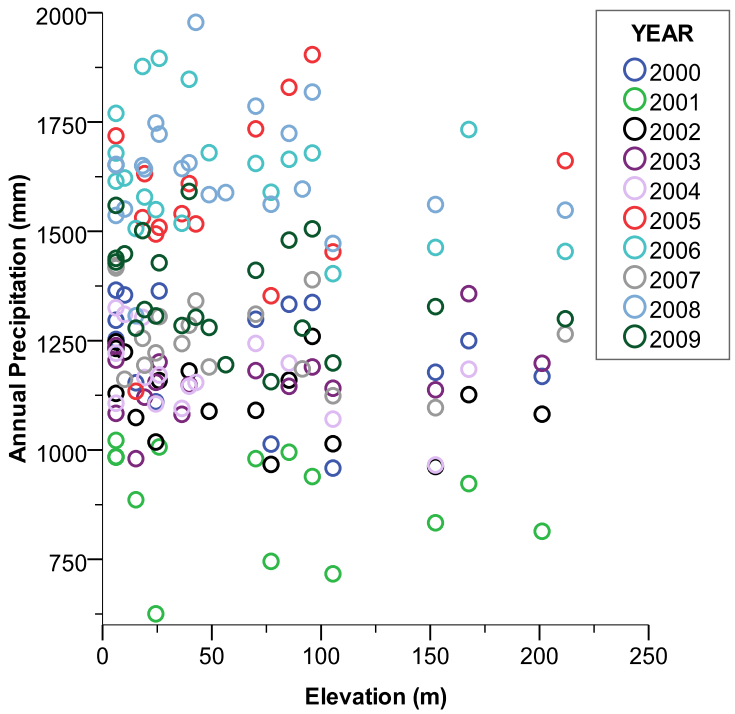

Fig. 2.15. Annual (calendar year) precipitation from 2000-2009 at NCDC and CRN stations in the Piscataqua Watershed or within 25 km compared to elevation.

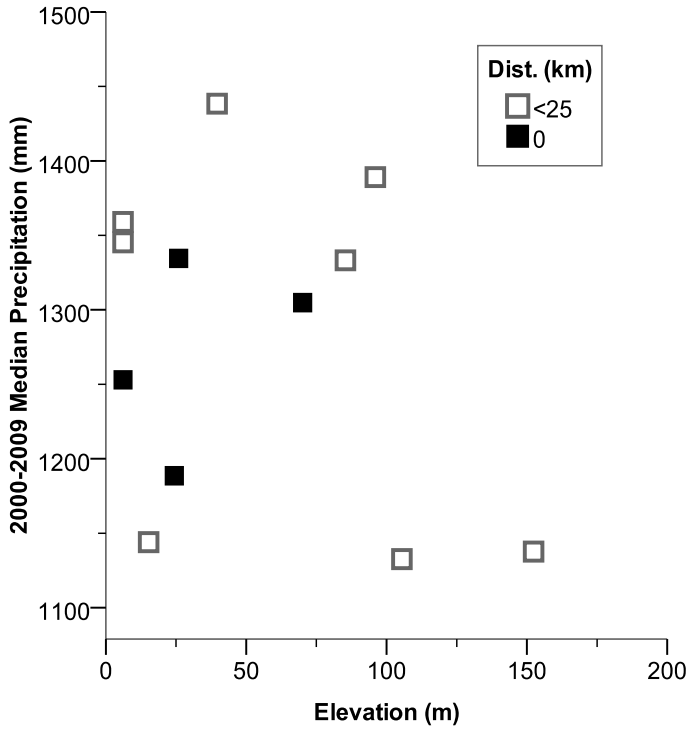

Fig. 2.16. Median annual (calendar year) precipitation from 2000-2009 at NCDC and CRN stations in the Piscataqua Watershed (0 km) and within 25 km compared to elevation.

Another approach to estimate wet deposition across the Piscatagua River watershed would be to use models. Ollinger et al. (1993) found that elevation was an important factor in predicting N deposition, but elevation alone cannot accurately predict annual (Fig. 2.17) or median (Fig. 2.18) wet N deposition among the monitoring sites closest to and within the Piscataqua River watershed. Below 100 MASL, wet deposition is variable and highest at TF (23 MASL), a low elevation site. Above 100 MASL, deposition generally increases with elevation up to approximately 300 MASL, but elevation alone is not a strong enough predictor to accurately estimate deposition across the full range of elevation. The ClimCalc model (http://www.pnet.sr.unh.edu/climcalc/; Ollinger et. al 2001) could be used to predict wet, dry and total deposition based on latitude, longitude, elevation, slope and aspect. Among the 8 NADP sites closest to the Piscataqua River watershed and the CASTNET sites in New England, ClimCalc both over- and under-predicted wet inorganic N deposition (Fig. 2.19), over-predicted total inorganic N deposition in most years (Fig. 2.20), and over predicted dry deposition in all years (1999-2008; Fig. 2.21). However, ClimCalc was a better predictor of median wet N deposition (0 to 49% over-predicted; Fig. 2.22) and median total N deposition (over-predicted 23 to 44%; Fig. 2.23) over the last 10 years, but over-predicted median dry deposition by more than 50% at 3 of the 5 sites (31 to 503% over-predicted; Fig. 2.24). This over-prediction of wet, total and dry deposition at NADP and CASTNET sites may reflect air quality improvements that have occurred since the 1980s and early 1990s which is the time period that the calibration data (NADP and CASTNET) for ClimCalc were collected. At TF, ClimCalc under-estimated annual wet deposition in most years (Fig. 2.19) and under-estimated median wet deposition by only 11% (Fig. 2.22). The under-estimation of deposition at TF could be reflective of local urban sources that the ClimCalc model was not designed to capture. Overall, the ClimCalc model does a fair job at predicting general patterns of wet deposition, but is not able to predict variability in deposition over time.

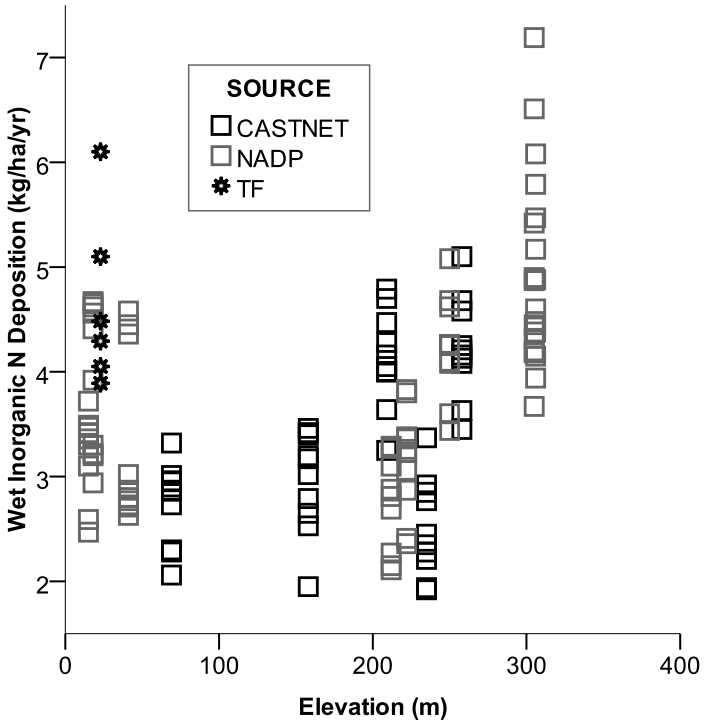

Fig. 2.17. Wet inorganic N deposition at the 8 NADP stations closest to the Piscataqua Watershed, the New England CASTNET stations and at Thompson Farm in Durham, NH (TF) over the last 10 years (1999-2008 for NADP and CASTNET and 2004-2009 for TF) compared to elevation (MASL).

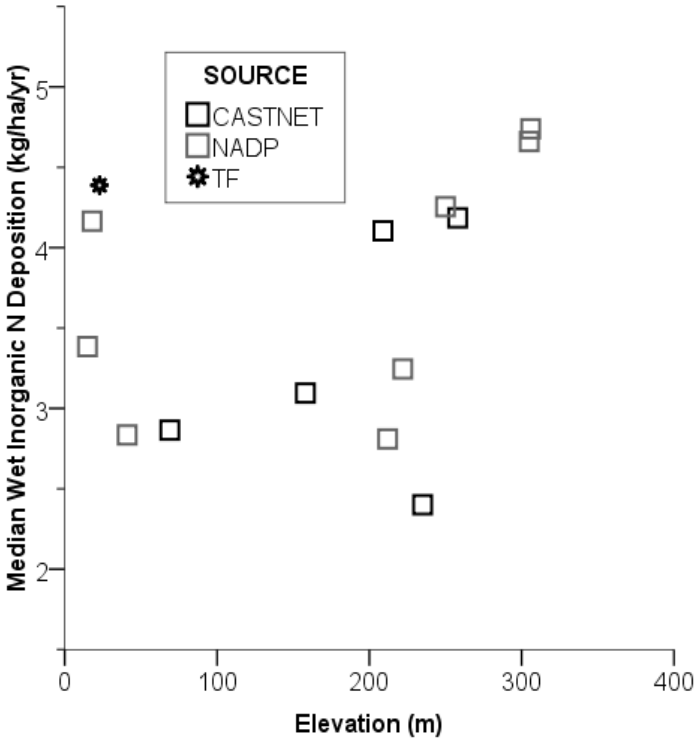

Fig. 2.18 Median wet inorganic N deposition at the 8 NADP stations closest to the Piscataqua Watershed, the New England CASTNET stations and at Thompson Farm in Durham, NH (TF) over the last 10 years (1999-2008 for NADP and CASTNET and 2004-2009 for TF) compared to elevation.

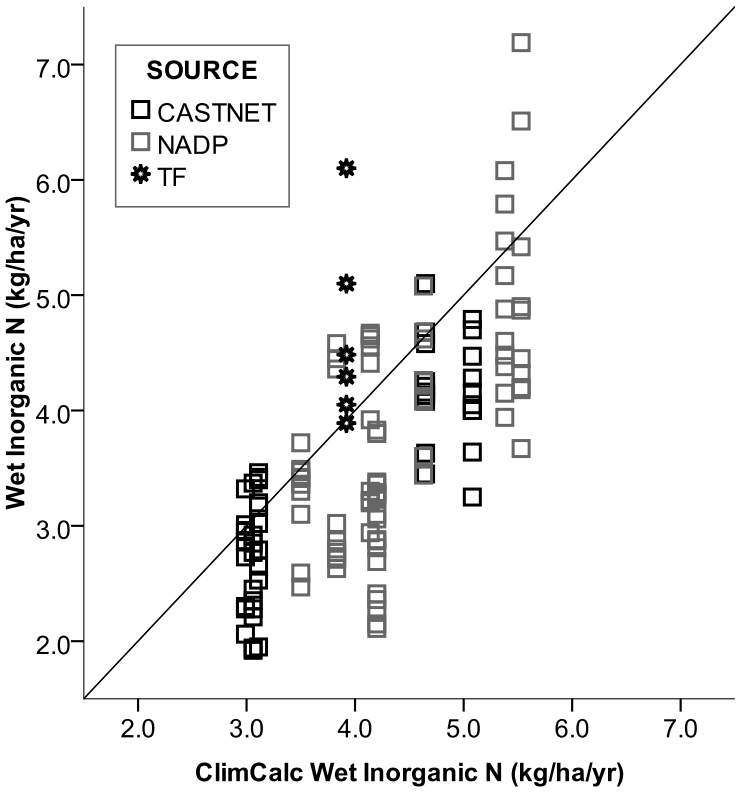

Fig. 2.19. Measured wet deposition of inorganic N at the 8 NADP stations closest to the Piscataqua Watershed, at Thompson Farm in Durham, NH and modeled wet deposition at the New England CASTNET stations over the last 10 years (calendar years 1999-2008 for NADP and CASTNET and 2004-2009 for TF) compared to ClimCalc predicted wet deposition. The 1:1 line is shown.

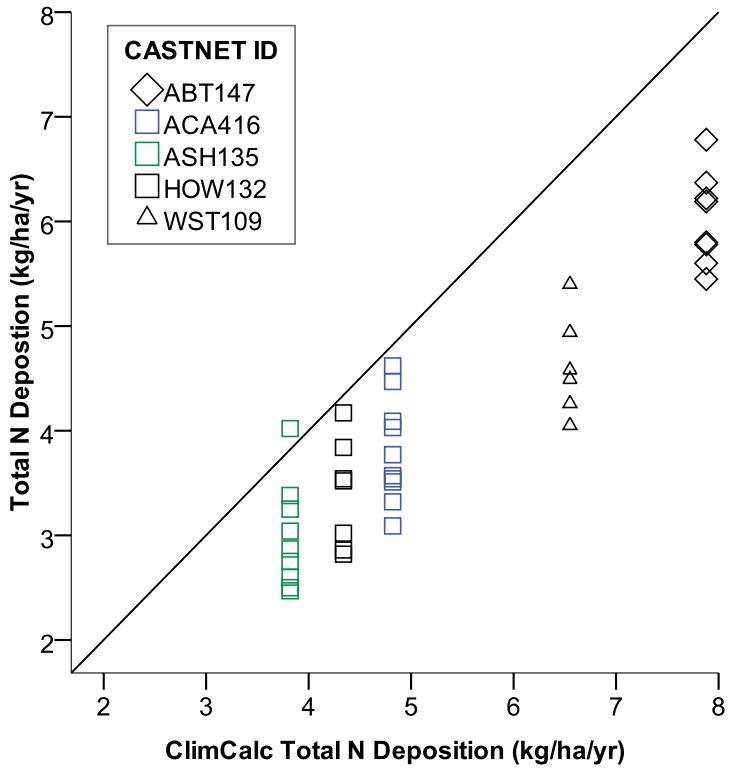

Fig. 2.20 Total N deposition (wet + dry) at New England CASTNET sites from 1999-2008 (calendar years) compared to ClimCalc predicted total N deposition. The 1:1 line is shown.

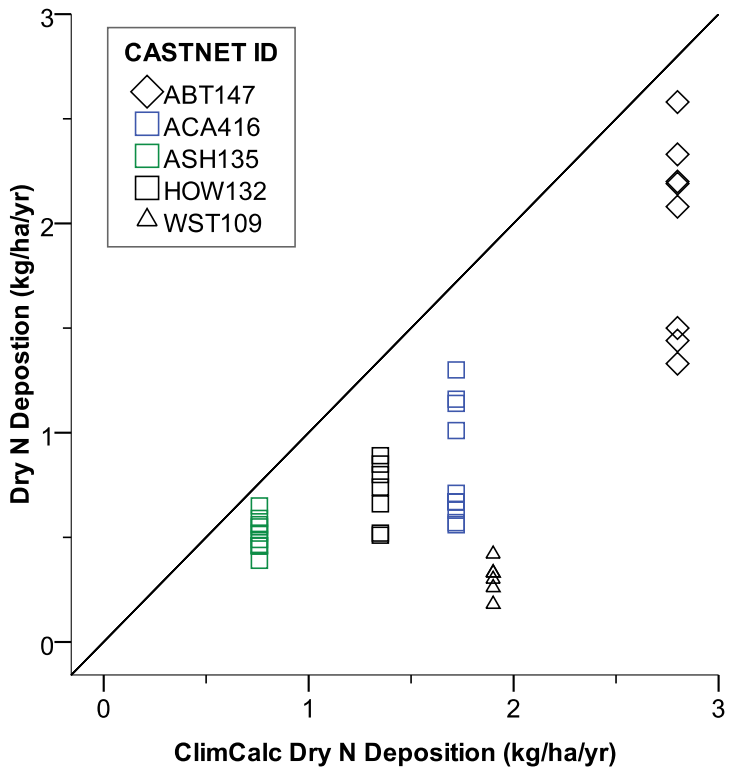

Fig. 2.21 Dry N deposition at New England CASTNET sites from 1999-2008 (calendar years) compared to ClimCalc predicted dry N deposition. The 1:1 line is shown.

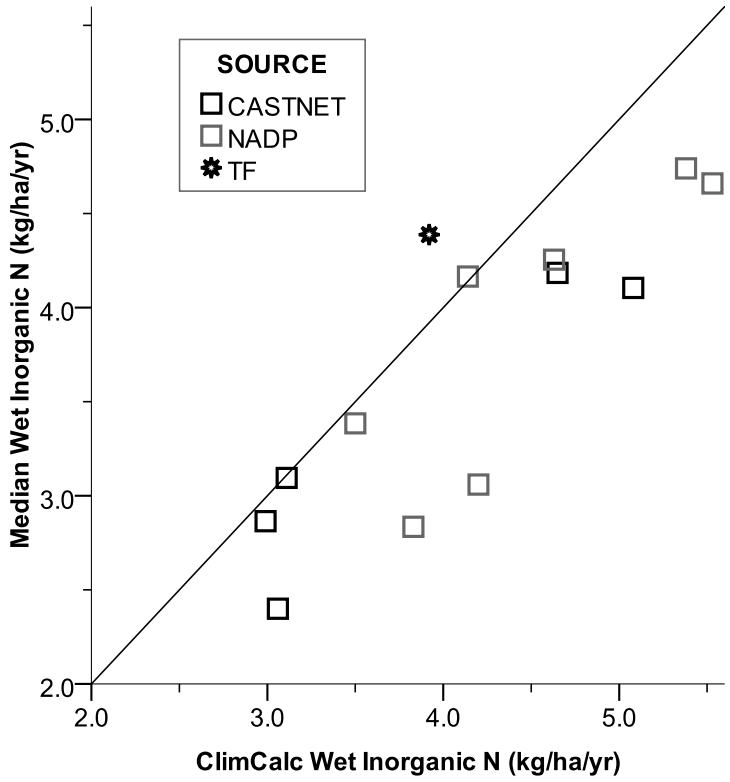

Fig. 2.22. Median measured wet deposition of inorganic N at the 8 NADP stations closest to the Piscataqua Watershed, at Thompson Farm in Durham, NH and modeled wet deposition at the New England CASTNET stations over the last 10 years (calendar years 1999-2008 for NADP and CASTNET and 2004-2009 for TF) compared to ClimCalc predicted wet deposition. The 1:1 line is shown.

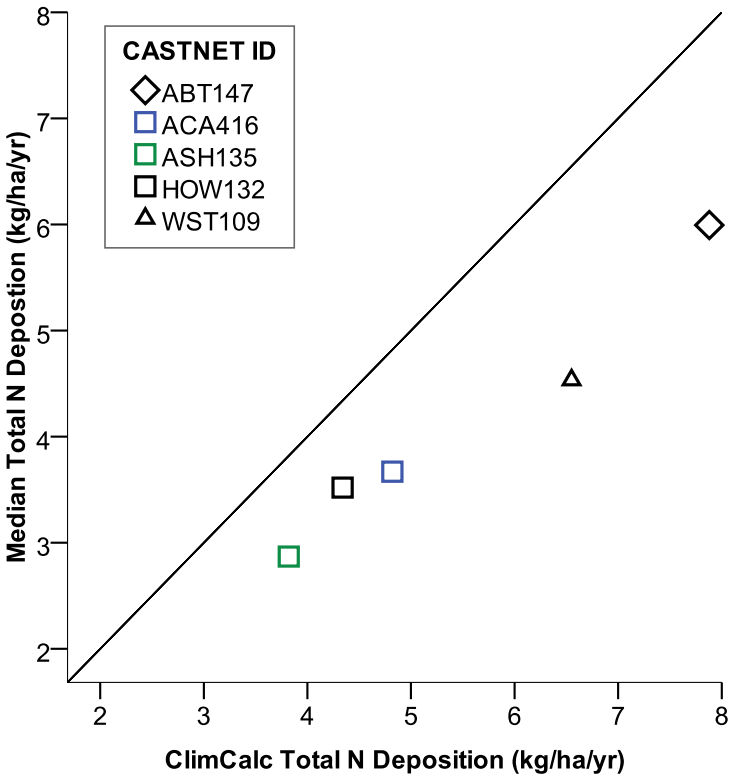

Fig. 2.23. Median total N deposition (wet + dry) at New England CASTNET sites from 1999-2008 (calendar years) compared to ClimCalc predicted total N deposition. The 1:1 line is shown.

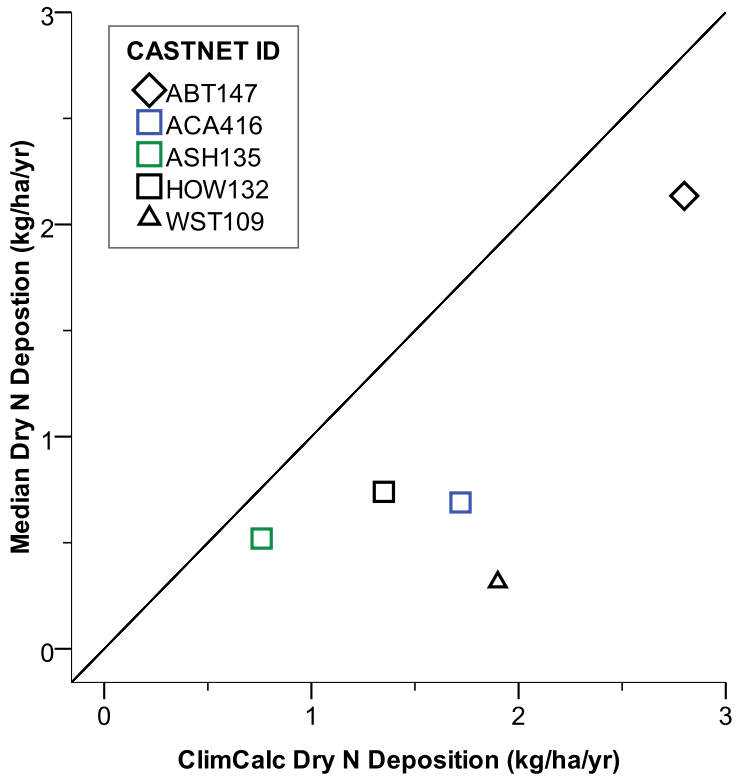

Fig. 2.24. Median dry N deposition at New England CASTNET sites from 1999-2008 (calendar years) compared to ClimCalc predicted total N deposition. The 1:1 line is shown.

There are two approaches that could be used to estimate dry deposition once wet deposition is estimated in addition to modeled estimates of dry deposition by ClimCalc. One approach would be to apply the ratio of dry to wet deposition at the closest CASTNET sites (ABT147 and WST109; Fig. 2.1) during the period of interest to the estimated wet deposition. However, these ratios vary over time (0.30 to 0.71 at ABT147 and 0.04 to 0.12 at WST109; Fig. 2.25) and median values are different between the two sites (0.50 at ABT147 and 0.07 at WST109; Fig. 2.26). Jordan and Talbot (2000) reported wet, dry and total deposition to the gulf of Maine for 1994-1997 and their data resulted in dry to wet deposition ratios ranging from 0.11-0.23 (median 0.19). This range is higher than the ratios at WST109, but lower than ratios at ABT147. The other approach to estimating dry to wet deposition ratios for the Piscataqua River watershed would be to calculate the ratio of dry to wet deposition predicted by ClimCalc and apply the ratio to estimated wet deposition (either directly from TF data or from volume-weighted TF concentration data and estimated precipitation volume from NCDC/CRN stations). ClimCalc predicts a dry to wet deposition ratio of 0.58 for TF and this is higher than the deposition ratio found by Jordan and Talbot (2000); however, dry deposition to surfaces waters is typically less than dry deposition on vegetated surfaces. Based on personal communications

with S. Ollinger, using ClimCalc to determine the ratio of dry to wet deposition and applying the ratio to wet deposition measured at TF would be the most appropriate approach for TF since wet deposition at TF is predicted fairly well by ClimCalc (Fig. 2.22). We used this approach to estimate annual dry and total deposition at TF (Table 2.9). From 2004-2009, median wet deposition was 4.62 kg N/ha/yr for calendar years and 4.78 kg N/ha/yr for water years at TF. This results in 2.68 kg N/ha/yr and 2.77 kg N/ha/yr in dry N deposition and 7.30 kg N/ha/yr and 7.55 kg N/ha/yr in total deposition, respectively.

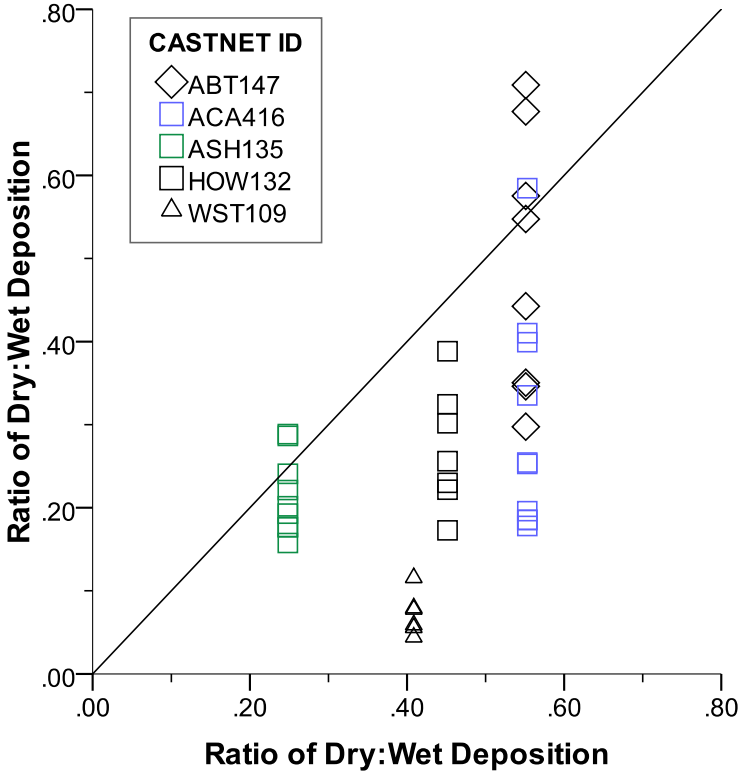

Fig. 2.25 Dry N deposition at New England CASTNET sites from 1999-2008 (calendar years) compared to ClimCalc predicted dry N deposition. The 1:1 line is shown.

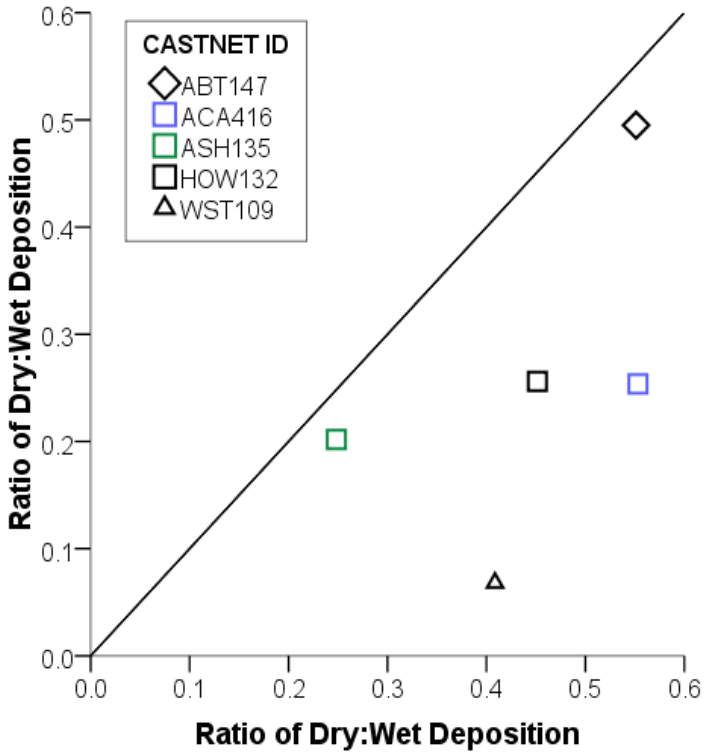

Fig. 2.26 Median dry N deposition at New England CASTNET sites from 1999-2008 (calendar years) compared to ClimCalc predicted dry N deposition. The 1:1 line is shown.

Table 2.9. Annual wet deposition of dissolved inorganic nitrogen (DIN), dissolved organic nitrogen (DON), wet deposition (DIN and DON), dry deposition and total deposition in kg N ha^{-1} yr^{-1} at Thompson Farm (TF) in Durham, NH). Both calendar years (CY) and water years (WY) are reported.

Year	Wet DIN Deposition (kg/ha/yr)		Wet DON Deposition (kg/ha/yr)		Wet Deposition (kg/ha/yr)		Dry Deposition (kg/ha/yr)		Total Deposition (kg/ha/yr)	
	CY	WY	CY	WY	CY	WY	CY	WY	CY	WY
2004	3.89	3.92	0.34	0.31	4.23	4.23	2.45	2.46	6.68	6.69
2005	5.10	4.76	0.25	0.26	5.35	5.02	3.10	2.91	8.45	7.94
2006	4.48	4.71	0.31	0.35	4.79	5.06	2.78	2.93	7.57	7.99
2007	4.05	3.91	0.24	0.17	4.29	4.08	2.49	2.37	6.78	6.44
2008	6.10	6.16	0.48	0.48	6.58	6.65	3.82	3.86	10.40	10.50
2009	4.29	4.38	0.15	0.16	4.44	4.54	2.57	2.63	7.01	7.17
Min	3.89	3.91	0.15	0.16	4.23	4.08	2.45	2.37	6.68	6.44
Max	6.10	6.16	0.48	0.48	6.58	6.65	3.82	3.86	10.40	10.50

3.0 LAMPREY AND OYSTER RIVER BASIN NITROGEN MODELS

Over the past 10 years, the NH WRRC has collected data on nitrogen concentrations in many streams in the Lamprey and Oyster basins. All sites were located upstream of tidal influence from Great Bay or Little Bay (Fig. 3.1). A total of 39 sites have been sampled at a monthly to weekly frequency and most sites are located within the Lamprey basin (N = 32). Samples have been collected for at least a year at all sites and we have 10 years of weekly data and storm event data for our most intensively sampled site, the Lamprey River at Packers Falls (our site L73 in Fig. 3.1) which is collocated with USGS site 1073500. Twenty-seven of the 39 stream sites sampled were independent, i.e. they were not downstream of any other sampling site. The number of samples taken per site ranged from 13 to 687 (mean: 87, median: 27). All samples were analyzed for TDN, NO_3-N, NH_4-N and DON in the Water Quality Analysis Laboratory (WQAL) at UNH.

We assembled our existing stream data to determine median annual fluxes of DIN (NO_3-N and NH_4-N), DON and TDN (DIN and DON) from the 39 basins during WY 2000-2009. Fluxes were calculated by multiplying the discharge-weighted mean N concentration by the median annual runoff from either the Lamprey (USGS station 1073500; 620 mm/yr) or Oyster (USGS station 01073000, 610 mm/yr) basin during WY 2000-2009. Three of the stream sites (Fig. 3.1) were continuously gauged for ~1 to 3 years by either the USGS (station 01073460 and 01073319) or by a hydrologist at the University of New Hampshire. Forward multiple regression models were developed for these temporarily gauged sites (R^2 = 0.70 to 0.97, $p<0.01$) to predict daily runoff based on the best combination of the following input variables: Lamprey or Oyster runoff on the corresponding day or Lamprey or Oyster River runoff on the following day (to account for travel time). Runoff on sampling dates used in calculating volume-weighted mean concentrations was obtained from either predicted runoff at the closest temporarily gauged site or measured runoff at the Lamprey or Oyster River USGS gauging stations, assuming that runoff was evenly distributed throughout those portions of the watershed. At the 23 smallest stations, stream discharge was directly measured 7 to 46 times to verify that the assumption of evenly distributed runoff was appropriate. For sites that are located downstream of other sites, incremental fluxes were used.

Landscape characteristics for the basins were assembled using a geographical information system (GIS) and data distributed by NH GRANIT (http://www.granit.unh.edu/). For sites that are located downstream of other sites, incremental landscape characteristics were used. Basins were delineated using digital raster graphics (DRGs) of scanned USGS topographic quadrangle maps (1:24,000). Percent land use/cover was determined from the NH Land Cover Assessment 2001 (LC2001; derived from LANDSAT thematic mapper (TM) imagery acquired from 1990 to 1996 and classified by NH GRANIT) after this land use coverage was updated with National Wetland Inventory (NWI) coverages (provided by the US Fish and Wildlife Service) to improve the accuracy of wetland identification. Percent impervious surfaces were determined based on

the coastal NH 2005 assessment (derived from LANDSAT 7 TM imagery and classified by NH GRANIT). Human population densities were determined using US Census 2000 block level data.

Median annual net nitrogen inputs were determined for each basin over the 10 year study period. Inputs of food (5.0 kg N $person^{-1}$ yr^{-1}; value used by Boyer et al. 2002), fertilizer (both agricultural and non-agricultural fertilizer) and animal manure represent direct anthropogenic inputs to the basins. Non-agricultural application rates were estimated for Rockingham and Strafford county as the total non-agricultural fertilizer imported in 2001 (Ruddy et al. 2006) divided by the area of pervious residential, commercial, industrial and cleared land (e.g. golf courses, recreational fields) in the respective county as determined by updating the LC2001 (Grid Codes 110 and 790) with 2005 impervious surface data (Table 3.1). Agricultural fertilizer application rates were estimated for Rockingham County as total farm fertilizer imported during 2001 (Ruddy et al. 2006) divided by area of row crops and orchards (LC2001 Grid Codes 211 and 221) in Rockingham county (Table 3.1). Animal manure inputs were estimated for Rockingham County as total livestock manure amount in 1997 (Ruddy et al. 2006) divided by the area of hay/pasture land (LC2001 212) in Rockingham county (Table 3.1). Wet and dry deposition represents potential indirect anthropogenic inputs and median wet deposition for the study period was 4.69 kg N ha^{-1} yr^{-1}. Wet deposition from WY 2000-2003 was estimated as the average of the two nearest NADP/NTN stations (MA08 and MA13; http://nadp.sws.uiuc.edu/) and wet deposition from WY 2004-2009 was measured directly at TF in Durham, NH. Dry deposition was estimated as 58% of the wet deposition which is the ratio that ClimCalc (Ollinger et al. 2001; http://www.pnet.sr.unh.edu/climcalc/) predicts for TF in Durham, NH. Median total N deposition (wet + dry) for the entire basin was estimated as 7.41 kg N ha^{-1} yr^{-1}. Both direct and potential indirect N inputs were summed to determine total N inputs. In five basins, human waste was removed via municipal sewage treatment and the effluent was not returned to the basin. For these sewered basins, food N inputs from the number of persons served by sewer systems were removed from the total N input, resulting in net N inputs to the basins. One basin contained a wastewater treatment facility (Epping, NH) and we calculated the amount of food N input from the total number of persons in this basin since no waste was transferred out of this basin and was delivered to the Lamprey River upstream of our sampling site.

Table 3.1 Estimates of N input rates used for Lamprey and Oyster sub-basins.

Category	**Variable or Area Rate Applies to**	**Rate**
Food	Number of people serviced by septic systems or WWTFs that release effluent inside the basin	5 kg/person/yr
Non-Agricultural Fertilizer	Pervious residential, commercial and industrial (LC2001 Grid Code 110) and pervious other cleared (LC2001 Grid Code 790) area (ha) in Rockingham or Strafford county	21.45 kg/ha/yr in Rockingham County 16.98 kg/ha/yr in Strafford County
Agricultural Fertilizer	Area (ha) of row crops (LC2001 Grid Code 211) and orchards (LC2001 Grid Code 221)	183.63 kg/ha/yr
Animal Manure	Area (ha) of hay/pasture (LC2001 212)	31.29 kg/ha/yr
Atmospheric Deposition (wet +dry)	sub-basin area (ha)	7.41 kg N/ha/yr

Various models to predict N flux and N retention were developed based on two different approaches. Simple regression analysis was used to examine the relationships between N fluxes and retention and net N inputs. Both simple regression analysis and backwards step wise regression analysis (p to remove > 0.05) were used to examine the relationships between DIN, DON and TDN flux and landscape characteristics that represent potential sources or sinks. A similar approach was taken with N retention. For N fluxes, we identified human population density, impervious surface cover and agriculture as potential sources of N and wetland cover as a potential source or sink. For N retention, we considered % forest, % water and % wetland cover to be landscape features that may increase N uptake and % impervious surfaces as a feature that would decrease N uptake. Impervious surfaces are an indicator of overall human activity but also are an indicator of the amount of storm water that may be generated and delivered quickly to the receiving water body. Both TDN outputs and N outputs excluding DON flux were compared to net N inputs to determine the fraction of N retention ((inputs-outputs)/inputs). We normalized median DIN and TDN fluxes and basin human population density with log(x)-transformations before statistical analysis, and normalized % agriculture, % impervious surface and % water with the arcsine transformation (Sin^{-1} √decimal percent cover). Basin % wetland, % forest, DON flux, N inputs, N retention and N retention excluding DON

outputs were normally distributed and did not require transformation. All statistical analyses were performed using SPSS 17.0.

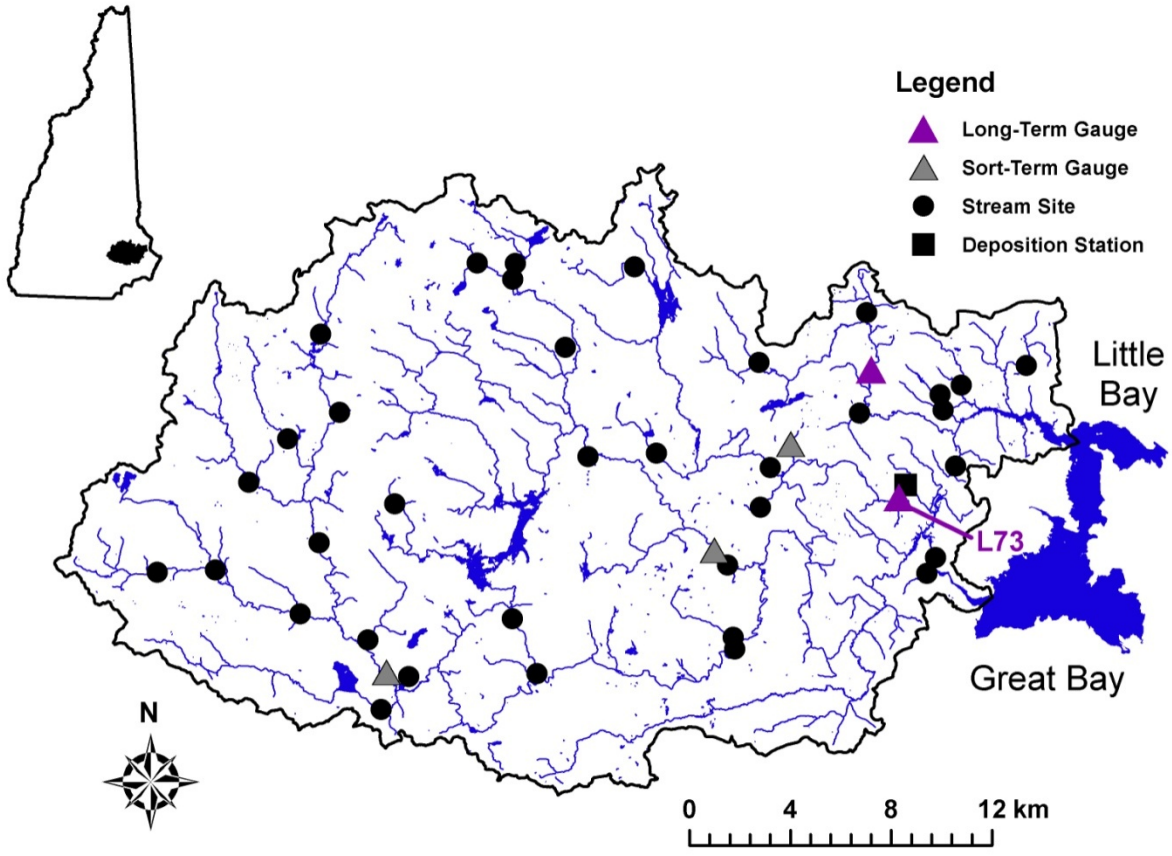

Fig. 3.1. Map of study sites. The Lamprey (USGS 1073500; site L73) and Oyster (01073000) River long-term gauges are shown, but samples were not collected from the Oyster River gauge. Other short-term gauges (including USGS station 01073460 and 01073319) were co-located with stream sampling sites. The deposition station at Thompson Farm (TF) is also shown.

3.1 MODELS THAT PREDICT N FLUX

3.1.1 N Outputs Compared to N Inputs

Median N inputs ranged from 8.05 to 24.9 kg N/ha/yr, median TDN flux ranged from 0.86 to 6.88 kg N/ha/yr and DIN flux accounted for 15 to 93% percent of TDN flux (Table 3.2). Increased N inputs resulted in increased median TDN ($r^2 = 0.62$, $p < 0.01$; Fig. 3.2) and DIN flux ($r^2 = 0.63$, $p < 0.01$; Fig. 3.3), but two sub-basins serviced completely by sewer systems that transfer human waste out of the basins were identified as outliers in the relationships. Based on

field experience and conversations with officials who are familiar with the sewer systems, we believe that there are either illicit discharges to these streams or leaks in the sewer lines resulting in the release of some of the human waste into the basin, violating our assumption that all sewage is transferred out of the basin. If these outliers were removed from the relationship, the variance explained by net N inputs in median TDN flux and median DIN flux would increase to 75 and 81%, respectively. Contrary to DIN flux, median DON flux does not respond to net N inputs (Fig. 3.4). The relationship between TDN flux and net N inputs (Fig. 3.2) must largely be driven by DIN flux which dominates TDN flux when net N inputs (and consequently TDN outputs) are high (Fig. 3.5).

Table 3.2. Basin characteristics, median N inputs, median N outputs (water years 2000-2009) and the fraction of N retention in Lamprey and Oyster sub-basins as well as the entire Lamprey River basin (L73).

	Basin Characteristics					N Inputs (kg N/ha/yr					N Outputs (kg N/ha/yr)				N Retention	
Station ID	Area (km^2)	People km^{-2}	% Forest	% Wetland	% Imperv.	Ag Fert.	Manure	non-Ag Fert.	Net Food	Total Net N Input	DON	DIN	TDN	% DIN	Fraction of N	Fraction Excluding DON
BVB-01	1.4	75.7	53.2%	20.7%	3.4%	0.00	1.96	1.76	3.78	14.92	1.20	0.73	1.93	38%	0.87	0.95
CB-02.2	1.9	398	26.3%	0.9%	23.0%	5.63	7.95	3.93	0.00	24.92	1.06	5.82	6.88	85%	0.72	0.77
CSB-02	3.74	116.3	51.5%	10.9%	4.7%	4.43	4.43	1.66	5.82	23.75	1.78	3.23	5.01	64%	0.79	0.86
DBE-02	3.42	63.4	66.6%	12.3%	3.8%	0.00	1.31	1.95	3.17	13.84	2.03	0.45	2.48	18%	0.82	0.97
DDB-06	6.33	51.4	82.5%	12.4%	0.5%	0.00	0.03	0.45	2.57	10.46	1.03	0.41	1.44	28%	0.86	0.96
FHB-03	1.81	58.3	68.9%	4.2%	4.3%	1.73	4.16	1.19	2.92	17.40	1.14	0.61	1.76	35%	0.90	0.96
GLT-04	3.53	105.2	60.7%	16.1%	4.1%	0.00	0.15	1.76	5.26	14.59	1.32	0.74	2.06	36%	0.86	0.95
HMB-05	11.3	50.6	75.9%	7.8%	3.7%	0.00	0.30	1.55	2.53	11.80	0.57	0.8	1.37	58%	0.88	0.93
HTB-09	21.85	23.3	81.9%	5.2%	2.4%	0.10	0.88	1.30	1.17	10.86	0.62	0.41	1.03	40%	0.91	0.96
JNC-03	5.41	365.2	40.2%	5.0%	17.7%	0.00	1.58	4.40	1.88	15.27	1.84	2.38	4.22	56%	0.72	0.84
LHB-01	1.26	634.5	68.6%	2.2%	13.3%	0.00	2.25	0.98	0.00	10.63	0.85	2.03	2.88	70%	0.73	0.81
LMB-02	1.27	42	58.7%	12.1%	2.7%	0.00	5.08	1.11	2.10	15.70	1.52	0.63	2.15	29%	0.86	0.96
LMP-07	15.17	20.3	87.1%	6.6%	1.1%	0.00	0.37	0.64	1.01	9.43	0.81	0.31	1.11	28%	0.88	0.97
LMP-19	35.02	38.8	71.7%	7.7%	4.7%	0.22	1.01	1.71	1.94	12.29	0.89	0.88	1.77	50%	0.86	0.93
LMP-27	16.65	81.2	61.8%	10.0%	6.2%	0.00	0.51	3.14	4.06	15.12	1.64	1.17	2.81	42%	0.81	0.92
LMP-39	25.8	167.5	46.3%	12.5%	14.0%	0.00	0.45	3.79	8.37	20.02	0.88	1.79	2.67	67%	0.87	0.91
LMP-51	27.84	116.5	56.2%	10.8%	9.0%	1.62	1.93	2.94	5.83	19.73	1.17	2.33	3.5	67%	0.82	0.88
LMP-78	109.7	87.1	57.6%	15.5%	6.2%	0.49	1.99	1.98	2.96	14.83	1.03	0.97	1.99	49%	0.87	0.93
LTR-05	5.47	24.2	82.3%	14.7%	0.4%	0.00	0.02	0.15	1.21	8.79	1.06	0.19	1.25	15%	0.86	0.98
LTR-20	42.75	49.2	71.6%	11.0%	4.6%	0.00	0.26	1.12	2.46	11.26	0.99	0.65	1.64	40%	0.85	0.94
MLB-01	0.91	797.8	50.1%	0.4%	25.5%	0.00	0.78	5.70	0.00	13.90	0.37	5.12	5.48	93%	0.61	0.63
MRW-02	1.83	27.2	74.4%	11.1%	0.9%	0.11	2.02	1.04	1.36	11.94	1.56	0.3	1.87	16%	0.84	0.97
NBR-07	12.04	17.8	76.9%	9.5%	1.5%	0.00	0.38	1.46	0.89	10.15	1.05	0.26	1.32	20%	0.87	0.97
NBR-12	17.86	53.3	74.5%	4.9%	5.1%	0.48	0.59	2.26	2.66	13.40	1.12	0.38	1.5	25%	0.89	0.97
NCB-07	6.58	36.2	78.8%	6.9%	2.0%	1.10	1.36	0.99	1.81	12.67	0.83	0.4	1.23	32%	0.90	0.97

Table 3.2 Continued

	Basin Characteristics					N Inputs (kg N/ha/yr					N Outputs (kg N/ha/yr)				N Retention	
Station ID	Area (km^2)	People km^{-2}	% Forest	% Wetland	% Imperv.	Ag Fert.	Manure	non-Ag Fert.	Net Food	Total Net N Input	DON	DIN	TDN	% DIN	Fraction of N	Fraction Excluding DON
NOR-17	77.98	19.9	75.1%	12.8%	1.8%	0.02	0.34	0.89	1.00	9.65	1.44	0.3	1.74	17%	0.82	0.97
NOR-27	31.06	36.1	70.7%	9.8%	3.4%	1.79	1.70	1.36	1.80	14.06	0.52	0.91	1.43	64%	0.90	0.94
NWR-03	5.15	80.9	72.6%	3.9%	6.7%	0.37	1.87	1.96	4.04	15.65	0.63	1.08	1.71	63%	0.89	0.93
NWR-05	5.18	51.9	71.2%	7.3%	5.0%	0.00	0.67	1.68	2.59	12.35	1.13	0.24	1.37	18%	0.89	0.98
OLT-13	23.67	60.6	67.1%	9.9%	5.7%	0.06	0.54	1.88	3.03	12.92	1.03	0.57	1.6	36%	0.88	0.96
OYS-04	11.75	91.5	74.3%	11.8%	3.8%	0.00	0.31	1.31	4.58	13.61	1.23	0.53	1.76	30%	0.87	0.96
PB-02.0	2.19	387.9	46.8%	10.5%	16.2%	4.94	3.22	3.12	0.00	18.69	1.40	3.24	4.64	70%	0.75	0.83
PPB-03	3.55	18.7	78.9%	14.3%	0.8%	0.00	0.26	0.37	0.94	8.97	1.19	0.25	1.44	17%	0.84	0.97
PWT-03	2.58	4.4	81.4%	14.2%	0.1%	0.00	0.00	0.42	0.22	8.05	0.92	0.16	1.08	15%	0.87	0.98
PWT-10	23.36	32.6	73.9%	7.2%	2.2%	0.00	0.35	0.69	1.63	10.09	0.49	0.36	0.86	42%	0.91	0.96
RMB-04	4.85	103.1	61.8%	11.7%	7.7%	0.00	2.11	2.09	5.16	16.76	1.18	1.11	2.29	49%	0.86	0.93
WHB-01	1.26	157.4	54.0%	5.7%	13.5%	0.12	3.51	1.97	7.87	20.87	0.91	4.01	4.92	81%	0.76	0.81
L73	**479.3**	**53.4**	**69.5%**	**10.2%**	**4.4%**	**0.34**	**0.90**	**1.63**	**2.67**	**12.96**	**1.30**	**0.77**	**2.07**	**37%**	**0.84**	**0.94**
Min	0.91	4.4	26.3%	0.4%	0.1%	0.00	0.00	0.15	0.00	8.05	0.37	0.16	0.86	15%	0.61	0.63
Max	109.7	797.8	87.1%	20.7%	25.5%	5.63	7.95	5.70	8.37	24.92	2.03	5.82	6.88	93%	0.91	0.98

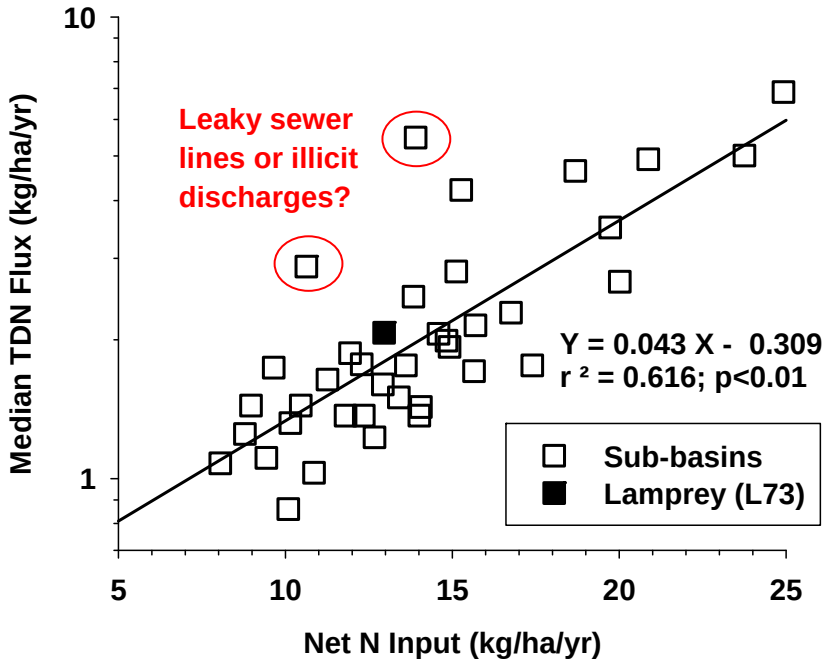

Fig. 3.2 Median TDN flux from 2000-2009 (water years) vs. net N inputs. The entire Lamprey River basin (L73) was not included in the linear regression analysis, but is shown here for comparison. Basins that may contain leaky sewer lines or illicit discharges were included in linear regression analysis and were identified as outliers. Regression equation improves (r^2 = 0.75, p <0.01; Log Y = 0.045 X – 0.359) when they are excluded.

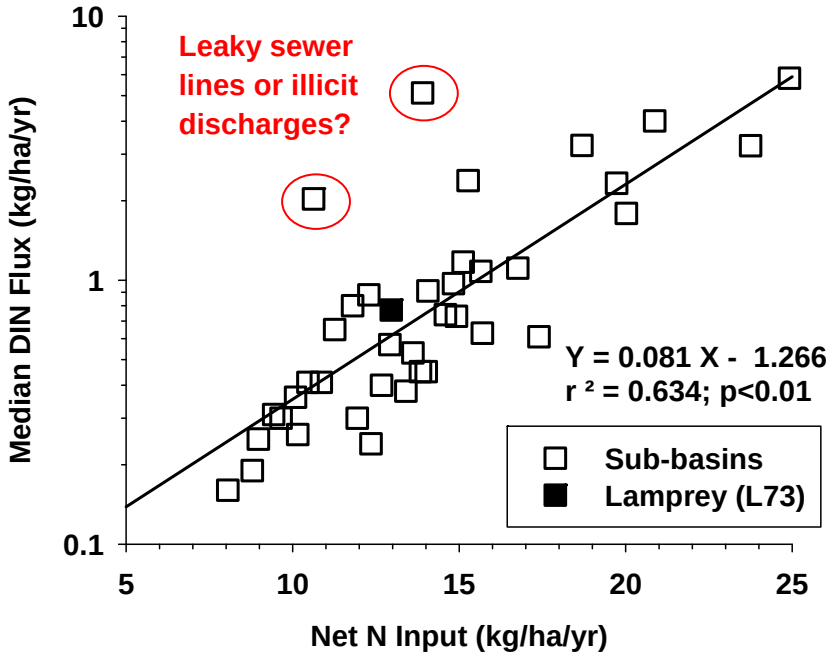

Fig. 3.3 Median DIN flux from 2000-2009 (water years) vs. net N inputs. The entire Lamprey River basin (L73) was not included in the linear regression analysis, but is shown here for comparison. Basins that may contain leaky sewer lines or illicit discharges were included in linear regression analysis and were identified as outliers. Regression equation improves (r^2 = 0.81, p <0.01; Log Y = 0.086 X - 1.373) when they are excluded.

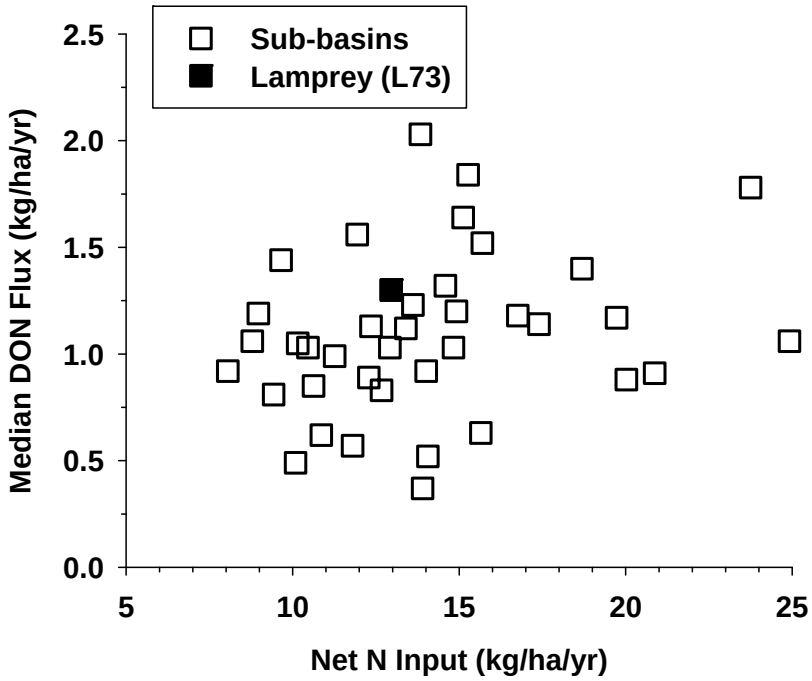

Fig. 3.4 Median DON flux from 2000-2009 (water years) was not related to net N inputs. The entire Lamprey River basin (L73) was not included in the linear regression analysis, but is shown here for comparison.

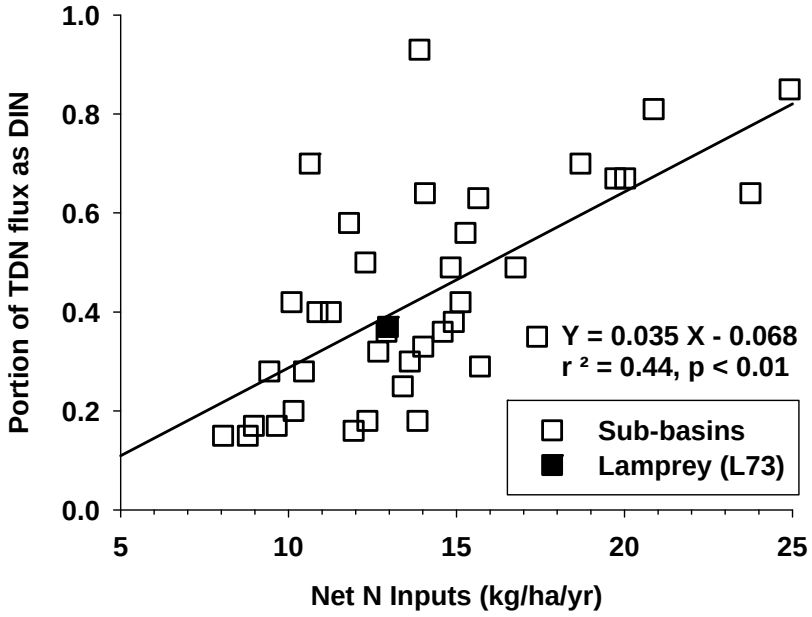

Fig. 3.5 Portion of median TDN flux as DIN flux vs. net N inputs. The entire Lamprey River basin (L73) was not included in the linear regression analysis, but is shown here for comparison.

3.1.1 Landscape Characteristics that predict N Flux

Landscape characteristics were strong predictors of DIN and TDN flux and significant, but weaker predictors of DON flux. Human population density was the single best predictor of median DIN flux ($r^2 = 0.76$, $p<0.01$; Fig. 3.6) and human population density, % impervious surface and % agriculture were retained in the backwards multiple regression model ($R^2=0.86$, $p<0.01$; Table 3.3). This indicates that each of these three variables explains a significant amount of unique variance in DIN flux. Arcsine impervious surface was the single best predictor of median TDN flux ($r^2 = 0.68$, $p<0.01$; Fig. 3.7) and arcsine impervious, arcsine agriculture and % wetlands were retained in the backwards multiple regression model ($R^2= 0.78$, $p<0.01$; Table 3.3). However, a sub-basin with one of the highest population densities and serviced by septic systems was identified as an outlier. This sub-basin was not an outlier in the relationship between TDN flux and human population density ($r^2 = 0.67$, $p<0.01$; Fig. 3.8) and even though human population density explains 1% less variance in TDN flux we conclude that human population density may be a better predictor of TDN flux than impervious cover. What is striking about the relationships between both DIN flux and TDN flux and human population density is that even though 5 of the sub-basins are more than 90% sewered and the sewage is transferred out of the basin, these sub-basins were not identified as outliers in the regression relationships. Percent wetland cover was a significant, but weak predictor of median DON flux ($r^2 = 0.14$, $p<0.05$; Fig. 3.9) and % wetlands and arcsine agriculture were retained in the backwards multiple regression model ($R^2= 0.28$, $p<0.01$; Table 3.3). These results show that DIN and DON respond to different factors in the landscape. DIN responds to human N inputs and DON weakly responds to natural features in the landscape. This has strong management implications for N reduction strategies in the Great Bay watershed and suggests that reductions of non-point source N inputs will not reduce DON flux in contributing rivers. The relationship between TDN flux and human population density must largely be driven by DIN flux which dominates TDN flux at high population densities (Fig. 3.10).

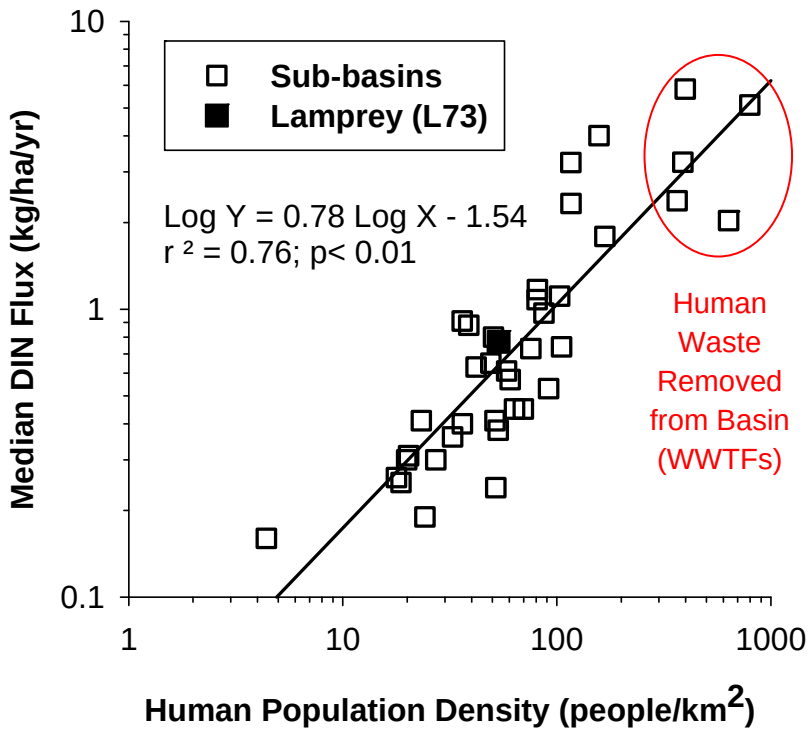

Fig. 3.6 Median DIN flux from 2000-2009 (water years) vs. human population density. The entire Lamprey River basin (L73) was not included in regression models, but is shown here for comparison.

Table 3.3 Regression models that predict N flux based on landscape characteristics.

Dependent Variable	Regression Model	Independent Variable(s)	Coefficient	Lower 95% CI	Upper 95% CI
Log Median DIN Flux (kg/ha/yr)	Simple $r^2 = 0.763$**	Constant	-1.539	-1.814	-1.264
		Log Population Density	0.778	0.631	0.924
	Multiple $R^2=0.857$**	Constant	-1.152	-1.471	-0.832
		Log Population Density	0.262	-0.034	0.557
		Arcsine % Impervious	0.768	0.250	1.286
		Arcsine % Agriculture	1.767	0.593	2.940
Log Median TDN Flux (kg/ha/yr)	Simple $r^2 = 0.684$**	Constant	-0.053	-0.145	0.038
		Arcsine % Impervious	1.560	1.202	1.918
	Multiple $R^2=0.783$**	Constant	-0.285	-0.435	-0.134
		Arcsine % Impervious	1.368	0.451	2.286
		Arcsine % Agriculture	0.566	0.235	0.897
		% Wetlands	1.518	1.137	1.898
Median DON Flux (kg/ha/yr)	Simple $r^2 = 0.139$*	Constant	0.788	0.510	1.066
		% Wetlands	0.032	0.005	0.059
	Multiple $R^2= 0.275$**	Constant	0.450	0.077	0.822
		% Wetlands	0.043	1.620	6.897
		Arcsine % Agriculture	1.199	0.251	2.148

* $p < 0.05$
** $p < 0.01$

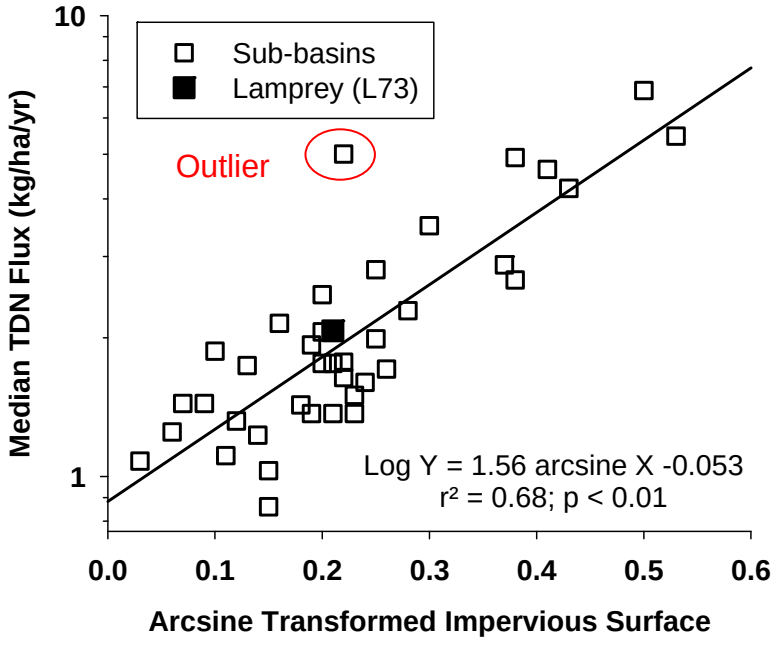

Fig. 3.7 Median TDN flux from 2000-2009 (water years) vs. arcsine impervious surface (Sin^{-1} √decimal percent cover). The entire Lamprey River basin (L73) was not included in regression models, but is shown here for comparison.

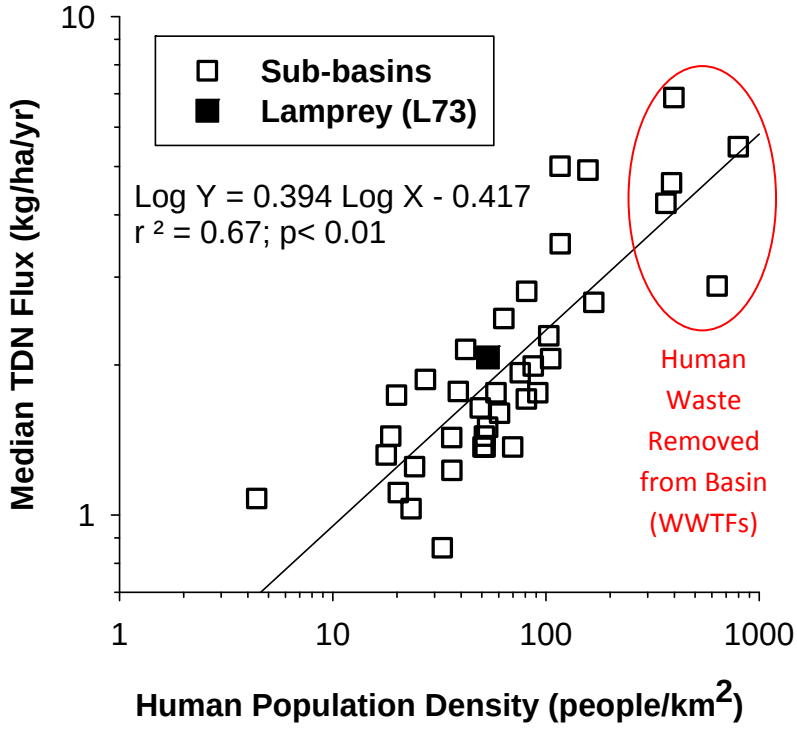

Fig. 3.8 Median TDN flux from 2000-2009 (water years) vs. human population density. The entire Lamprey River basin (L73) was not included in regression models, but is shown here for comparison.

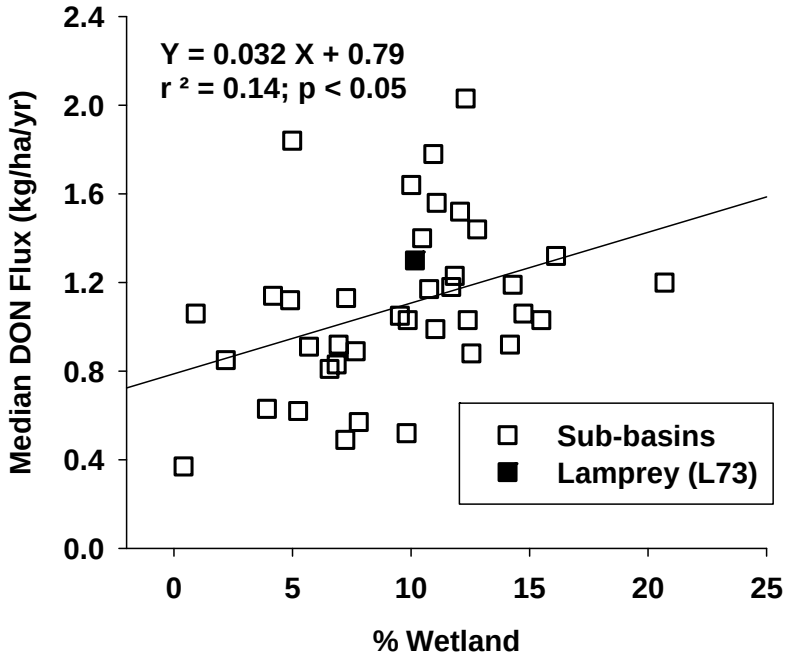

Fig. 3.9 Median DON flux from 2000-2009 (water years) vs. % wetland cover. The entire Lamprey River basin (L73) was not included in regression models, but is shown here for comparison.

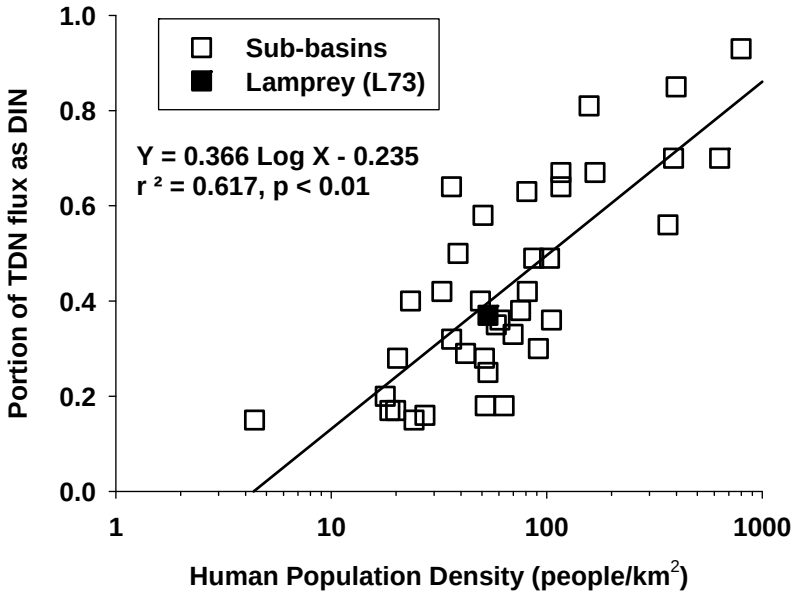

Fig. 3.10 Portion of median TDN flux as DIN flux vs. human population density. The entire Lamprey River basin (L73) was not included in the linear regression analysis, but is shown here for comparison.

Global models have been developed to predict both nitrate concentration and flux in large rivers worldwide. Peierls et al. (1991) developed the global human population density model which predicts NO_3-N concentration and flux based on watershed human population density. This model over-predicts nitrate concentration for most Lamprey and Oyster sub-basins (Fig. 3.11) and also over-predicts nitrate flux for some basins (Fig. 3.12), although to a lesser extent than nitrate concentration. Caraco and Cole (1999) developed the global human activity model to predict NO_3-N flux based on the equation below. This model also over-predicted NO_3-N flux for most Lamprey and Oyster sub-basins (Fig. 3.13). These global models that were developed based on large river systems (e.g. the Mississippi and Nile Rivers) may under-estimate the amount of storage or N loss through denitrification in smaller watersheds. Contrary to our findings, Caraco and Cole (2003) found that the human population density model becomes weaker at smaller scales and can only explain 8% of the 1000-fold variation in NO_3-N flux among basins less than 100 km^2. Lamprey and Oyster sub-basins ranged in size from 0.9 to 110 km^2 and population density alone could explain 76% of the ~100-fold variation in NO_3-N flux.

Human Activity Model (Caraco and Cole 1999):

$$NO_3\text{-N Flux (kg N km}^{-2}\text{ yr}^{-1}) = 0.7 \times [\text{Point Inputs} + (0.4 \times WL^{0.8}) \times (\text{WS Inputs})]$$

where:

Point Inputs = human sewage (kg N km^{-2} yr^{-1})

WS Inputs = Watershed Inputs (e.g. fertilizer application, septic inputs and atmospheric deposition; (kg N km^{-2} yr^{-1})

WL = average water output from watershed in m yr^{-1}

Others have found agriculture to be a significant driver of N export and concentrations. Among basins in northeastern US, % agriculture alone could explain 77% of the variability in N export (Boyer et al.2002). Agriculture was retained in the multiple regression models that predict TDN and DIN export in the Lamprey and Oyster sub-basins (Table 3.3), but alone could only explain 42 and 37 % of the variability, respectively. In Finland, agriculture was also positively correlated to DON flux (Mattsson et al. 2005). Although agriculture alone was not a significant predictor of DON flux in southeast NH, it was retained in the multiple regression analysis as being positively related to DON flux. Others have found that DON concentrations are positively related to wetland cover (Daley et al. 2002, $r^2 = 0.65$, $p < 0.01$; Pellerin et al. 2004, $r^2 = 0.56$, $p < 0.01$) in NH and MA basins. Pellerin et al. (2004) noted that excluding basins with direct human and animal wastewater inputs to surface waters improved the percentage of variability in DON concentrations explained by wetland cover from 60% to 79% among northeastern US basins. Wetland cover was not as strong a predictor of median DON flux ($r^2 = 0.14$, Fig. 3.9) or concentration ($r^2 = 0.19$, $p < 0.01$) among Lamprey and Oyster sub-basins as it was in the northeastern US basins (Pellerin et al. 2004) or in an earlier Lamprey River study

(Daley 2002). It is possible that the four-fold fluctuation in annual runoff (0.26 to 1.06 m/yr) over the last ten years (water years 2000-2009) made it difficult to compare 10 year median DON fluxes estimated from steam sites sampled in different years since DON is negatively related to stream flow in the Lamprey ($r^2 = 0.05$, $p < 0.01$). Lamprey DIN concentrations were not significantly related to stream flow and we believe that sampling different sites in different years had little influence on 10 year median DIN fluxes in the sub-basins.

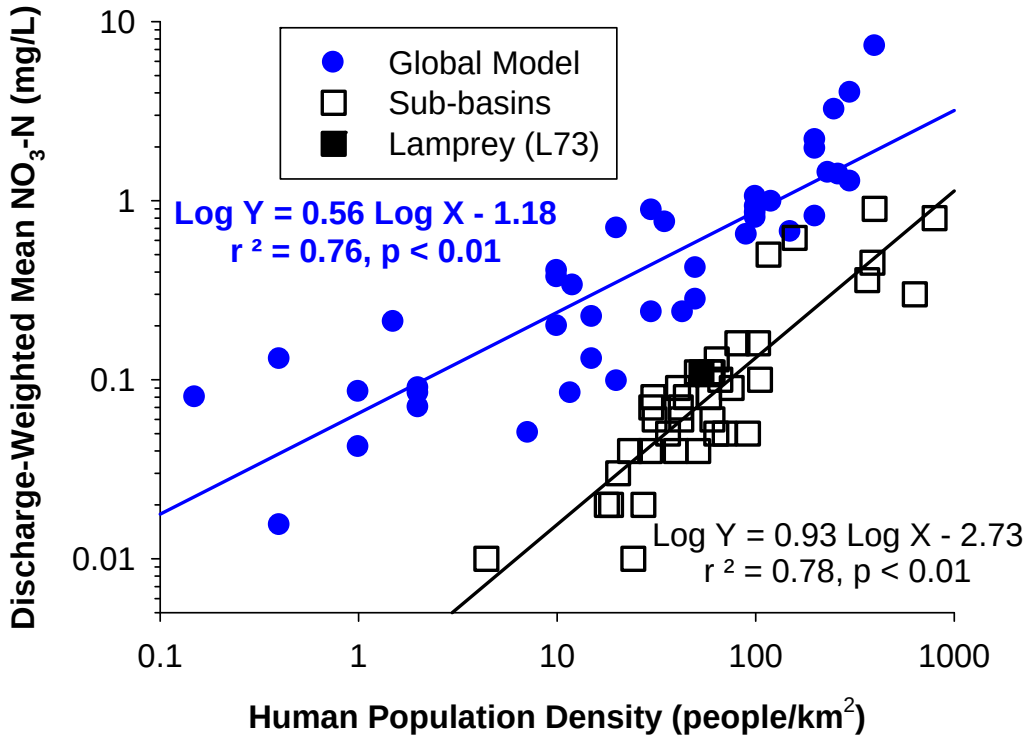

Fig. 3.11. Relationship between discharge-weighted mean NO_3-N (mg/L) for Lamprey and Oyster sub-basins and cumulative sub-basin human population density compared to the global population density model (Peierls et al. 1991).

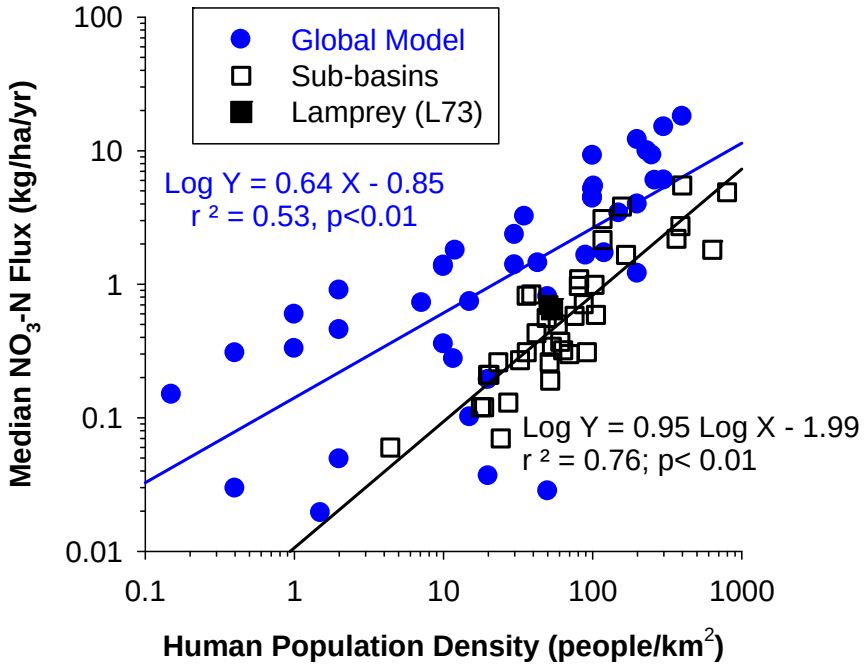

Fig. 3.12. Relationship between median NO_3-N flux for Lamprey and Oyster sub-basins compared to the global population density model (Peierls et al. 1991).

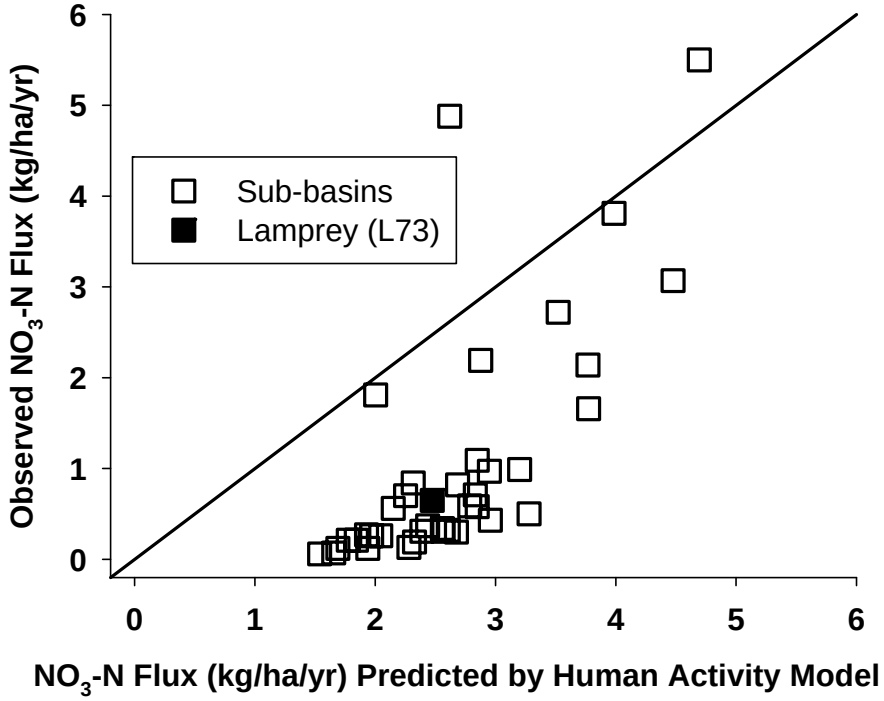

Fig. 3.13. Observed NO_3-N flux compared to NO3-N flux predicted by global human activity model (Caraco and Cole 1999). The 1:1 line is shown.

3.2 MODELS THAT PREDICT N RETENTION

3.2.1 N Retention Compared to N Inputs

Watershed N retention is fairly high among the sub-basins (72 – 91 % in most sub-basins, as low as 61% in one sub-basin), but declines with increased N inputs (Fig. 3.14). There is a stronger decline in retention with increased N inputs when DON flux (which does not respond to N inputs, Fig. 3.4) is excluded from outputs used in calculating N retention (Fig. 3.15). Nitrogen retention excluding DON outputs ranges from 75 to 98% in most sub-basins. This decline in watershed retention (which represents both in-stream and terrestrial retention) with increased N inputs is analogous to the decrease in efficiency (V_f) of overall N uptake and denitrification that occurs in stream networks (Mulholland et al. 2008). In addition to instream losses, these watershed-wide retention rates may also be attributed to N storage in soils, vegetation and groundwater, or denitrification in riparian zones or elsewhere in the basin.

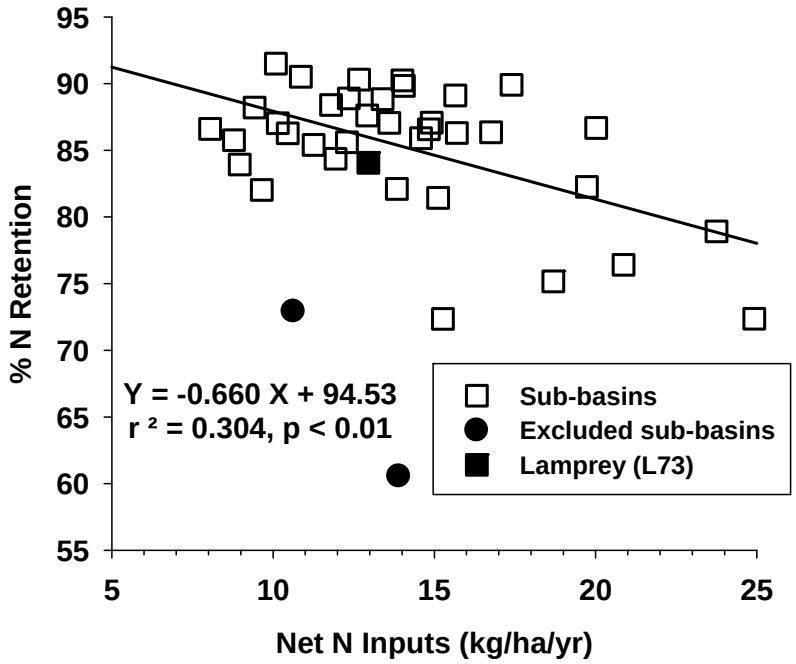

Fig. 3.14. Percent N retention vs. net N inputs. The entire Lamprey River basin (L73) was not included in the linear regression analysis, but is shown here for comparison. Outliers were excluded from the linear regression analysis but are identified as basins that may contain leaky sewer lines or illicit discharges.

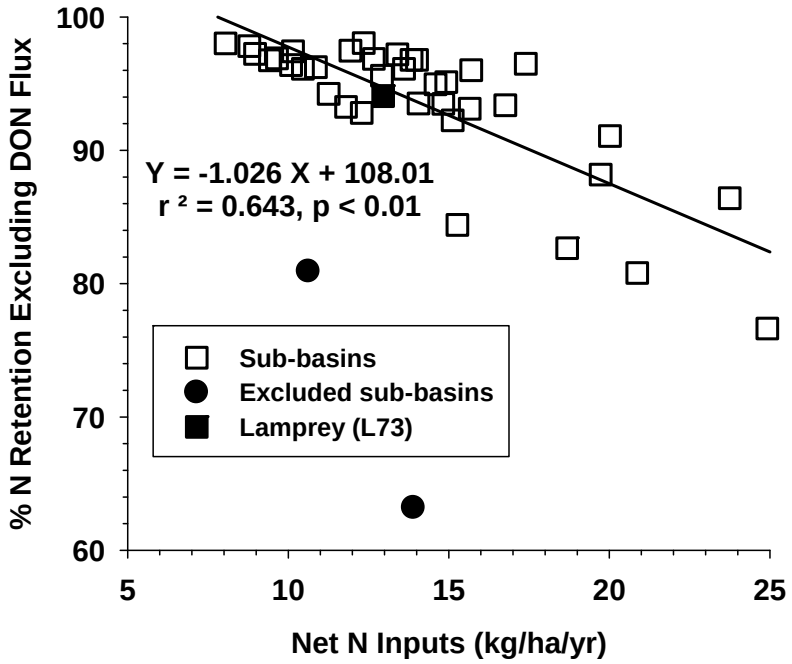

Fig. 3.15. Percent N retention excluding DON flux vs. net N inputs. The entire Lamprey River basin (L73) was not included in the linear regression analysis, but is shown here for comparison. Outliers were excluded from the linear regression analysis but are identified as basins that may contain leaky sewer lines or illicit discharges.

Relatively high watershed retention rates have also been documented in other studies. Boyer et al. (2002) found that among northeastern US basins (Maine to Virginia) river N outputs were well correlated with inputs, but that outputs only accounted for 25% of the inputs on average. The range of N inputs to Lamprey and Oyster sub-basins (8.05 to 24.9 kg N/ha/yr; Table 3.2) was in the lower range of N inputs to northeastern US basins (8.35 to 57.2 kg N/ha/yr) measured by Boyer et al. (2002), and N outputs from Lamprey and Oyster sub-basins (0.86 to 6.68 kg N/ha/yr) were also lower than N outputs measured by Boyer et al. (3.14 to 17.56 kg/ha/yr). Despite the differences in the ranges of inputs and exports, the Lamprey and Oyster sub-basins had the same range of retention rates (61 to 92%) as found by Boyer et al. (2002; 60 to 89%). At the Baltimore long-term ecological research (LTER) ecosystem, retention of N inputs was highest in the forested basin (95%) and surprisingly high in the sub-urban (75%) and agricultural (77%) basins. Inputs ranged from 11.2 kg N/ha/yr in the forested basin to 71.2 kg N/ha/yr in the Agricultural basin and outputs ranged from less than 1 (forested) to71.2 kg/ha/yr (agricultural) in basins of the Baltimore LTER (Groffman 2004). At the Plum Island Ecosystem LTER in MA watershed N retention was 65–85% in the urban basin and was 93–97% in the forested basin (Wollheim et al. 2005). Inputs ranged from 17.7 to 25.8 kg N/ha/yr and outputs ranged from 0.59 to 5.79 kg N/ha/yr among these forested and urban basins, respectively (Wollheim et al. 2005).

These large retention rates found among the Lamprey and Oyster sub-basins as well as in other studies imply that a large fraction of N inputs is either removed (e.g. denitrified) from watersheds or stored in vegetation, soils or groundwater. We have seen elevated groundwater nitrate in the Lamprey basin (Fig. 1.8) suggesting that N is being stored, and groundwater could also be driving a long-term increase in Lamprey River nitrate (Fig. 3.16). This increase in nitrate over time could be a "bleeding out" of the comparatively high groundwater N pool or a response to increased human population density over time. In concert with this long-term increase in nitrate, we may see a corresponding decline in N retention over time. However, at this time we cannot accurately estimate the annual variability in net N inputs (e.g. fertilizer application and manure inputs) to accurately assess any long-term changes in N retention.

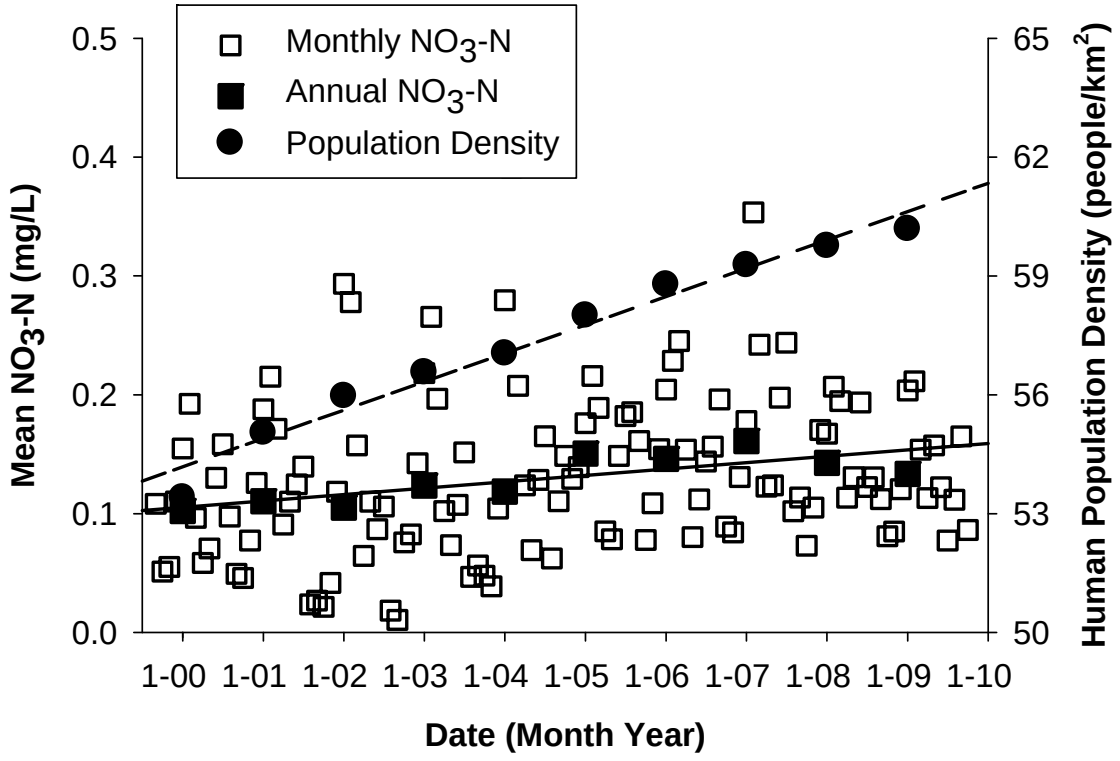

Fig. 3.16 Monthly and annual (water year) nitrate concentration and estimated annual human population density over time in the Lamprey River basin (L73). We have applied the Seasonal–Kendall Test (SKT; seasons set to 52) to weekly data from Sept. 1999 through Sept. 2009 and flow-adjusted nitrate concentrations have increased significantly over this time period (SKT $t = 0.27$, $p < 0.01$). The trend through mean annual concentrations is shown. Annual estimates of population density were generated using town wide rates of increase calculated from US Census data and applying these rates to population densities within the Lamprey basin calculated from 2000 census block data on a town by town basis.

3.2.2 Landscape Characteristics that predict N Retention in Lamprey and Oyster basins

Landscape characteristics were also significant predictors of N retention. Forest cover was the single best predictor of N retention ($r^2 = 0.51$, $p<0.01$; Fig. 3.17) and % forest cover and arcsine transformed % water were retained in the backwards multiple regression model ($R^2=0.58$, $p<0.01$; Table 3.4). Arcsine transformed % impervious surface cover was the single best predictor of N retention excluding DON flux ($r^2 = 0.72$, $p<0.01$; Fig. 3.18) and % forest cover, arcsine transformed % impervious surface cover and arcsine transformed % water were retained in the backwards multiple regression model ($R^2=0.78$, $p<0.01$; Table 3.4). These results suggest that forests and streams, rivers, lakes, ponds and water pooled up behind dams are areas where N retention can occur. Impervious surfaces often reduce N retention by by-passing forests and other potential areas for terrestrial N retention and deliver N quickly to surface waters. According to the backwards multiple regression models, forests have a greater influence on N retention and N retention excluding DON flux than surface water (i.e. forests have a larger standardized β than water; Table 3.5), but our results suggest both are significant mechanisms for N retention.

Table 3.4 Regression models that predict N retention based on landscape characteristics.

Dependent Variable	Regression Model	Independent Variable(s)	Coefficient	Lower 95% CI	Upper 95% CI
% N Retention	Simple $r^2 = 0.508$ **	Constant % Forest	67.857 0.260	61.791 0.171	73.924 0.349
	Multiple $R^2=0.579$ **	Constant % Forest Arcsine % Water	67.617 0.228 16.552	61.910 0.140 2.272	73.323 0.317 30.832
% N Retention Excluding DON Flux	Simple $r^2 = 0.723$**	Constant Arcsine % Impervious	102.467 -41.826	100.315 -50.851	104.619 -32.801
	Multiple $R^2=0.782$**	Constant % Forest Arcsine % Impervious Arcsine % Water	87.093 0.143 -22.496 11.946	74.461 0.009 -39.948 0.458	99.725 0.277 -5.045 23.434

** $p < 0.01$

Table 3.5 Standardized coefficients for models that predict N retention based on landscape characteristics. Standardized β allows for comparison of the relative influence of independent variables on the dependent variable regardless of the units of the independent variable.

Dependent Variable	Regression Model	Independent Variable(s)	Standardized Coefficient (β)
% N Retention	Simple	% Forest	0.713
	Multiple	% Forest	0.626
		Arcsine % Water	0.280
% N Retention Excluding DON Flux	Simple	Arcsine % Impervious	-0.850
	Multiple	% Forest	0.367
		Arcsine % Impervious	-0.457
		Arcsine % Water	0.189

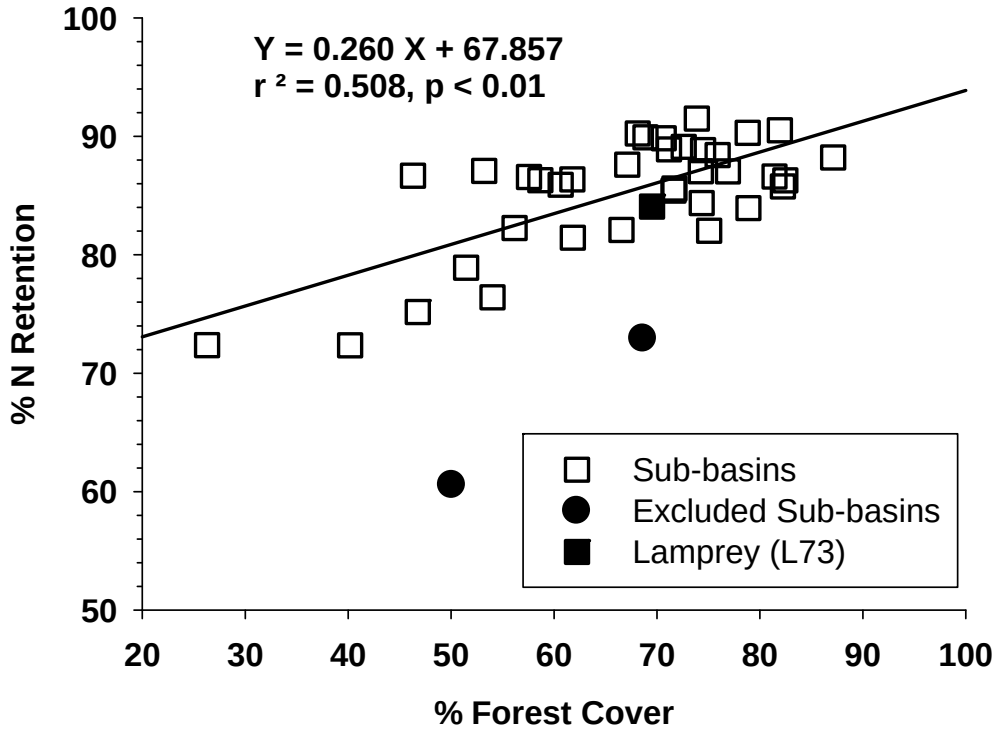

Fig. 3.17. Percent N retention vs. %forest cover. The entire Lamprey River basin (L73) was not included in the linear regression analysis, but is shown here for comparison. Outliers were excluded from the linear regression analysis but are identified as basins that may contain leaky sewer lines or illicit discharges.

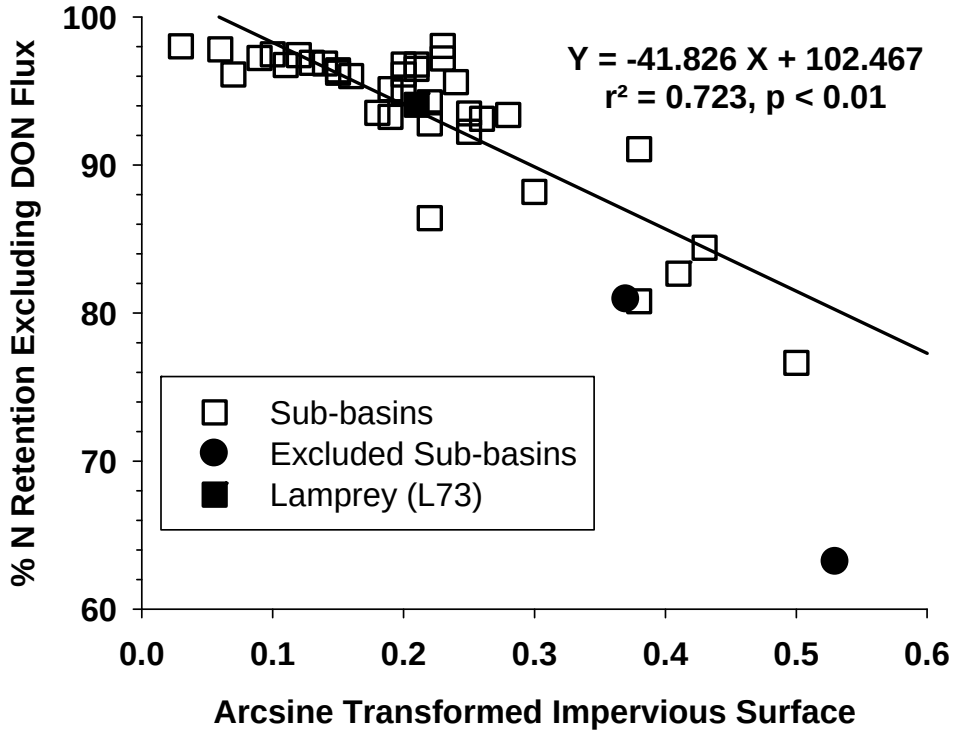

Fig. 3.18. Percent N retention excluding DON flux vs. arcsine transformed impervious surface cover. The entire Lamprey River basin (L73) was not included in the linear regression analysis, but is shown here for comparison. Outliers were excluded from the linear regression analysis but are identified as basins that may contain leaky sewer lines or illicit discharges.

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Appendix A. Annual precipitation at NCDC (http://www.ncdc.noaa.gov/oa/ncdc.html) and CRN (http://www.ncdc.noaa.gov/crn/) stations located within the Piscataqua River watershed (Dist = 0) or within 25 km (Dist = <25) from its border. Data for calendar years (CY) and water years (WY) from 1980-2009 are included.

SOURCE	STATION NAME	COUNTY	STATE	ELEV. (m)	DIST. (km)	YEAR	CY PPT. (mm)	WY PPT. (mm)
NCDC	ELIOT	YORK	ME	6.1	0	1993	1036	1014
NCDC	ELIOT	YORK	ME	6.1	0	1995	1147	965
NCDC	ELIOT	YORK	ME	6.1	0	1996	1535	1414
NCDC	ELIOT	YORK	ME	6.1	0	1997	1130	1444
NCDC	ELIOT	YORK	ME	6.1	0	1998	1318	1337
NCDC	ELIOT	YORK	ME	6.1	0	1999	1055	1079
NCDC	ELIOT	YORK	ME	6.1	0	2000	1253	1148
NCDC	ELIOT	YORK	ME	6.1	0	2001	985	1126
NCDC	ELIOT	YORK	ME	6.1	0	2002	1130	935
NCDC	ELIOT	YORK	ME	6.1	0	2003	1084	1149
NCDC	ELIOT	YORK	ME	6.1	0	2004	1107	1168
NCDC	ELIOT	YORK	ME	6.1	0	2006	1770	1919
NCDC	ELIOT	YORK	ME	6.1	0	2007	1417	1500
NCDC	ELIOT	YORK	ME	6.1	0	2008	1653	1687
NCDC	ELIOT	YORK	ME	6.1	0	2009	1438	1371
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1980	803	837
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1981	1120	1040
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1982	928	1054
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1983	1424	1066
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1984	1140	1414
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1985	762	826
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1986	1051	959
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1987	1015	1119
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1988	1144	1030
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1989	1094	1120
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1990	1173	1032
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1991	1191	1345
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1992	950	934
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1993	924	887
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1994	1062	1160
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1995	1048	850
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1996	1328	1246
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1997	1022	1250
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1998	1100	1158
NCDC	DURHAM	STRAFFORD	NH	24.4	0	1999	983	1007
NCDC	DURHAM	STRAFFORD	NH	24.4	0	2000	1110	1067
NCDC	DURHAM	STRAFFORD	NH	24.4	0	2001	625	712

Appendix A. Continued

SOURCE	STATION NAME	COUNTY	STATE	ELEV. (m)	DIST. (km)	YEAR	CY PPT. (mm)	WY PPT. (mm)
NCDC	DURHAM	STRAFFORD	NH	24.4	0	2002	1018	837
NCDC	DURHAM	STRAFFORD	NH	24.4	0	2003	1155	1118
NCDC	DURHAM	STRAFFORD	NH	24.4	0	2004	1105	1248
NCDC	DURHAM	STRAFFORD	NH	24.4	0	2005	1494	1144
NCDC	DURHAM	STRAFFORD	NH	24.4	0	2006	1550	1675
NCDC	DURHAM	STRAFFORD	NH	24.4	0	2007	1222	1364
NCDC	DURHAM	STRAFFORD	NH	24.4	0	2008	1748	1679
NCDC	DURHAM	STRAFFORD	NH	24.4	0	2009	1307	1324
CRN	Durham 2 SSW	STRAFFORD	NH	19.2	0	2003	1121	
CRN	Durham 2 SSW	STRAFFORD	NH	19.2	0	2004	1166	1209
CRN	Durham 2 SSW	STRAFFORD	NH	19.2	0	2005	1650	1307
CRN	Durham 2 SSW	STRAFFORD	NH	19.2	0	2006	1661	1822
CRN	Durham 2 SSW	STRAFFORD	NH	19.2	0	2007	1199	1311
CRN	Durham 2 SSW	STRAFFORD	NH	19.2	0	2008	1643	1611
CRN	Durham 2 SSW	STRAFFORD	NH	19.2	0	2009	1321	1338
CRN	Durham 2N	STRAFFORD	NH	36.3	0	2003	1082	
CRN	Durham 2N	STRAFFORD	NH	36.3	0	2004	1095	1157
CRN	Durham 2N	STRAFFORD	NH	36.3	0	2005	1540	1175
CRN	Durham 2N	STRAFFORD	NH	36.3	0	2006	1519	1681
CRN	Durham 2N	STRAFFORD	NH	36.3	0	2007	1243	1383
CRN	Durham 2N	STRAFFORD	NH	36.3	0	2008	1644	1617
CRN	Durham 2N	STRAFFORD	NH	36.3	0	2009	1285	1279
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	1980	809	833
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	1981	1183	1074
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	1982		1169
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	1983	1412	
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	1984	1184	1312
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	1985	921	961
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	1986	1172	1064
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	1988	1234	1111
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	1989	1106	1148
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	1990	1204	1101
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	1991	1198	1318
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	1992	959	924
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	1994	1101	1176
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	1995	976	812
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	1996	1438	1323
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	1997	990	1224
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	1998	1102	1184
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	1999	1029	1055

Appendix A. Continued

SOURCE	STATION NAME	COUNTY	STATE	ELEV. (m)	DIST. (km)	YEAR	CY PPT. (mm)	WY PPT. (mm)
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	2000		1116
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	2002	1089	878
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	2006	1680	1855
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	2007	1190	1323
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	2008	1584	1531
NCDC	EPPING	ROCKINGHAM	NH	48.8	0	2009	1280	1299
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	1980	910	885
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	1982	1037	1243
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	1983	1810	1337
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	1984	1368	1694
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	1985	1020	1030
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	1986	1287	1156
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	1987	1184	1452
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	1988	1150	1031
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	1989	1163	1108
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	1990	1354	1199
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	1991	1342	1570
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	1992	997	983
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	1993	1092	1035
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	1994	1182	1293
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	1995	1114	933
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	1996	1549	1342
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	1997	1107	1423
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	1998	1354	1396
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	1999	1138	1186
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	2000	1364	1213
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	2001	1007	1211
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	2002	1159	938
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	2003	1201	1264
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	2004	1172	1241
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	2005	1509	1191
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	2006	1896	2030
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	2007	1305	1411
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	2008	1722	1711
NCDC	GREENLAND	ROCKINGHAM	NH	25.9	0	2009	1428	1370
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	1990	1281	1045
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	1991	1351	1505
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	1992	999	1040
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	1993	1213	1111
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	1994	1196	1318

Appendix A. Continued

SOURCE	STATION NAME	COUNTY	STATE	ELEV. (m)	DIST. (km)	YEAR	CY PPT. (mm)	WY PPT. (mm)
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	1995	1149	916
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	1996	1567	1516
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	1997	1331	1577
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	1998	1306	1379
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	1999	1221	1178
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	2000	1299	1242
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	2001	980	1137
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	2002	1091	900
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	2003	1181	1196
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	2004	1244	1302
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	2005	1734	1339
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	2006	1655	1839
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	2007	1311	1481
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	2008	1786	1756
NCDC	ROCHESTER	STRAFFORD	NH	70.1	0	2009	1411	1357
NCDC	WEST HAMPSTEAD	ROCKINGHAM	NH	91.4	0	2007	1185	
NCDC	WEST HAMPSTEAD	ROCKINGHAM	NH	91.4	0	2008	1597	1576
NCDC	WEST HAMPSTEAD	ROCKINGHAM	NH	91.4	0	2009	1279	1273
NCDC	GROVELAND	ESSEX	MA	10.1	< 25	1993	1166	1143
NCDC	GROVELAND	ESSEX	MA	10.1	< 25	1994	1216	1300
NCDC	GROVELAND	ESSEX	MA	10.1	< 25	1995	979	817
NCDC	GROVELAND	ESSEX	MA	10.1	< 25	1996	1704	1506
NCDC	GROVELAND	ESSEX	MA	10.1	< 25	1997	1036	1347
NCDC	GROVELAND	ESSEX	MA	10.1	< 25	1998		1439
NCDC	GROVELAND	ESSEX	MA	10.1	< 25	1999	1058	
NCDC	GROVELAND	ESSEX	MA	10.1	< 25	2000	1355	1227
NCDC	GROVELAND	ESSEX	MA	10.1	< 25	2002	1224	956
NCDC	GROVELAND	ESSEX	MA	10.1	< 25	2004	1310	1351
NCDC	GROVELAND	ESSEX	MA	10.1	< 25	2005		1197
NCDC	GROVELAND	ESSEX	MA	10.1	< 25	2006	1622	
NCDC	GROVELAND	ESSEX	MA	10.1	< 25	2007	1162	1200
NCDC	GROVELAND	ESSEX	MA	10.1	< 25	2008	1551	1520
NCDC	GROVELAND	ESSEX	MA	10.1	< 25	2009	1449	1423
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1980	832	799
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1981	1080	980
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1982	1163	1266
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1983	1541	1231
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1984	1280	1571
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1985	1011	989
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1986	1271	1152

Appendix A. Continued

SOURCE	STATION NAME	COUNTY	STATE	ELEV. (m)	DIST. (km)	YEAR	CY PPT. (mm)	WY PPT. (mm)
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1987	1137	1348
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1988	1126	1027
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1989	1134	1131
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1990	1316	1161
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1991	1257	1456
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1992	1096	1030
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1993	1186	1153
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1994	1235	1305
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1995	1012	875
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1996	1637	1562
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1997	982	1195
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1998	1401	1441
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	1999	1030	1078
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	2000	1366	1227
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	2001	1022	1216
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	2002	1232	978
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	2003	1239	1316
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	2004	1325	1360
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	2006	1615	1890
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	2008	1536	1441
NCDC	HAVERHILL	ESSEX	MA	6.1	< 25	2009	1430	1403
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	1980	831	841
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	1981	1092	994
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	1982	1135	1243
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	1985	979	937
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	1986	1214	1148
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	1989	876	980
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	1991	1142	1285
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	1992	963	937
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	1994	1313	1311
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	1996	1539	1433
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	1997	908	1149
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	1998	1203	1257
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	1999	1090	1098
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	2000	1154	1130
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	2001	886	975
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	2002	1074	894
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	2003	980	1045
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	2005	1134	810
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	2006	1506	1700

Appendix A. Continued

SOURCE	STATION NAME	COUNTY	STATE	ELEV. (m)	DIST. (km)	YEAR	CY PPT. (mm)	WY PPT. (mm)
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	2008	1307	1277
NCDC	LAWRENCE	ESSEX	MA	15.2	< 25	2009	1279	1279
NCDC	CAPE NEDDICK	YORK	ME	39.6	< 25	2002	1181	
NCDC	CAPE NEDDICK	YORK	ME	39.6	< 25	2003	1149	1190
NCDC	CAPE NEDDICK	YORK	ME	39.6	< 25	2004	1146	1189
NCDC	CAPE NEDDICK	YORK	ME	39.6	< 25	2005	1609	1264
NCDC	CAPE NEDDICK	YORK	ME	39.6	< 25	2006	1848	2014
NCDC	CAPE NEDDICK	YORK	ME	39.6	< 25	2007	1285	1428
NCDC	CAPE NEDDICK	YORK	ME	39.6	< 25	2008	1657	1617
NCDC	CAPE NEDDICK	YORK	ME	39.6	< 25	2009	1591	1470
NCDC	HOLLIS	YORK	ME	96	< 25	1995	1186	964
NCDC	HOLLIS	YORK	ME	96	< 25	1996	1512	1438
NCDC	HOLLIS	YORK	ME	96	< 25	1997	1235	1507
NCDC	HOLLIS	YORK	ME	96	< 25	1998	1501	1468
NCDC	HOLLIS	YORK	ME	96	< 25	1999	1275	1352
NCDC	HOLLIS	YORK	ME	96	< 25	2000	1337	1245
NCDC	HOLLIS	YORK	ME	96	< 25	2001	939	1116
NCDC	HOLLIS	YORK	ME	96	< 25	2002	1260	1050
NCDC	HOLLIS	YORK	ME	96	< 25	2003	1190	1145
NCDC	HOLLIS	YORK	ME	96	< 25	2005	1904	1428
NCDC	HOLLIS	YORK	ME	96	< 25	2006	1679	1941
NCDC	HOLLIS	YORK	ME	96	< 25	2007	1389	1469
NCDC	HOLLIS	YORK	ME	96	< 25	2008	1819	1864
NCDC	HOLLIS	YORK	ME	96	< 25	2009	1506	1422
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	1990	1354	1132
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	1991	1432	1676
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	1992	946	931
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	1993	1094	1043
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	1994	1145	1214
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	1995	1243	1035
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	1996	1544	1418
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	1997	1151	1503
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	1998	1389	1278
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	1999	1031	1167
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	2000	1297	1203
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	2001	983	1139
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	2002	1247	1069
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	2003	1206	1218
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	2004	1221	1243
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	2005	1718	1367

Appendix A. Continued

SOURCE	STATION NAME	COUNTY	STATE	ELEV. (m)	DIST. (km)	YEAR	CY PPT. (mm)	WY PPT. (mm)
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	2006	1679	1885
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	2007	1421	1436
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	2008	1651	1755
NCDC	KENNEBUNKPORT	YORK	ME	6.1	< 25	2009	1559	1468
NCDC	SACO	YORK	ME	24.1	< 25	1980	905	955
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	1980	884	972
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	1981	1215	1113
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	1983	1791	1358
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	1984	1318	1645
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	1985	1052	1034
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	1986	1191	1134
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	1988	1262	1157
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	1989	1212	1241
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	1990	1276	1114
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	1991	1342	1538
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	1992	1051	1065
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	1993	1164	1062
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	1994	1266	1359
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	1995	1232	1046
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	1996	1375	1431
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	1997	1299	1402
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	1998	1392	1435
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	1999	1318	1340
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	2000	1333	1241
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	2001	995	1165
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	2002	1160	936
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	2003	1146	1140
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	2004	1199	1336
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	2005	1829	1406
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	2006	1665	1877
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	2008	1724	1733
NCDC	SANFORD 2 NNW	YORK	ME	85.3	< 25	2009	1480	1394
NCDC	WATERBORO	YORK	ME	148	< 25	1992	954	1027
NCDC	WATERBORO	YORK	ME	148	< 25	1993	1000	940
NCDC	WATERBORO	YORK	ME	148	< 25	1994	1071	1123
NCDC	WATERBORO	YORK	ME	148	< 25	1995	1156	929
NCDC	WATERBORO	YORK	ME	148	< 25	1996	1412	1377
NCDC	WATERBORO	YORK	ME	148	< 25	1997	1080	1337
NCDC	WATERBORO	YORK	ME	148	< 25	1998	1383	1393
NCDC	WATERBORO	YORK	ME	148	< 25	1999	1213	1186

Appendix A. Continued

SOURCE	STATION NAME	COUNTY	STATE	ELEV. (m)	DIST. (km)	YEAR	CY PPT. (mm)	WY PPT. (mm)
NCDC	ALTON	BELKNAP	NH	219	< 25	1980	944	994
NCDC	ALTON	BELKNAP	NH	219	< 25	1981	1414	1220
NCDC	ALTON	BELKNAP	NH	219	< 25	1982	1095	1367
NCDC	ALTON	BELKNAP	NH	219	< 25	1984	1251	1488
NCDC	ALTON	BELKNAP	NH	219	< 25	1985	915	931
NCDC	ALTON	BELKNAP	NH	219	< 25	1986	1140	1126
NCDC	BARNSTEAD 5N	BELKNAP	NH	212	< 25	2005	1661	1228
NCDC	BARNSTEAD 5N	BELKNAP	NH	212	< 25	2006	1454	1742
NCDC	BARNSTEAD 5N	BELKNAP	NH	212	< 25	2007	1265	1323
NCDC	BARNSTEAD 5N	BELKNAP	NH	212	< 25	2008	1548	1554
NCDC	BARNSTEAD 5N	BELKNAP	NH	212	< 25	2009	1300	1318
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1980	687	741
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1981	1164	970
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1982	882	1107
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1983	1221	951
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1984	1073	1230
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1985	785	783
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1986	1023	1002
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1987	935	998
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1988	846	867
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1989	971	926
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1990	1049	967
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1991	1003	1016
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1992	756	848
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1993	803	740
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1994	918	992
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1995	976	796
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1996	1202	1173
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1997	872	1050
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1998	913	972
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	1999	1024	1030
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	2000	959	902
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	2001	717	828
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	2002	1014	821
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	2003	1141	1146
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	2004	1071	1143
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	2005	1453	1066
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	2006	1403	1629
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	2007	1124	1199
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	2008	1472	1492

Appendix A. Continued

SOURCE	STATION NAME	COUNTY	STATE	ELEV. (m)	DIST. (km)	YEAR	CY PPT. (mm)	WY PPT. (mm)
NCDC	CONCORD MUNI AP	MERRIMACK	NH	105	< 25	2009	1199	1175
NCDC	HUDSON 1 SSE	HILLSBOROUGH	NH	56.4	< 25	2008	1589	1557
NCDC	HUDSON 1 SSE	HILLSBOROUGH	NH	56.4	< 25	2009	1195	1199
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1980	723	802
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1981	1077	940
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1982	818	980
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1983	1315	1033
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1984	1205	1395
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1985	768	760
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1986	1064	1057
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1987	997	1024
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1988	951	959
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1989	1037	998
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1990	1172	1086
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1991	1059	1117
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1992	843	924
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1993	1080	990
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1994	1076	1147
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1995	1004	794
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1996	1532	1482
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1997	1012	1256
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1998	1157	1186
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	1999	1117	1129
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	2000	1178	1094
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	2001	834	974
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	2002	962	833
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	2003	1138	1069
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	2004	966	1060
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	2006	1463	1748
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	2007	1097	1125
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	2008	1561	1589
NCDC	LAKEPORT 2	BELKNAP	NH	152	< 25	2009	1328	1260
NCDC	MANCHESTER	HILLSBOROUGH	NH	64	< 25	1997	957	
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	1980	727	848
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	1981	1078	937
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	1983		872
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	1985	841	
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	1986	1000	952
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	1987	924	996
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	1988	1053	1040

Appendix A. Continued

SOURCE	STATION NAME	COUNTY	STATE	ELEV. (m)	DIST. (km)	YEAR	CY PPT. (mm)	WY PPT. (mm)
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	1989	1026	1037
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	1990	1336	1179
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	1992	878	905
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	1993	930	865
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	1996	1256	1186
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	1997	771	979
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	1998	951	1027
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	1999	923	879
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	2000	1013	977
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	2001	745	908
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	2002	967	754
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	2003		1082
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	2005	1353	1013
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	2006	1590	1793
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	2008	1562	1589
NCDC	MASSABESIC LAKE	HILLSBOROUGH	NH	77.1	< 25	2009	1156	1195
NCDC	MERRIMACK	HILLSBOROUGH	NH	73.2	< 25	1992	1146	1141
NCDC	MERRIMACK	HILLSBOROUGH	NH	73.2	< 25	1993	1193	1123
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	1980	953	1001
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	1981	1195	1064
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	1982	1081	1236
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	1983	1461	1223
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	1984	1243	1407
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	1985	998	991
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	1986	1154	1100
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	1987	1053	1135
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	1988	1029	1035
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	1989	1102	1099
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	1990	1303	1175
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	1991	1271	1369
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	1992	957	969
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	1993	1082	1014
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	1994	1089	1192
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	1995	1032	852
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	1999	1098	1066
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	2004	1154	1229
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	2005	1517	1201
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	2007	1341	1352
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	2008	1978	1980
NCDC	NASHUA 2 NNW	HILLSBOROUGH	NH	42.7	< 25	2009	1304	1352

Appendix A. Continued

SOURCE	STATION NAME	COUNTY	STATE	ELEV. (m)	DIST. (km)	YEAR	CY PPT. (mm)	WY PPT. (mm)
NCDC	NEW DURHAM 4 N	STRAFFORD	NH	201	< 25	1994	981	1078
NCDC	NEW DURHAM 4 N	STRAFFORD	NH	201	< 25	1995	1046	828
NCDC	NEW DURHAM 4 N	STRAFFORD	NH	201	< 25	1996	1461	1369
NCDC	NEW DURHAM 4 N	STRAFFORD	NH	201	< 25	1997	1074	1344
NCDC	NEW DURHAM 4 N	STRAFFORD	NH	201	< 25	1998	1146	1186
NCDC	NEW DURHAM 4 N	STRAFFORD	NH	201	< 25	1999	1046	1041
NCDC	NEW DURHAM 4 N	STRAFFORD	NH	201	< 25	2000	1168	1093
NCDC	NEW DURHAM 4 N	STRAFFORD	NH	201	< 25	2001	814	978
NCDC	NEW DURHAM 4 N	STRAFFORD	NH	201	< 25	2002	1082	908
NCDC	NEW DURHAM 4 N	STRAFFORD	NH	201	< 25	2003	1198	1152
NCDC	NORTH HAMPTON	ROCKINGHAM	NH	18.3	< 25	2004	1303	
NCDC	NORTH HAMPTON	ROCKINGHAM	NH	18.3	< 25	2005	1531	1248
NCDC	NORTH HAMPTON	ROCKINGHAM	NH	18.3	< 25	2006	1877	2044
NCDC	NORTH HAMPTON	ROCKINGHAM	NH	18.3	< 25	2007	1256	1303
NCDC	NORTH HAMPTON	ROCKINGHAM	NH	18.3	< 25	2008	1650	1663
NCDC	NORTH HAMPTON	ROCKINGHAM	NH	18.3	< 25	2009	1502	1423
NCDC	SALISBURY	MERRIMACK	NH	168	< 25	1997	1112	1331
NCDC	SALISBURY	MERRIMACK	NH	168	< 25	1998	1196	1249
NCDC	SALISBURY	MERRIMACK	NH	168	< 25	1999	1239	1274
NCDC	SALISBURY	MERRIMACK	NH	168	< 25	2000	1250	1178
NCDC	SALISBURY	MERRIMACK	NH	168	< 25	2001	923	1050
NCDC	SALISBURY	MERRIMACK	NH	168	< 25	2002	1126	951
NCDC	SALISBURY	MERRIMACK	NH	168	< 25	2003	1357	1285
NCDC	SALISBURY	MERRIMACK	NH	168	< 25	2004	1185	1322
NCDC	SALISBURY	MERRIMACK	NH	168	< 25	2006	1733	2001