Tracing anthropogenic nutrient inputs into coastal plain ponds on an urban-rural gradient: a study using stable isotopes

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Abstract
Coastal plain ponds in Cape Cod are unique ecosystems that are hotspots for rare plant species. In recent years, increasing urbanization and development surrounding these ponds have increased nutrient loading via groundwater. This can damage rare plant species and cause eutrophication. This study examined six different coastal plain ponds in Cape Cod along an urban-rural gradient to determine changes in DIN concentrations in pond water and soils, $\delta^{15}N$ of shoreline vegetation, and $\delta^{15}N$ of lake sediments as a response to urbanization. These measurements were analyzed with DIN concentrations in incoming groundwater and the $\delta^{15}N$ of incoming groundwater.

Key Words
dissolved inorganic nitrogen (DIN), nutrient loading, coastal plain ponds, N-stable isotope

Introduction
Coastal plain ponds in New England are unique ecosystems that are hotspots for rare plant and animal species (Sorrie, 1994). These ponds were formed during the last glaciation roughly 12,000 years ago during which large pieces of ice left depressions on the surface of the earth and eventually melted to create small isolated bodies of fresh water. The majority of these New England coastal ponds are concentrated in southeastern Massachusetts on Cape Cod and in southern Rhode Island (Sorrie, 1994).

Coastal plain ponds are only recharged by groundwater, and water levels rise and fall seasonally (Sorrie, 1994). These fluctuations result in a frequently shifting shoreline, which leads to concentric bands of plant species that germinate at different levels of the shoreline. This unique condition of coastal plain ponds has allowed the growth of rare plant species native to Cape Cod (Neill et. al, 2009). Such include the Plymouth gentian (*Sabatia kennedyana*), tread-leaved sundew (*Drosera filiformis*), Maryland meadow beauty (*Rhexia mariana*), as well as various other species listed on the Natural Heritage and Endangered Species list (Neill et. al, 2009).

In recent years, increasing levels of urbanization and human disturbance near coastal plain ponds has damaged the water quality of these ponds as well as the long-term persistence of rare plant species in the area (Neill et. al, 2009). Human disturbance factors include the use of off-road vehicles near the ponds, drawing down of water levels due to excessive groundwater pumping, and the input of excess nutrients from nearby developed communities (Sorrie, 1994). This study will focus specifically on the disturbance factor of excess nitrogen (N) inputs via groundwater into coastal plain ponds in Cape Cod.

The influx of excess N from fertilizers and human wastewater upstream from the pond can damage native plant species while encouraging the growth of invasive species (Sorrie, 1994). This is because native plant species are accustomed to living atop nutrient poor soils derived from glacial till based bedrock. Thus, the excess input of nutrients will not benefit these native species, but will encourage the growth of invasive ones such as *Phragmites australis*, which thrives in high nutrient environments (Minchinton and Bertness, 2003).

This study examined the degree and impact of anthropogenically derived N inputs into six coastal plain ponds along an urban-rural gradient in Cape Cod, MA. Three of these ponds were located in the town of Barnstable, MA and three were located in Falmouth, MA . These ponds were chosen based on rough estimated levels of development surrounding them. Increased development was expected to correlate with increased loading of anthropogenic N into the ponds. This anthropogenic N is derived from human wastewater which is carried to the ponds via groundwater flow.

I measured the following from six ponds:

- DIN concentration of surface water of ponds
- DIN in shoreline soils
- $\delta^{15}N$ of plant biota
- $\delta^{15}N$ down sediment cores of ponds
- Lead (Pb) down sediment cores of ponds

The DIN concentrations of the surface water and shoreline soils was measured to determine the magnitude of N-loading and retention in the six coastal plain ponds. The DIN concentration in shoreline soil could specifically influence the type of vegetation that would most optimally grows in that area. The $\delta^{15}N$ of shoreline vegetation was measured to determine type of N used by plants for growth. N derived from human and animal wastewater has a higher $\delta^{15}N$ compared to N derived from natural soils due to the volatilization of ammonia and denitrification that occurs while it is carried by groundwater (McClelland and Valiela 1998); these processes remove the isotopically lighter ^{14}N from NO_3^-, which causes the NO_3^- to become isotopically heavier when compared to NO_3^- from natural soil samples (McClelland and Valiela 1998). Because the $\delta^{15}N$ of primary producers depends on the $\delta^{15}N$ of its N nutrient source (McClelland et al 1998), $\delta^{15}N$ analysis of shoreline vegetation can reveal the primary source of N used by plants (anthropogenic wastewater v. NO_3^- in natural soils). The $\delta^{15}N$ was measured down a sediment core to reveals changes in anthropogenic N-input over time. Lastly, Pb concentrations down a core were measured to place a rough time markers on the core which I used to correlate $\delta^{15}N$ values with.

While this project aimed to determine the measurable impacts of N-loading into ponds by examining water, soil, plants, and sediments in situ, my collaborator Ian Yue determined the actual concentration of DIN inputs into the six ponds by determining DIN concentrations in groundwater. Ian also determined the $\delta^{15}N$ of this groundwater. I used these two forms of data and compared my findings with them. This allowed me to assess what correlations exist between degree of urbanization, magnitude of anthropogenic N-loading into each pond, DIN concentrations in soils and water, and $\delta^{15}N$ of shoreline vegetation.

Methods

Study Sites

The six sites that Ian and I studied were Hathaway Pond, Mary Dunn Pond, Crooked Pond, Shubael Pond, Jenkins Pond, and Weeks Pond. We chose sites based on the level of estimated development up the gradient from which water moves into the pond. This was done by a rough estimation of housing density. We assigned three categories for the ponds we we studied: low degree of impact, medium, and high; we chose two ponds from each category.

- Low contamination (rural): Hathaway (B), Mary Dunn (B)
- Medium contamination: Crooked (F), Shubael (B)
- High contamination (urban): Jenkins (F), Weeks (F)

(B) = located in Barnstable
(F) = located in Falmouth

Field Collections
From the six ponds I studied, I collected the following: pond water, shoreline soil, shoreline vegetation, and pond sediment cores.

Pond water was collected by hand into 300mL bottles and stored on ice until it was delivered to the laboratory where it was refrigerated.

Shoreline soil was collected using a metal trowel or a shovel. The soil was stored in a "zip-lock" bag on ice and until it was delivered to the laboratory where it was refrigerated.

Shoreline plants were collected by hand or using clippers. I collected a total of eight different types of plants at the six ponds. They included: Myriophyllum aquaticum, Utricularia, Eriocaulon, Juncus militaris, Eleocharis acicularis, Gratiola aurea, Vallisneria, and an unidentified aquatic moss. Although not every plant was uniformly found at all six ponds, there were several specie overlaps in the six ponds so that relative comparisons of $\delta^{15}N$ of plant biomass were possible (Table 1).

I collected pond sediment cores at all six sites using a gravity corer. The sediment cores ranged in height from 5.5 cm to 30 cm. The sediment cores were capped on deck and stored vertically until I reached the laboratory. In the laboratory, overlying water was removed with care as to not disturb the sediment-water interface. The sediment was then vertically extruded using a rubber O-ring stopper and was sliced at 1cm increments. These sediments were homogenized and dried for $\delta^{15}N$ and Pb analysis.

Laboratory Analysis
I tested the pond water and shoreline soil for nitrate (NO_3^-) and ammonium (NH_4^+). To determine the NO_3^- concentrations from pond water, I used the procedure adapted from Wood et. al (1967) along with a Lachat flow injection analyzer. To analyze ammonium concentrations in the pond water, I used a protocol modified from Solarzano (1969) and Strickland and Parsons (1972). To analyze NO_3^- and NH_4^+ concentrations in shoreline soil, I used KCl to extract these ions and will analyze the resulting solution in the same method used to analyze pond water. I followed the extraction protocol used in the Semester in Environmental Science (SES) program in the Week 1 and Week 6 terrestrial laboratory exercises.

I analyzed the $\delta^{15}N$ of the plant biomass and the sediment cores using an isotope ratio mass spectrometer. All samples were dried, weighed, and loaded into tin capsules and were be combusted in an element analyzer. The resulting gases were separated before being introduced to the mass spectrometer (McClelland et al, 1997). While I loaded the dry plant biomass and sediment core samples into the tin boats, an isotope analyst at the Marine Biological Laboratory's Stable Isotope Laboratory operated the mass spectrometer and element analyzer.

I measured the lead concentrations from sections within sediment cores by first extracting the lead into a solution of 10% HNO_3 solution as outlined by Forstner and Salomons (1980). I then used an atomic absorption spectrophotometer to determine lead concentrations in these solutions, which were then converted to Pb (mg/g dry weight sediment).

Results

NO_3^- was the largest contributor to the total DIN concentration of inflowing groundwater in Crooked Pond, Shubael Pond, Jenkins Pond, and Weeks Pond (Yue 2010; Fig. 1). In Hathaway Pond and Mary Dunn Pond, there was a larger concentration of NH_4^+ in groundwater inputs compared to NO_3^- (Yue 2010; Fig. 1).

The DIN concentrations in incoming groundwater were highest at Jenkins Pond and Weeks Pond at 196μM and 183μM respectively (Yue 2010; Fig. 2). These concentrations were more than five times of those found at Crooked Pond and Shubael Pond, and more than ten times of those found in Mary Dunn Pond and Hathaway Pond (Yue 2010; Fig. 2).

The $\delta^{15}N$ of NO_3^- from groundwater was highest at Jenkins Pond and Weeks Pond (Yue 2010; Fig. 3) at 27.7‰ and 14.9‰ respectively. At Crooked Pond and Shubael Pond, these values were 7.7‰ and 7.8‰ respectively (Yue 2010; Fig. 3). At Hathaway Pond and Mary Dunn Pond, these values were under 1‰ (Yue 2010; Fig. 3).

The DIN concentrations of pond water was highest at Weeks Pond with 57.03 μM NO_3^- and 35.32 μM of NH_4^+ (Fig. 4). The DIN concentrations at Jenkins Pond was surprisingly low at 6.12 μM NH_4^+ and 3.91 μM NO_3^- (Fig. 4). The NH_4^+ concentrations at Hathaway Pond and Mary Dunn Pond were 1.76 μM and 6.00 μM respectively, while those at Shubael Pond and Jenkins Pond were 5.15 μM and 6.12 μM respectively (Fig. 4).

DIN concentrations in shoreline soils were consistently low (under 1 μM) at all six sampled sites (Fig. 5). NH_4^+ was the primary form of inorganic nitrogen in all soils. Within the sites, Jenkins Pond had the highest NH_4^+ concentration followed by Mary Dunn and Hathaway Ponds (Fig. 5). Weeks Pond had the lowest concentration of DIN in its soil with 0.030 uM NH_4^+.

In the shoreline vegetation that was sampled, all five different species of vegetation from Mary Dunn Pond consistently had the lowest δ15N value when compared to those same species taken from all other sites (Fig. 6A, 6B, 6C, 6D, 6E).
When the $\delta^{15}N$ values of all vegetation found at each of the six ponds were averaged, the vegetation at Weeks Pond and Jenkins Pond had the highest average $\delta^{15}N$ —9.8‰ and 8.5‰ respectively—while those from Shubael Pond and Crooked Pond had average $\delta^{15}N$ values of 6.4‰ and 7.6‰ respectively (Fig. 7). Vegetation from Mary Dunn Pond and Hathaway Ponds had the lowest average $\delta^{15}N$ values—2.7‰ and 2.0‰ respectively (Fig.7).

The Pb profiles of cores taken from Jenkins Pond, Crooked Pond, Hathaway Pond, and Shubael Pond all showed that Pb concentrations began to increase around 5cm down the cores (Fig. 8). In Mary Dunn Pond, the lead concentration began to increase around 12cm down the core. In Weeks Pond, the lead began to increase around 9cm down the core. While the Pb concentrations remained below 0.50mg Pb/g dry sediment throughout the cores from Jenkins, Crooked, Hathaway, and Shubael ponds, in both Mary Dunn and Weeks Ponds, the surface Pb concentrations were found to be around 3.3 mg Pb/g dry sediment (Fig. 8).

The $\delta^{15}N$ values in the six cores varied. The cores from Jenkins Pond and Shubael Pond had the highest surface layer $\delta^{15}N$ values—7.5‰ and 7.0‰ respectively (Fig. 9). Hathaway Pond and Weeks Pond had lower surface $\delta^{15}N$ values—4.9‰, and 4.8‰ respectively (Fig. 9). Crooked Pond and Mary Dunn Pond had the lowest surface $\delta^{15}N$ values—both under 3‰. Both Crooked Pond and Mary Dunn Pond had $\delta^{15}N$ values under 3.5‰ throughout the whole cores (Fig. 9).

While there was a range of $\delta^{15}N$ values by depth and between sites, there was a general trend of increasing $\delta^{15}N$ values towards the surface of the surface of the cores. This trend was not exhibited in Mary Dunn Pond and Hathaway Pond. In Mary Dunn Pond, the $\delta^{15}N$ peaked between 10 and 20cm down the core at 3.2‰, but values decreased toward the surface layers. In Hathaway Pond, the $\delta^{15}N$ peaked around 4cm down the core at 5.9‰, and decreased at the surface of the core.

Discussion

There was a distinct correlation between the expected levels of development at the six ponds and the $\delta^{15}N$ of groundwater inputs in these ponds. The two estimated high-development and high-impact ponds—Jenkins Pond and Weeks Pond—both had groundwater NO_3^- with $\delta^{15}N$ values which were above 14‰ (Fig 3). Groundwater NO_3^- with $\delta^{15}N$ values between 10 to 20‰ is typically derived from human or animal waste (Aravena 1993; Macko and Ostrom 1994; McClelland et al.1997). Such values were expected near the two high-impact ponds, as groundwater in these sites moves directly past numerous septic systems before entering the ponds. In Jenkins Pond, the $\delta^{15}N$ of NO_3^- was extraordinarily high at 27.7‰. While there could have been an error during isotope analysis, the standards run during the process indicate that this is unlikely. It is also possible that there was contamination during the collection or preparation of this groundwater sample before isotope analysis. Discounting this possibility, this high value could simply be derived from groundwater that was directly exposed to a plume of septic waste from an individual home. At the two estimated medium-impact ponds—Crooked and Shubael ponds--the $\delta^{15}N$ of $NO_3^-$was within the 2‰-8‰ range, suggesting that the NO_3^- is mostly derived from natural soils that is not exposed to human waste (Macko and Ostrom, 1994). In the two estimated low-impact ponds—Hathaway and Mary Dunn ponds—the $\delta^{15}N$ of NO_3^- was very low—under 1‰. While these values are typically found in NO_3^- that is derived from synthetic fertilizers (McClelland et al. 1997), it is more likely that the values were so low because there was such a small concentration of NO_3^- found in the groundwater samples that were analyzed (Fig.1). Because a limited number of groundwater samples were taken from each pond site, it was difficult to attain a perfectly representative average $\delta^{15}N$ of NO_3^-, as there is variability in values within short distance down a shoreline. With a greater sample size, one could more accurately characterize this average isotope values with a smaller standard error.

The $\delta^{15}N$ of shoreline vegetation at all sites was strongly linked to the $\delta^{15}N$ of NO_3^- from groundwater inputs (Fig. 10). The $\delta^{15}N$ of vegetation was highest at Jenkins Pond and Weeks Pond where the average $\delta^{15}N$ of groundwater NO_3^- was highest as well (Fig. 7). While the $\delta^{15}N$ of groundwater NO_3^- was higher at Jenkins Pond than at Weeks Pond, the vegetation from

Jenkins Pond had a lower average $\delta^{15}N$, probably because this value was derived using the $\delta^{15}N$ average of only two plant species, while four different plant species were sampled and averaged at Weeks Pond (Fig. 6C, 6D, 6E, 6F, 6G, 6H). The $\delta^{15}N$ values of vegetation from Crooked and Shubael ponds were in the mid range of all ponds, while those from Hathaway and Mary Dunn Ponds were in the low range (Fig. 7)—both of which were expected. Mary Dunn Pond was confirmed to be a low-impact pond, as all of the vegetation sampled at this site had routinely lower $\delta^{15}N$ compared to those same vegetation sampled at all other sites (6A, 6B, 6D, 6E, 6F, Table 1)

It is interesting to note that there was a greater difference between the $\delta^{15}N$ of groundwater NO_3^- and $\delta^{15}N$ of vegetation in Jenkins Pond and Weeks Pond than there was in Crooked Pond or Shubael Pond (Fig. 3, Fig. 7) . This can be attributed to differences in DIN concentrations that is entering the ponds. The greater the available DIN for plant uptake, the greater the fractionation there is in plant biomass. This is because a greater amount of the lighter ^{14}N isotope is available when there is a large concentration of DIN, and plants will preferentially take up this lighter isotope (Pennock et al 1996; McClelland and Valiela 1998). Thus, in Jenkins Pond and Weeks Pond, the $\delta^{15}N$ of vegetation is much lower than their respective $\delta^{15}N$ of groundwater NO_3^- values, where as in Shubael Pond and Crooked Pond, the $\delta^{15}N$ of vegetation more closely reflects groundwater inputs.

I found a rough correlation between DIN concentrations of groundwater and the $\delta^{15}N$ values of shoreline vegetation (Fig. 11). This was expected since increased DIN inputs are generally attributed to human wastewater in these ponds, and vegetation biomass reflect $\delta^{15}N$ values similar to their source of N.

There was a less distinct correlation between the DIN concentrations found in incoming groundwater and the DIN in pond water and shoreline soils. The DIN concentrations of incoming groundwater were the greatest in Jenkins Pond followed by Weeks Pond (Fig. 2). While the pond water concentration was very high in Weeks Pond, it was unexpectedly low in Jenkins Pond (Fig. 4). The NH_4^+ concentration in Jenkins Pond was in fact lower than that in Crooked Pond and Mary Dunn Pond. This suggests that DIN pools in pond water are not perfectly reflective of incoming DIN concentrations. This is likely due to the uptake of DIN by primary producers within the water. Similarly, the DIN concentrations in shoreline soils were not very reflective of groundwater DIN inputs. The DIN concentrations in soils were very low overall (Fig. 5) , and there was little consistency with expected values determined by DIN concentrations from groundwater inputs. This too, may be attributed to the uptake of DIN by primary producers that are rooted in the soils. It may also be attributed to denitrification occurring within the soils, which converts NO_3^- to N_2 gas. Overall, I found that pools of DIN generally do not build up in soils and pond water in a way that reflects the magnitude of their inputs.

The Pb profiles down sediment cores revealed the general trend of increasing concentrations near the surface layers, which was expected (Fig 8). A previous study by Charles and Norton (1986) estimated that visible increases in Pb concentrations in sediment cores correlate roughly with the years 1850-1900, and that Pb accumulations increase dramatically after 1900. Various subsequent studies have used the year1900 as a rough time marker in sediments (Giblin et al. 1990; Charles et al, 1987). While most cores followed an expected trend if increasing Pb levels around 5cm, which peaked at under 3mg/g dry sediment, the trend from Mary Dunn Pond and Jenkins Pond were unexpected. While the high Pb concentrations from Jenkins Pond could be attributed to its proximity to high levels of development, those values from Mary Dunn Pond, a low-impact pond, was not fully understood. One possible explanation is that hunting activity near Mary Dunn Pond caused deposition of lead shot in the pond over the last 50-100 years, creating high concentrations of lead within the top 5cm of the sediments.

Using the assumption that the depths where Pb peaks in sediment cores roughly correlates with 1900, I found that Crooked Pond, Shubael Pond, Jenkins Pond, and Weeks Pond all experienced increases in anthropogenically derived N-loading before the 1900 (Fig. 9). Within these four sites, it was expected for Jenkins to have the highest surface $\delta^{15}N$ value. However, it was not expected for Shubael Pond to have such high $\delta^{15}N$ surface values—greater than those from Weeks Pond. Similarly, the $\delta^{15}N$ value at the surface sediment of Mary Dunn Pond was surprisingly high. These discrepancies were not fully understood. Within Hathaway Pond and Mary Dunn Pond, the $\delta^{15}N$ values decreased toward the surface layers of sediment, which was another trend that was not expected. While there was variability in the $\delta^{15}N$ values down several sediment cores, I did find the general trend of increasing $\delta^{15}N$ values toward the top layers of most cores. Further, Jenkins Pond was shown to have the highest surface $\delta^{15}N$ values amongst all sampled ponds, which was also an expected trend, considering that is a high impact urban pond.

Conclusion

I found that there was a distinct correlation between estimated levels of development near the six sampled coastal plain pond and the $\delta^{15}N$ of groundwater inputs into these ponds. There was also a definite link between the $\delta^{15}N$ of shoreline vegetation and the $\delta^{15}N$ of groundwater inputs. While there were distinct couplings in such isotope values within these ecosystems, there were less visible correlations in the DIN concentrations of groundwater inputs and DIN in soils and vegetation.

The $\delta^{15}N$ down various sediment cores showed general trends of increasing $\delta^{15}N$ values towards the surface of the cores, although there were discrepancies in two sampled cores.

Overall, the strongest demonstrated trend was that between the $\delta^{15}N$ of NO_3^- of groundwater and $\delta^{15}N$ of shoreline vegetation. The distinct coupling suggests that plant species are significantly and directly impacted by anthropogenic N-inputs into the sampled ponds; this has important implications for the management of these ponds, as the rare native species will undoubtedly be affected by greater anthropogenic N-inputs in the future. This finding is also significant in that it demonstrates shoreline plants' ability to act as a gauge for the type of N entering the pond; the shoreline plants were the measure that most effectively showed anthropogenic N-inputs. While pond sediments also recorded elevated $\delta^{15}N$ inputs, due to slow sedimentation rates, these records were not sensitive nor accurate enough to determine recent changes in N-loading. While elevated DIN concentrations in systems can also be telling about anthropogenic N inputs, this was not found to be true in many of the sampled ponds where DIN concentrations in pond water and soil were variable and not perfectly correlated with the concentration of DIN inputs. Thus, the plant biota was found to be the most effective gauge of changes in N-loading into the six sampled coastal plain ponds.

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Figures

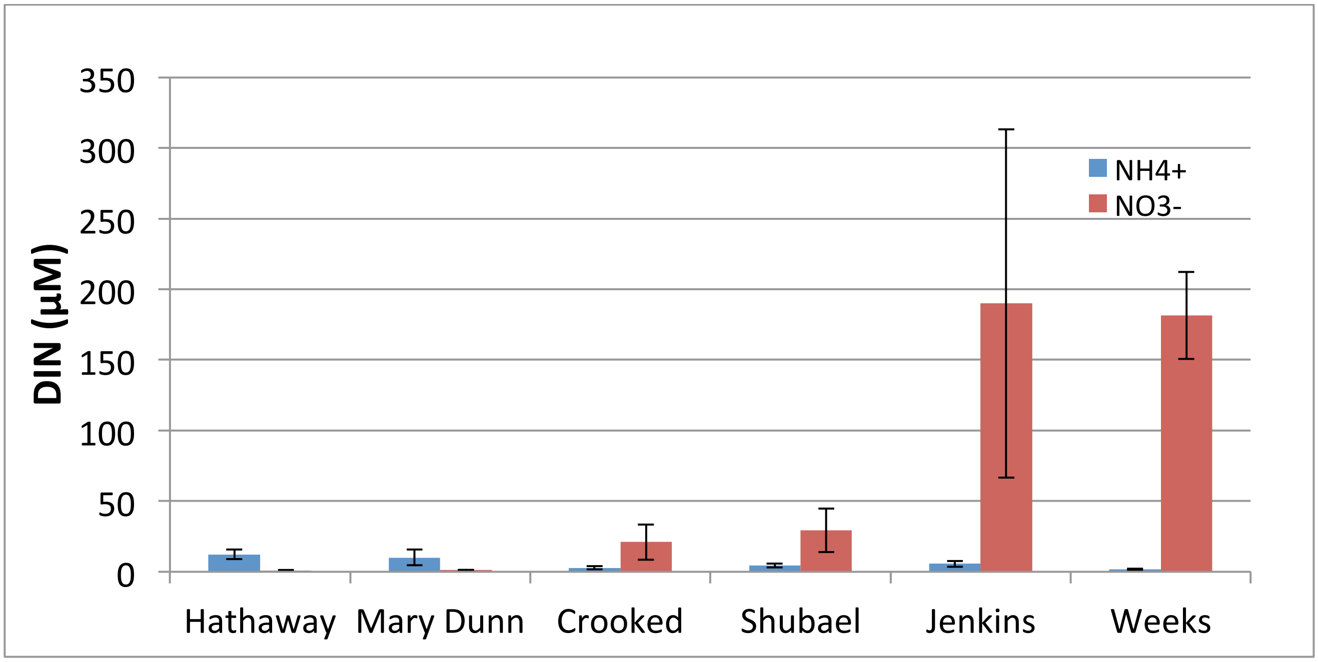

Figure 1. NH_4^+ and NO_3^- (µM) in groundwater at six sampled ponds.

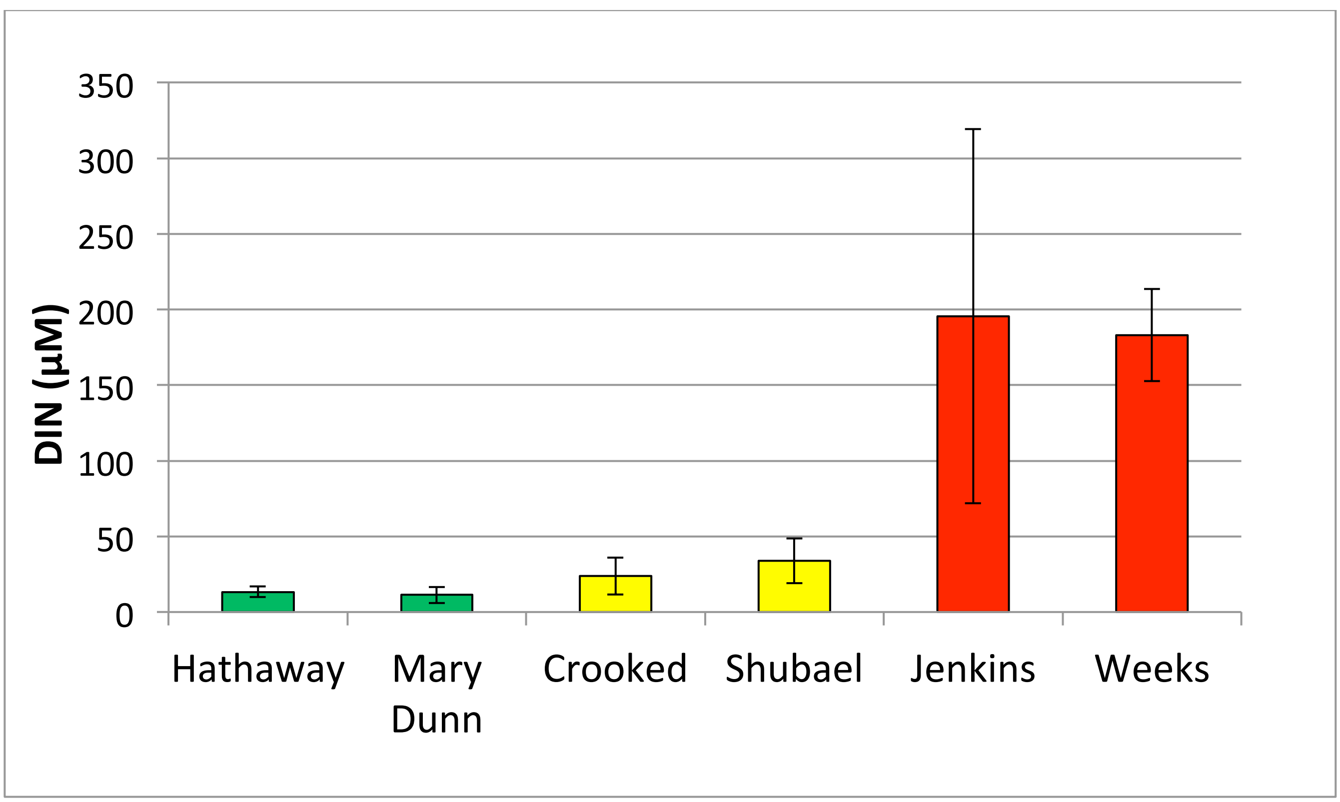

Figure 2. Average DIN (µM) in groundwater of six sampled ponds.

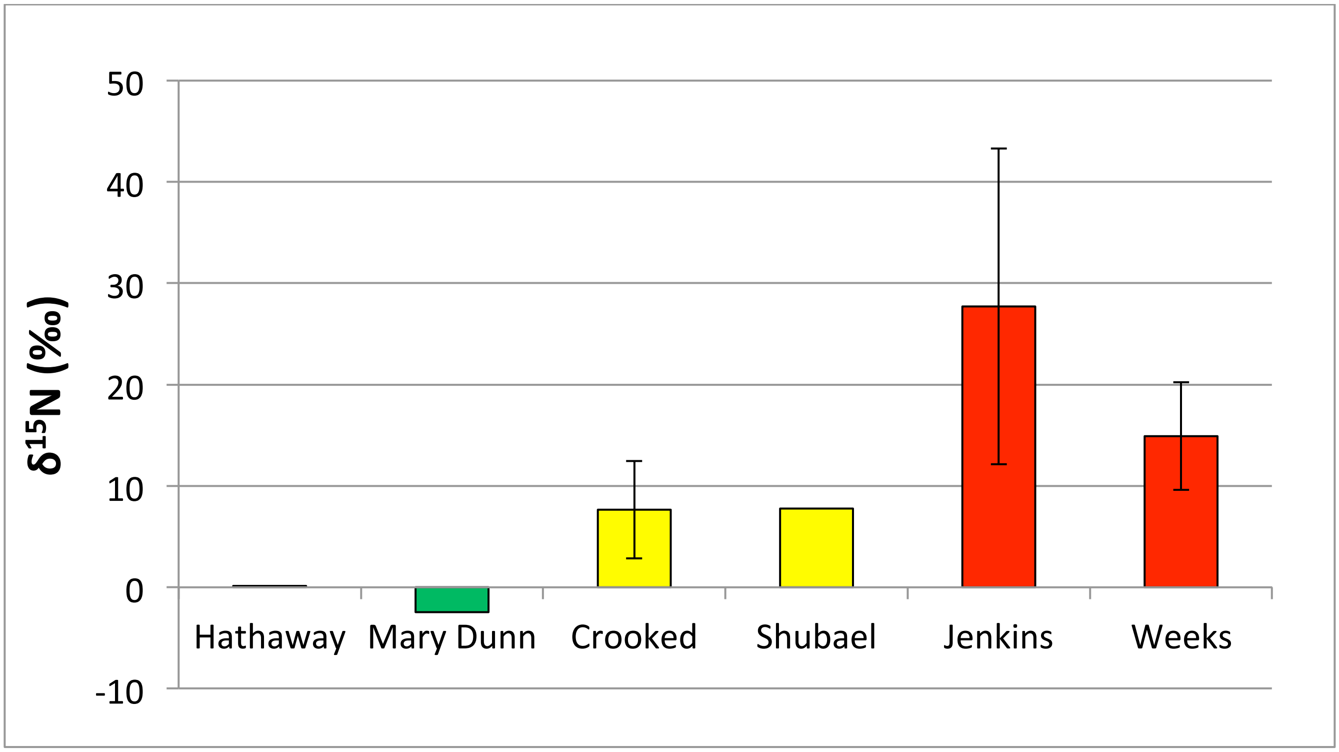

Figure 3. $\delta^{15}N$ of NO_3^- from Groundwater (‰) of six sampled ponds.

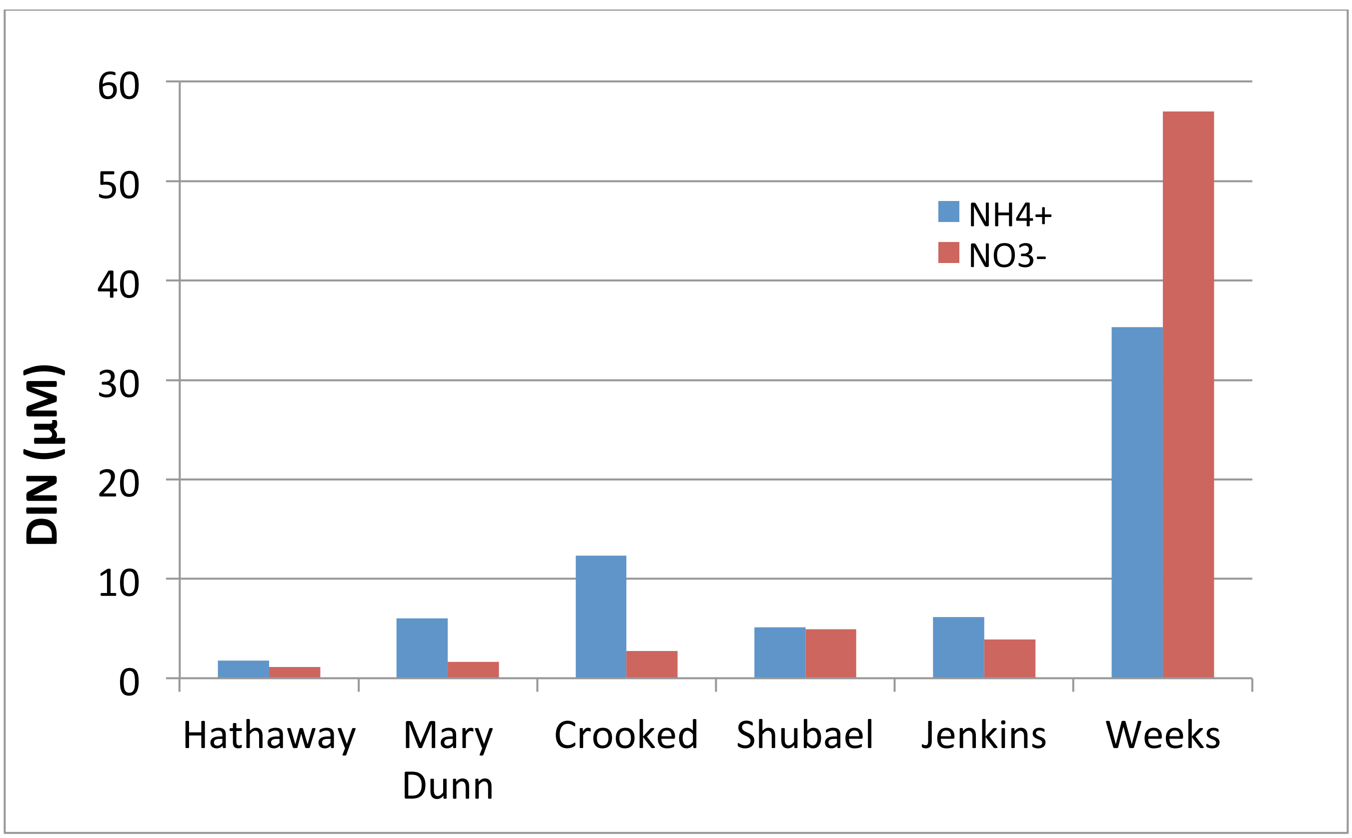

Figure 4. NH_4^+ and NO_3^- (µM) in Pond Water of six sampled ponds

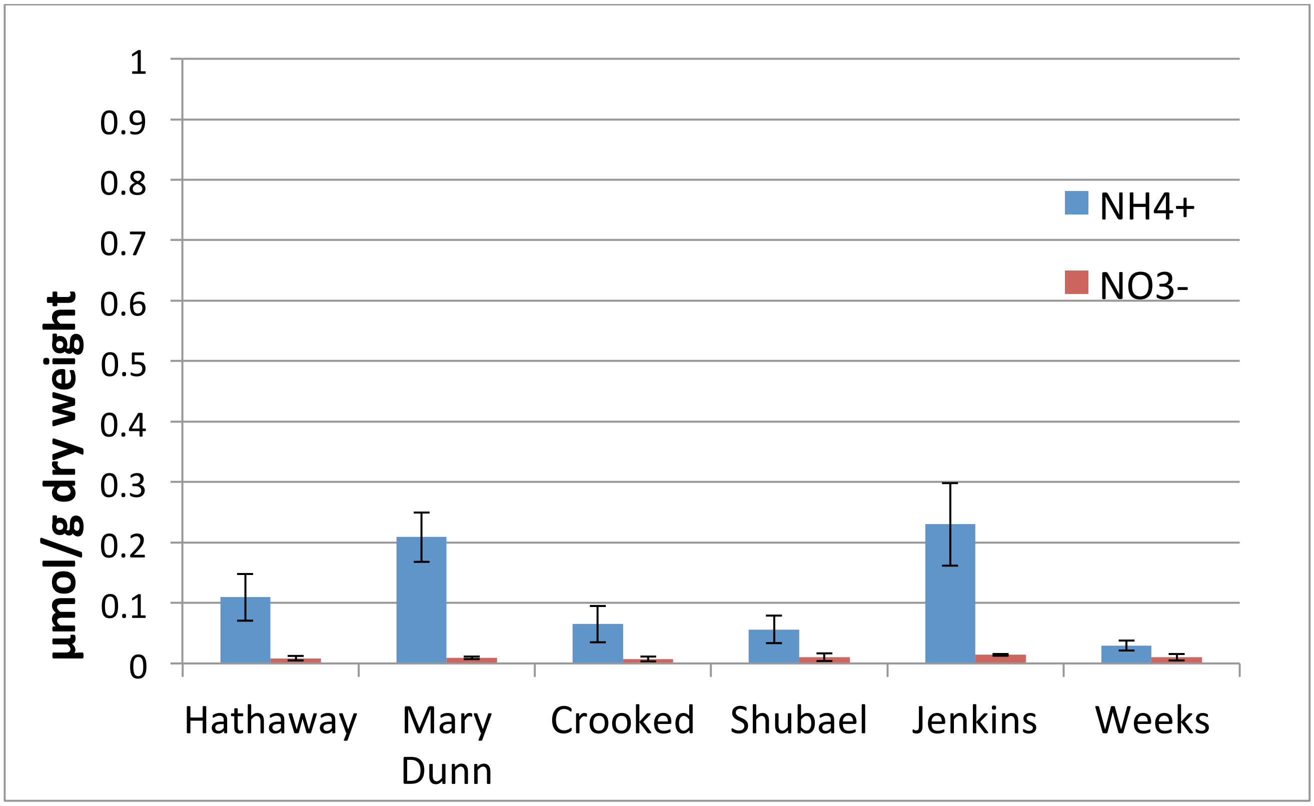

Figure 5. NH_4^+ and NO_3^- in Shoreline Soils at six sampled sites

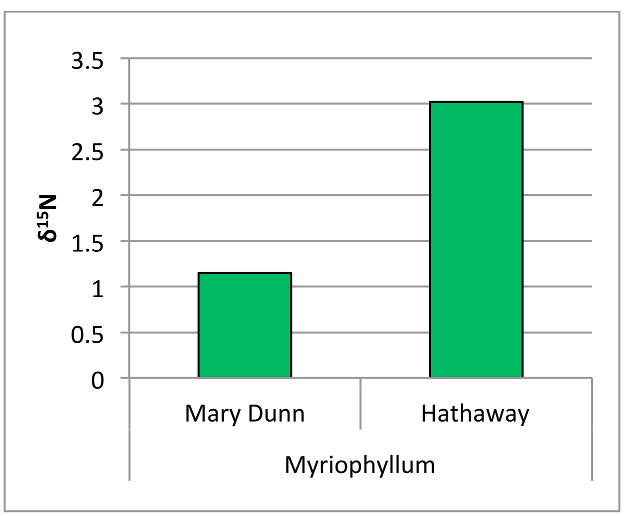

Figure 6A. $\delta^{15}N$ of Myriophyllum aquaticum sampled at Mary Dunn Pond and Hathaway Pond

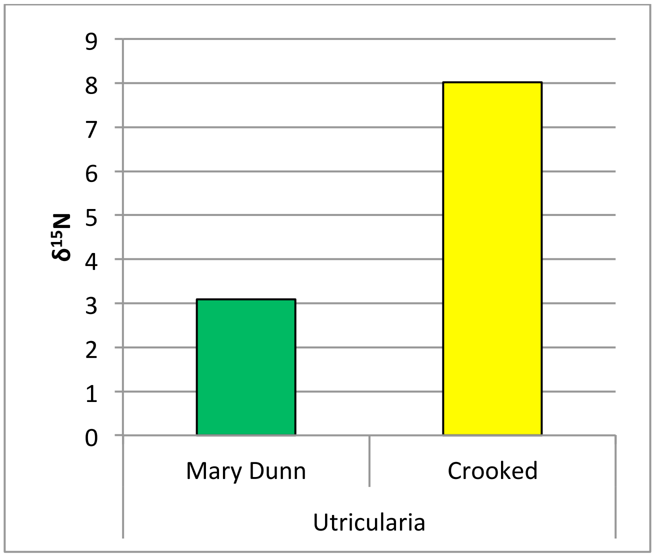

Figure 6B. $\delta^{15}N$ of Utricularia sampled at Mary Dunn Pond and Crooked Pond

Figure 6C. $\delta^{15}N$ of Eriocaulon sampled at Crooked Pond, Shubael Pond, and Jenkins Pond

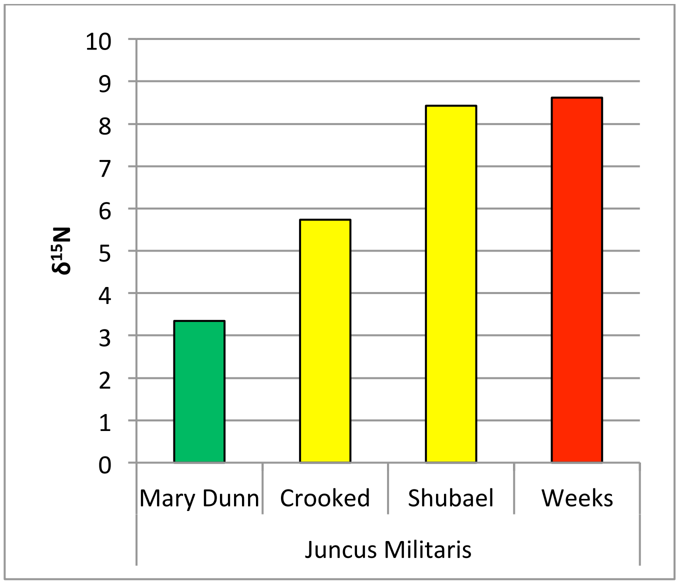

Figure 6D. $\delta^{15}N$ of Juncus militaris sampled at Mary Dunn Pond, Crooked Pond, Shubael Pond, and Weeks Pond

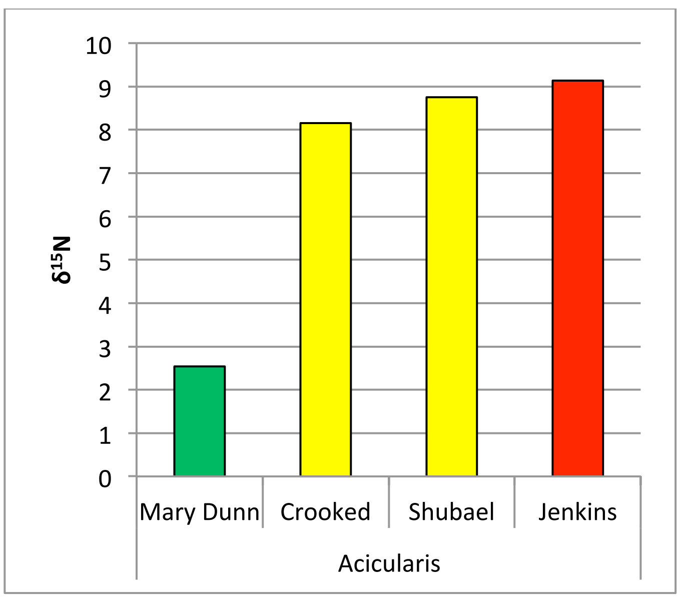

Figure 6E. $\delta^{15}N$ of Eleocharis acicularis sampled at Mary Dunn Pond, Crooked Pond, Shubael Pond, and Jenkins Pond

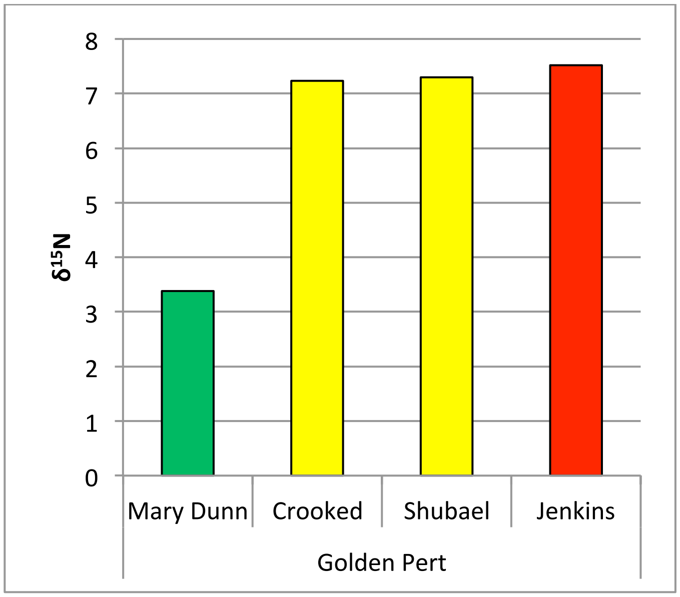

Figure 6F. $\delta^{15}N$ of Gratiola aurea sampled at Mary Dunn Pond, Crooked Pond, Shubael Pond, and Jenkins Pond

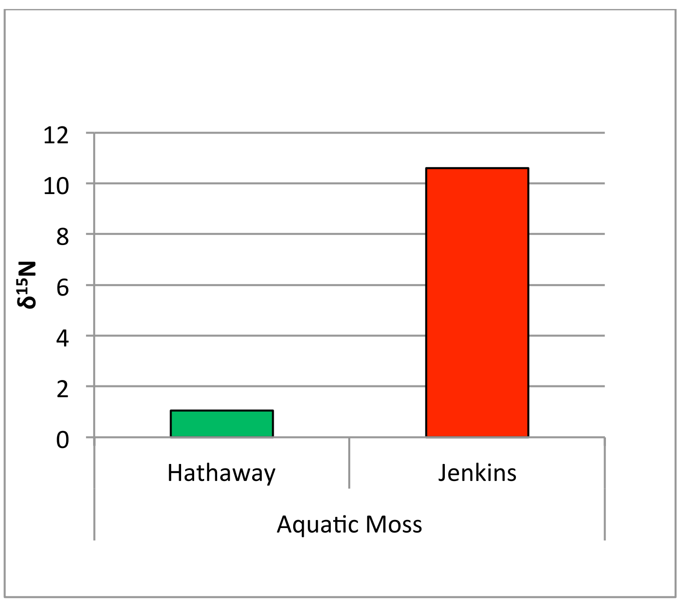

Figure 6G. $\delta^{15}N$ of aquatic moss sampled at Hathaway Pond and Jenkins Pond

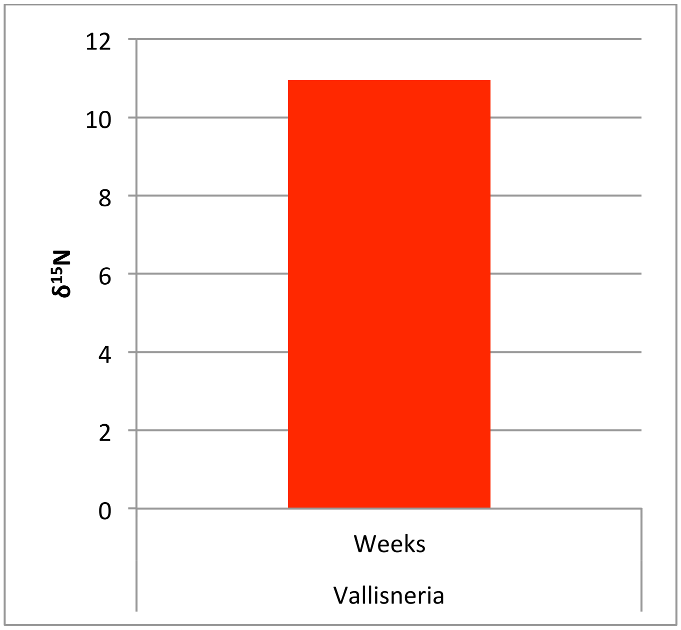

Figure 6H. $\delta^{15}N$ of Vallisneria sampled at Weeks Pond

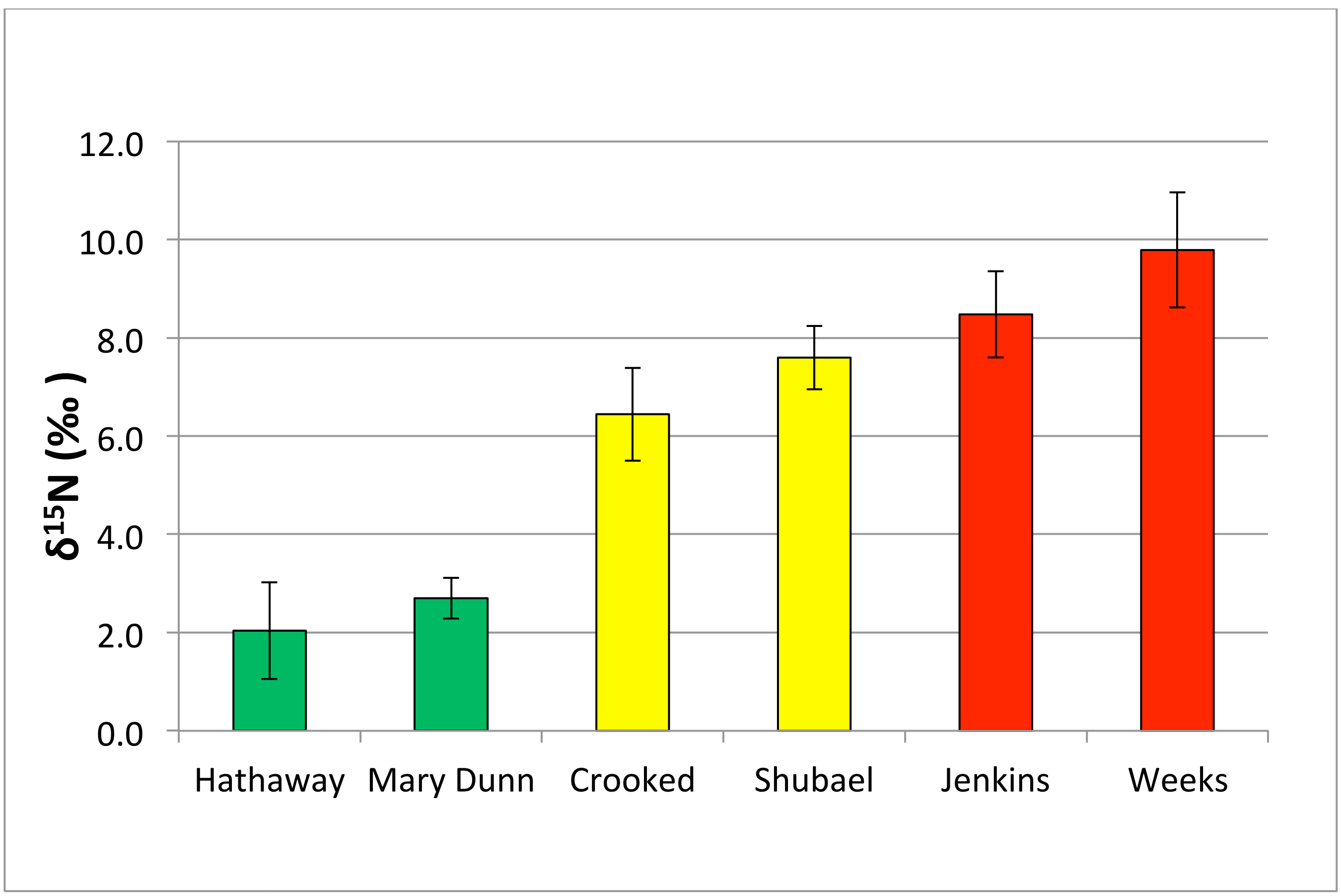

Figure 7. Average $\delta^{15}N$ of Sampled Shoreline Vegetation in Ponds

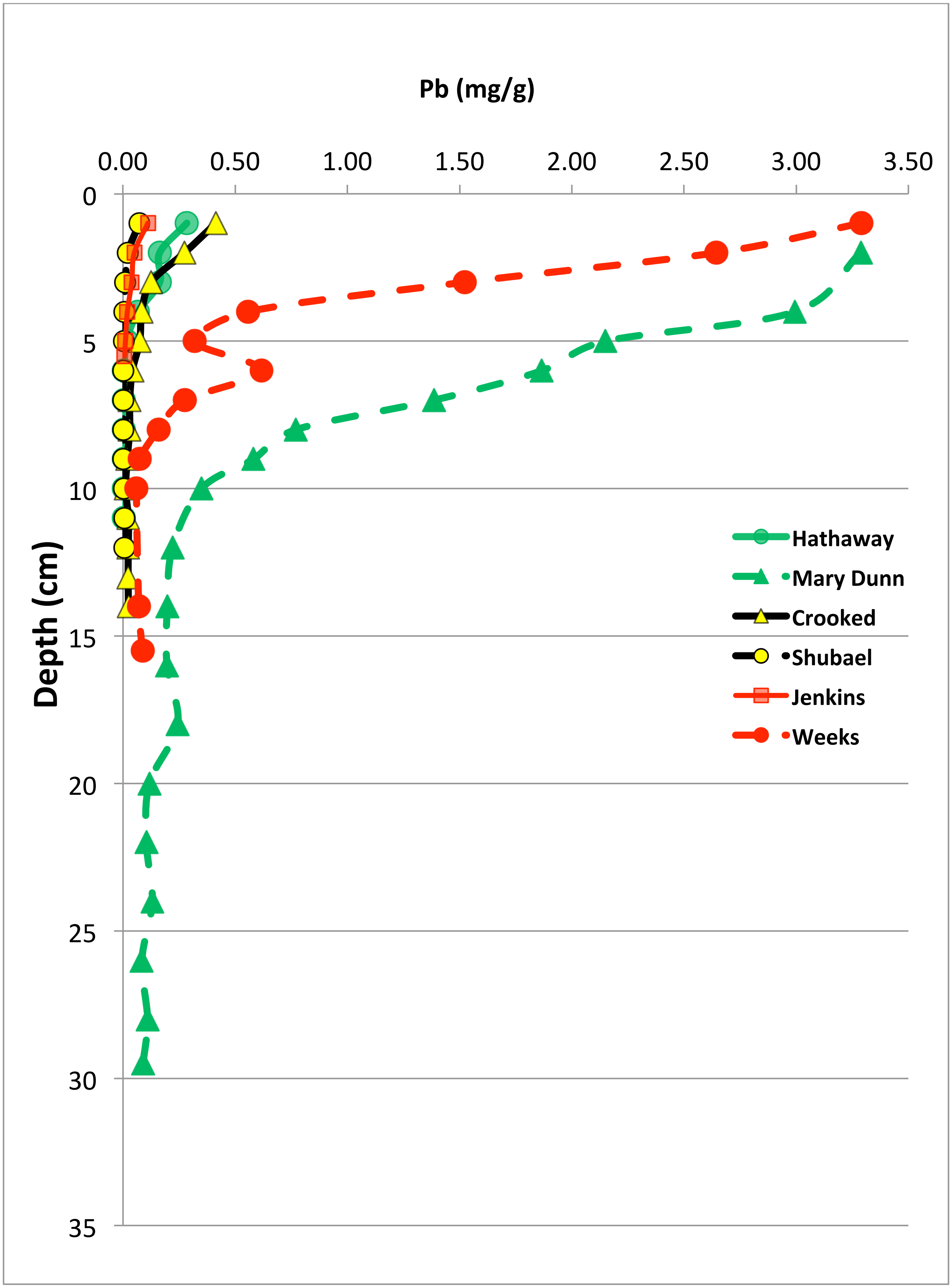

Figure 8. Pb (mg/g) in all Sampled Ponds

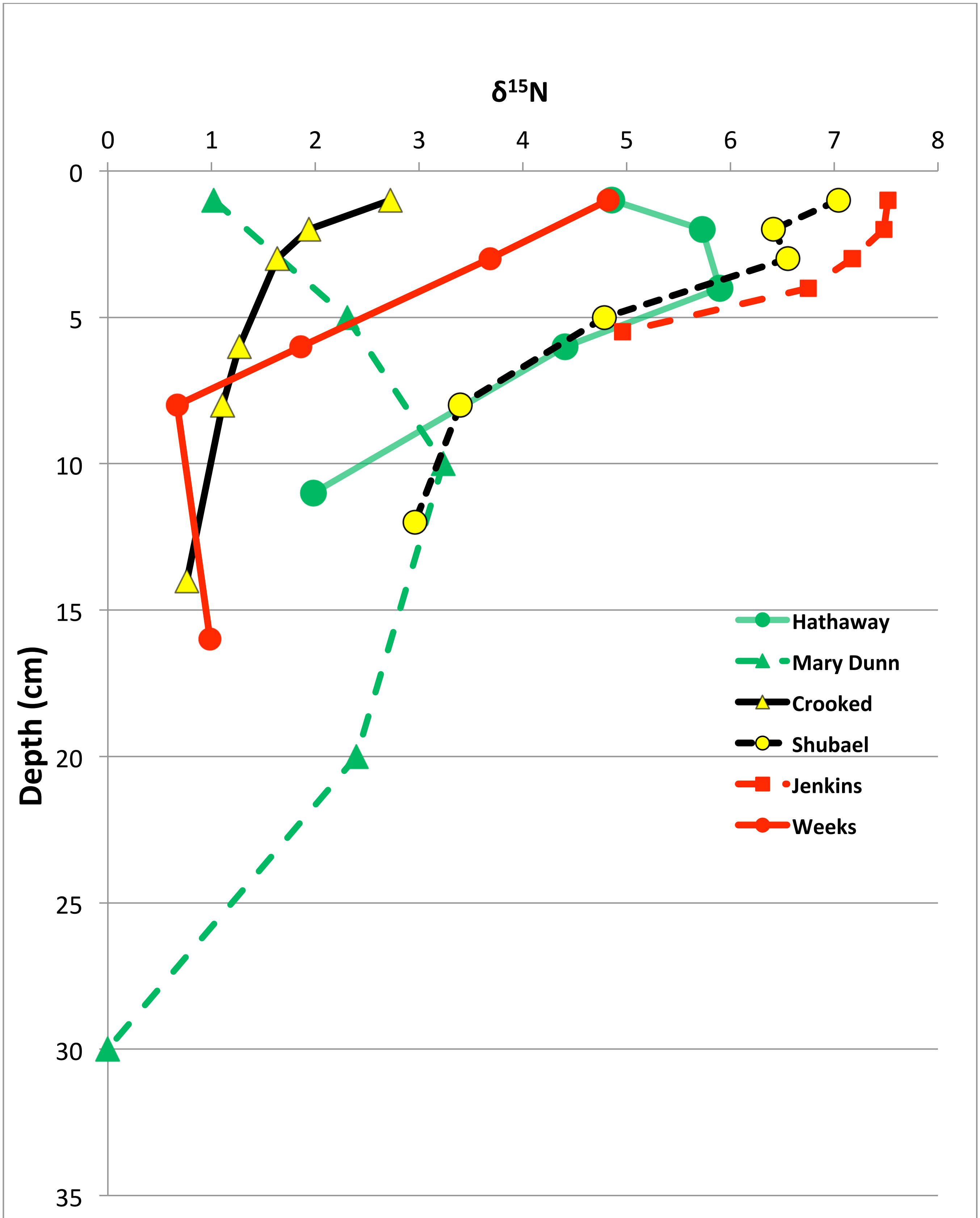

Figure 9. $\delta^{15}N$ in all Sampled Ponds

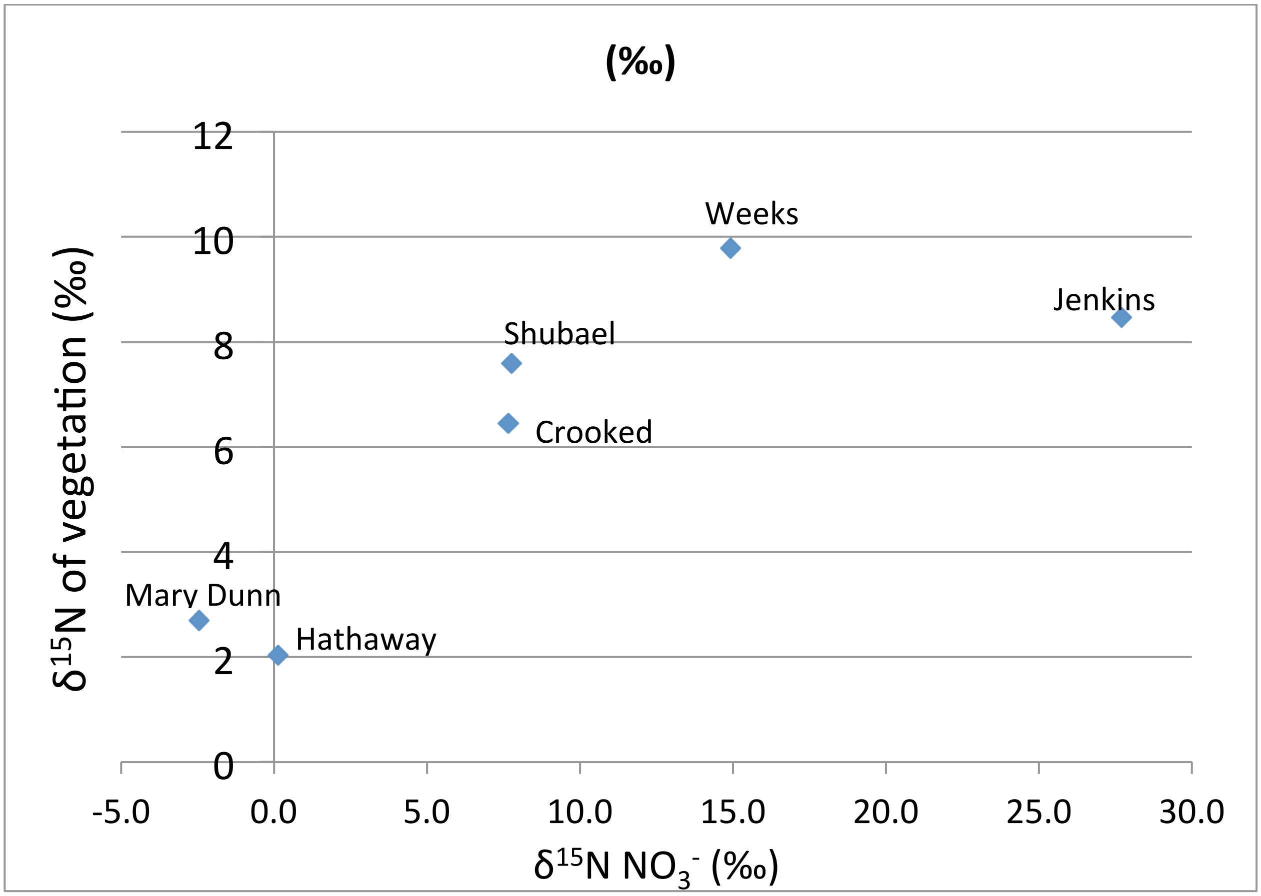

Figure 10. $\delta^{15}N$ of Groundwater NO_3^- (‰) v. $\delta^{15}N$ of Vegetation

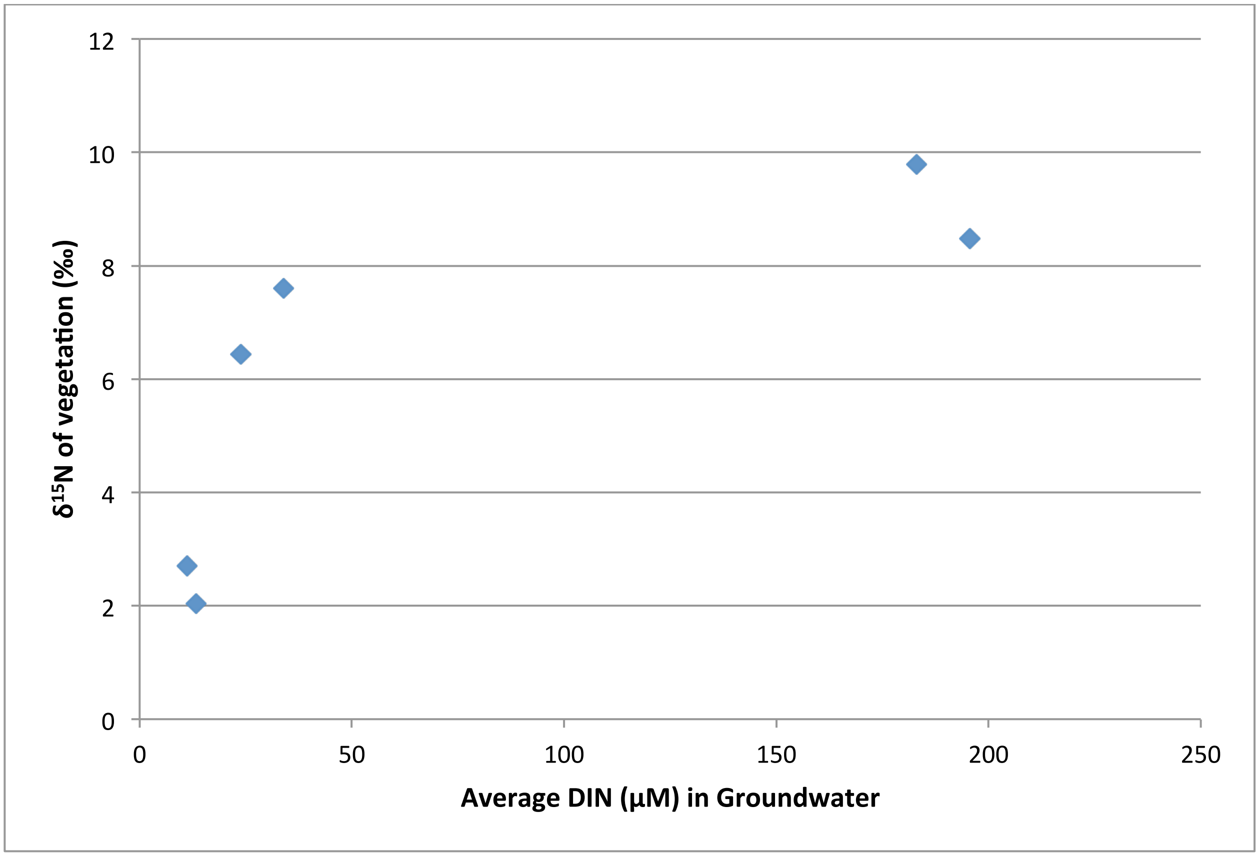

Figure 11. Average DIN (µM) in Groundwater v. $\delta^{15}N$ of Vegetation (‰)

Tables

Pond	δ^{15}N (‰) of Sampled Vegetation							
	Aquatic Moss	Eleocharis acicularis	Gratiola aurea	Juncus Militaris	Myriophyllum aquaticum	Utricularia	Utricularia eriocaulon	Vallisneria
Hathaway	1.1				3.0			
Mary Dunn		2.5	3.4	3.3	1.1	3.1		
Crooked		8.2	7.2	5.7		8.0	3.1	
Shubael		8.8	7.3	8.4			5.9	
Jenkins	10.6	9.1	7.5				6.6	
Weeks				8.6				11.0

Table 1. δ^{15}N (‰) of Vegetation sampled at six ponds