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Trends in nitrogen isotope ratios of juvenile winter flounder reflect changing nitrogen inputs to Rhode Island, USA estuarine systems

Richard J. Pruell^{1,*}, Bryan K. Taplin¹, and Kenneth M. Miller²

¹U.S. Environmental Protection Agency, Office of Research and Development, National Health and Environmental Effects Research Laboratory, Atlantic Ecology Division, 27 Tarzwell Drive, Narragansett, RI 02882, USA

²CSRA, 6361 Walker Lane, Suite 300, Alexandria, VA 22310, USA

Abstract

Nitrogen isotope ratios ($\delta^{15}\text{N}$) in juvenile winter flounder, *Pseudopleuronectes americanus*, were used to examine changes in nitrogen inputs to several Rhode Island, USA estuarine systems. Fish were collected over two three-year periods with a ten-year interval between sampling periods (2002–2004 and 2012–2014). During that interval numerous changes to nutrient management practices were initiated in the watersheds of these estuarine systems including the upgrade of several major wastewater treatment facilities that discharge to Narragansett Bay, which significantly reduced nitrogen inputs. Following these reductions, the $\delta^{15}\text{N}$ values of flounder in several of the systems decreased as expected; however, isotope ratios in fish from upper Narragansett Bay significantly increased. We believe that low $\delta^{15}\text{N}$ values measured in 2002–2004 were related to concentration-dependant fractionation at this location. Increased $\delta^{15}\text{N}$ values measured between 2012 and 2014 may indicate reduced fractionation or that changes in wastewater treatment processes altered the nitrogen isotopic ratios of the effluents.

Keywords

Stable nitrogen isotopes; fish; winter flounder; estuaries

Introduction

Efforts are underway worldwide to reduce nutrient inputs to coastal systems by changing land management practices or improving wastewater treatment systems (Pitt et al. 2009; Longphuir et al. 2015; Riemann et al. 2016). As an example, between 2002 and 2010 Rhode Island has upgraded all 11 of its wastewater treatment facilities (WWTFs) that directly discharge to upper Narragansett Bay or its tributaries to tertiary treatment. In addition, Massachusetts has upgraded 9 of 11 of its WWTFs that discharge into tributaries entering upper Narragansett Bay (RIDEM 2005; Krumholtz 2012; NBEP 2016). This has resulted in a reduction in the nitrogen load to the Bay from sewage of about 57 % compared to 2000–

*Corresponding author. pruell.richard@epa.gov.

2004 inputs (NBEP 2016). In addition to sewage upgrades, a combined storm sewer overflow reservoir constructed under the city of Providence was completed in 2008. Together, these changes have resulted in large decreases in nitrogen inputs to the Bay. Also, in the watersheds of the Rhode Island coastal ponds, located on the south coast of the State adjacent to Narragansett Bay, improved nutrient management practices have been initiated including the increased use of denitrifying septic systems. In the watershed of Narrow River, a sub-estuary emptying into the southern part of Narragansett Bay, many households have been connected to municipal WWTFs.

Nitrogen isotope ratios ($\delta^{15}\text{N}$) have been widely used to track anthropogenic nitrogen inputs to aquatic systems (Martinetto et al. 2006; Bannon and Roman 2008). Nitrogen from sewage treatment facilities or in groundwater influenced by septic systems has elevated levels of ^{15}N relative to ^{14}N (Heaton 1986; McClelland and Valiela 1998). Therefore, measurements of $\delta^{15}\text{N}$ in water, sediments and biota can be used to indicate the relative importance of sewage inputs to aquatic systems and to monitor changes over time at specific locations.

In a previous study (Pruell and Taplin 2015) we measured nitrogen isotopes in juvenile winter flounder, *Pseudopleuronectes americanus*, collected between 2002 and 2004 from several Rhode Island estuarine systems, including Narragansett Bay, several coastal ponds and the Narrow River. The objective of the present study was to resample juvenile winter flounder from those same locations after ten years to assess whether changes in nutrient inputs could be detected in the biota of these systems. Therefore, flounder were collected over two three-year periods, ten years apart (2002–2004 and 2012–2014), before and after numerous steps had been undertaken in the watersheds to reduce nitrogen inputs to these estuarine systems.

Materials and methods

Fish Collection

During two sampling periods (2002–2004 and 2012–2014) young-of-the-year winter flounder were collected from each of 20 stations (Figure 1; Table 1) along the coast of Rhode Island, USA. These sampling locations included three shallow coastal lagoons, which are locally called salt ponds (Point Judith Pond, Ninigret Pond and Green Hill Pond), an estuarine river system (Narrow River) and the upper, middle and lower portions of Narragansett Bay (Figure 1). Details on the locations and physical characteristics of these stations are included in previous publications (Pruell et al. 2012; Pruell and Taplin 2015). Figure 1 provides information on the locations and discharge flows (averaged from 2002 – 2004) from the WWTFs in the Narragansett Bay watershed.

Winter flounder was selected as the target species because they were available at all of our sampling locations and the juveniles show little movement after settlement during their first summer (Saucerman and Deegan 1991). Ten juvenile winter flounder (45 to 65 mm) were collected from each station, each year. Fish were collected using a 100-foot beach seine during June and July. Ten randomly selected individuals from each site were placed on ice, returned to the laboratory and then frozen.

Sample Preparation and Isotopic Analysis

Samples were thawed, measured for total length and the muscle tissues removed for isotopic analysis. The samples were placed into aluminum pans and dried at 60 °C for 24 h. Dried tissues were then ground to a fine powder using a mortar and pestle and stored in glass vials until analyzed. About 1.0 mg of each sample was analyzed using a GV Instruments IsoPrime (Elementar Americas, INC.) continuous flow - isotope ratio mass spectrometer (CF-IRMS). Nitrogen isotope ratios were determined using a three-point calibration line that included the international standards IAEA-N-2 (ammonium sulfate) and IAEA-N-3 (potassium nitrate) and an in-house standard (DORM-2 - dogfish tissue). Each of these standards was analyzed at the beginning, middle and end of each instrumental run. The average relative standard deviation for these triplicate standard runs was 1.3 %. A blank sample was also analyzed at the start of each sample run to verify that N₂ backgrounds were low. Blank levels were always less than 1 % of sample values, the data were not blank corrected. A sample duplicate was analyzed after every ten samples; the average isotopic difference measured for these duplicate samples was about 0.3 ‰ for $\delta^{15}\text{N}$ measurements.

Statistical Analyses

Multiple pairwise comparisons with the Bonferroni adjustment were used to test for differences in fish length. These analyses did not detect any consistent differences among years or sampling locations. In addition, regression analysis was used to test the relationship between fish length and $\delta^{15}\text{N}$ at each sampling location. No significant relationships were observed.

A series of multi-factor general linear models were used for the statistical analyses. For testing differences among sampling years, analyses were performed using a three-factor model, with sampling year and waterbody as crossed factors, station as a nested effect with waterbody, and an additional interaction term between sampling year and waterbody. Because of some year-to-year differences observed at the station level and not all stations being sampled in all years, comparisons between waterbodies were performed using a separate two-factor model for each year, with waterbody type and station (nested within waterbody) as model factors.

When assessing differences between the earlier and recent three-year sampling periods, models were fit to the combined 2002–2004 and 2012–2014 data. Due to significant interactions observed between sampling year and location (waterbody or station) for the recent three-year periods, comparisons needed to be performed specific to each location. For the river and Bay comparisons, two-factor ANOVAs were fit, with year and station included in the model. For the pond comparisons, a three-factor ANOVA was fit, with waterbody, year and station included in the model. For the river and Bay comparisons, concentrations for each of the 2012–2014 years were compared to the mean concentration across 2002–2004, using single degree-of-freedom contrasts. For individual Bay station comparisons, each 2012–2014 year was compared to each 2002–2004 year rather than to the three-year mean. This is because of differences between Bay stations among 2002–2004. Comparisons of watershed population densities and flounder $\delta^{15}\text{N}$ values were conducted using Pearson correlations.

For each model, overall F-tests for the significance of a factor or interaction term were performed at the $\alpha = 0.05$ significance level. To assess statistically significant pairwise differences between years, waterbodies or stations when all comparisons were of equal interest, Tukey-Kramer comparisons were used. To assess statistically significant pairwise differences when not all comparisons were of interest (such as between year/location combinations when a significant interaction was present), Bonferroni pairwise comparisons were performed. In both cases, the pairwise comparisons were performed to ensure a consistent overall Type I error rate ($P < 0.05$) across all comparisons being made. The assumption that the data followed a normal distribution within each year and station for all models was assessed by using both graphical review and testing the model residuals using the Anderson-Darling test, performed at $\alpha = 0.01$. Homoscedasticity (constant variance across all stations and years) was assessed by plotting the residuals against the fitted model estimates and in all cases the assumptions were met.

Results

Waterbody Comparisons

The same general trends were observed among the five different waterbodies (Figure 2; Table 1) for the 2012–2014 sampling period as had previously been reported for 2002–2004 (Pruell and Taplin 2015). However, because of variability in $\delta^{15}\text{N}$ among years (2012, 2013 and 2014) for Ninigret Pond and decreases in $\delta^{15}\text{N}$ for Green Hill Pond and Narrow River, $\delta^{15}\text{N}$ values were not consistently different among the waterbodies as was seen for the 2002–2004 period. For 2012 and 2013, Narragansett Bay > Narrow River/Pt. Judith Pond > Green Hill Pond/Ninigret Pond. However, for 2014, the pattern was Narragansett Bay > Narrow River > Pt. Judith Pond > Green Hill Pond > Ninigret Pond which was consistent with the 2002–2004 results ($P < 0.05$; F-statistics 2012: 581.2; 2013: 1236; 2014: 1365; 4 degrees of freedom (df) per contrast).

Since no significant differences in $\delta^{15}\text{N}$ ($P > 0.05$; $F=2.48$; 2 df) were observed for any of the waterbodies among the three sampling years during 2002–2004, the yearly data for each waterbody were combined for comparison to 2012, 2013 and 2014 which could not be combined due to yearly differences among the three later years. For Ninigret Pond and Narragansett Bay no clear trends emerged between the recent sampling years and 2002–2004 (Figure 3). However, for Green Hill Pond, Point Judith Pond and Narrow River, $\delta^{15}\text{N}$ values for winter flounder samples collected during 2012, 2013 and 2014 were all significantly lower than those measured during 2002–2004 from the respective waterbodies. These decreases in $\delta^{15}\text{N}$ generally ranged between 0.5 and 1.5 ‰ for all three waterbodies (Green Hill Pond 2012, $P < 0.0001$, F-statistic 29.93; 2013, $P < 0.0001$, F-statistic 175.92; 2014, $P < 0.0001$, F-statistic 53.56 : Point Judith Pond 2012, $P = 0.002$, F-statistic 9.67; 2013, $P < 0.0001$, F-statistic 49.88; 2014, $P < 0.0001$, F-statistic 53.56 : Narrow River 2012, $P < 0.0001$, F-statistic 91.75; 2013, $P < 0.0001$, F-statistic 373.55; 2014, $P < 0.0001$, F-statistic 231.46; 1 df per contrast).

Although there were some differences in the $\delta^{15}\text{N}$ values of the winter flounder among waterbodies between the two sampling periods, there continued to be strong relationships between the 2010 human population density of the watersheds (RIGIS 2013) and the $\delta^{15}\text{N}$

of the flounder. For each of the three years (2012, 2013 and 2014) highly significant correlations were observed between watershed population density and mean $\delta^{15}\text{N}$ in the fish (Figure 4) collected from the respective waterbodies (2012: $r = 0.96$; $P < 0.01$, 4 df; 2013: $r = 0.95$; $P < 0.01$, 4 df; 2014: $r > 0.99$; $P < 0.01$, 4 df).

Narragansett Bay

Although a large gradient in nitrogen concentrations existed in Narragansett Bay, a relatively small range of $\delta^{15}\text{N}$ ratios was observed in the flounder tissues throughout the Bay between 2002 and 2004 (Table 1; Figure 5a). The nitrogen isotope ratios of winter flounder measured at the eight Narragansett Bay stations between 2012 and 2014, however, showed a trend that is more consistent with the anthropogenic nutrient gradient in the Bay (Figure 5b). For all but one station in Narragansett Bay there were few or no yearly differences in $\delta^{15}\text{N}$ among the six sampling years. The exception to this was station PCM. The nitrogen isotope ratios in winter flounder collected from PCM during 2012, 2013 and 2014 were significantly higher all three years ($\delta^{15}\text{N}$ mean values; 2002, 13.6; 2003, 12.7; 2004, 13.4 vs 2012, 15.9; 2013, 15.7; 2014, 17.0 ‰) when compared to those samples collected during 2002, 2003 and 2004 (overall test for differences across years: $F=164.5$; Bonferroni pairwise comparisons, t between 10.6 and 22.9; $P < 0.0001$ for all pairs).

Discussion

A comparison of the $\delta^{15}\text{N}$ values of winter flounder collected during our two sampling periods (2002–2004 and 2012–2014) shows that there were some statistically significant decreases in the $\delta^{15}\text{N}$ values of the fish during the decade between sampling periods. The most consistent decreases in $\delta^{15}\text{N}$ occurred in Green Hill Pond, Pt. Judith Pond and Narrow River. In Green Hill Pond and Pt. Judith Pond these changes may be related to the increased use of denitrifying wastewater treatment systems by homeowners in the watershed due to regulations enacted by the Rhode Island Department of Environmental Management in 2008 (RIDEM 2008). In the late 1980s and early 1990s many houses in the Narrow River watershed were connected to the municipal sewer system and several stormwater management projects were initiated and resulted in decreasing anthropogenic loadings to Narrow River (Berounsky and DeSilva 2012) which may have contributed to the changes observed in that system. Only small changes in the overall mean $\delta^{15}\text{N}$ values were seen in Narragansett Bay between the two sampling periods.

Estimates of anthropogenic nitrogen inputs to these estuarine systems are available; however, input estimates are highly variable among studies. Also, comparisons between the coastal ponds and Narragansett Bay are difficult because the coastal lagoons and Narrow River receive nutrient inputs mostly from groundwater whereas the largest source of nutrients to Narragansett Bay is from wastewater treatment facilities (Pruell and Taplin 2015). Agricultural activity is limited in all of these watersheds so only minor inputs from these activities occur. Despite all of the differences in nitrogen input routes and estuarine characteristics a strong correlation between $\delta^{15}\text{N}$ in juvenile winter flounder in these estuarine systems and human population density of the watersheds had previously been reported during the 2002–2004 sampling period (Pruell and Taplin 2015). Again, in the

present study there was a highly significant correlation between $\delta^{15}\text{N}$ in the flounder and human population density of the watersheds for each of the three more recent sampling years. These findings are consistent with Bowen and Valiela (2001) who reported that the contribution of wastewater to total nitrogen inputs increased with watershed population density in a series of New England estuaries. Bannon and Roman (2008) found that population density was the best predictor of $\delta^{15}\text{N}$ in estuarine biota of all the watershed characteristics considered. Also, Morris et al. (2015) reported that $\delta^{15}\text{N}$ values in fish from several Australian estuaries were significantly correlated with percent urbanization and Schlacher et al. (2005) showed that $\delta^{15}\text{N}$ values in numerous fish species were related to sewage inputs in three Australian estuarine systems.

Greater than 80% of the sewage inputs to Narragansett Bay occur in the northernmost areas of the estuary (Oczkowski et al. 2008). Because of this there has been a large north-south gradient in surface seawater nitrogen concentrations in the Bay (Oviatt et al. 2002; Pruell et al. 2006; Thornber et al. 2008). Since $\delta^{15}\text{N}$ values of sewage NO_3^- as high as 28 ‰ were reported for WWTFs in the Narragansett Bay watershed (Schmidt et al 2016) and because the oceanic $\delta^{15}\text{N}$ endmember is generally considered to be in the range of 5 to 8 ‰ for NO_3^- (Fry 2002; Dahnke et al 2008), a large gradient in $\delta^{15}\text{N}$ values would be expected along the Bay gradient. However, the $\delta^{15}\text{N}$ trends observed in winter flounder collected along this gradient between 2002 and 2004 did not show a decrease with distance down the estuary. A possible reason for this unexpected trend could have been the differential fractionation of nitrogen isotopes during uptake by phytoplankton, benthic photosynthesizers and bacteria at the base of the food web along the nutrient concentration gradient in Narragansett Bay (Pruell and Taplin 2015). Cifuentes et al. (1989) documented nitrogen fractionation by phytoplankton in Delaware Bay and Hoch et al. (1992) reported that marine bacteria fractionate nitrogen isotopes at high NH_4^+ concentrations. Montoya et al. (1991) detected different degrees of $\delta^{15}\text{N}$ fractionation between NH_4^+ and particulate nitrogen along the nutrient gradient in Chesapeake Bay and Cifuentes et al. (1988) reported that fractionation of NH_4^+ by phytoplankton occurred at concentrations above 20 μM in Delaware Bay. Liu et al. (2013) examined the relationship between ammonium concentration and the degree of nitrogen isotope fractionation by phytoplankton and found that the greatest amount of isotopic fractionation occurred at ammonium concentrations between about 10 and 60 μM . Pruell et al. (2006) reported concentrations of NH_4^+ and NO_3^- as high as 46.7 and 15.9 μM , respectively, at the PCM station in 1999. Therefore, the nitrogen concentrations measured at this site may have been high enough to cause significant concentration-dependent nitrogen isotope fractionation in phytoplankton and bacteria. This may have led to the preferential accumulation of a lighter $\delta^{15}\text{N}$ signature by the base of the food web at this location which was then passed to consumers such as winter flounder.

During the time between our two sampling periods (2002–2004 and 2012–2014) all of the major WWTFs in the Bay were upgraded (Krumholz 2012; NBEP 2016). Also, a combined storm sewer overflow reservoir designed to reduce the release of untreated effluents to Narragansett was completed. These changes have resulted in significant decreases in the nitrogen inputs to Narragansett Bay (Krumholz 2012; NBEP 2016). It would generally be expected that decreases in sewage inputs would result in reductions in $\delta^{15}\text{N}$ values. Pitt et al. (2009) found that $\delta^{15}\text{N}$ of algae and crabs decreased following the upgrade of WWTFs in an

Australian estuarine system. Also, Costanzo et al. (2005) found that $\delta^{15}\text{N}$ values in the red macroalga, *Catenella nipae*, decreased after wastewater treatment upgrades in Moreton Bay, Australia and Savage (2005) reported similar findings for a macroalgal species (*Fucus vesiculosus*) from a Baltic Sea embayment. However, instead of decreasing, $\delta^{15}\text{N}$ values in the winter flounder significantly increased by about 2–3 ‰ all three years for the more recent sampling period (2012 to 2014) at the northernmost station (PCM) in Narragansett Bay. As a result, the $\delta^{15}\text{N}$ trend for winter flounder now shows the expected decrease with distance down the estuary.

We believe that there are two possible explanations for the large increases in $\delta^{15}\text{N}$ observed at our northernmost station (PCM). The first is that in systems such as upper Narragansett Bay with high nutrient concentrations, $\delta^{15}\text{N}$ values in the food web at this site may be low due to concentration-dependent fractionation at the primary producer level as described above. This effect may have been the reason for the depleted $\delta^{15}\text{N}$ values measured between 2002 and 2004 at the PCM station (Pruell and Taplin 2015). Decreasing nutrient inputs during the decade between our sampling periods may have reduced nutrient concentrations below levels that cause concentration-dependent fractionation and thereby have resulted in increased nitrogen isotope ratios in this region. Data collected by the Narragansett Bay Commission at the Bullucks Reach Buoy (Narragansett Bay Commission 2014), which is very close to our PCM station, showed that the mean concentrations observed between 2012 and 2014 ($n = 32$) were 4.5 and 3.1 μM for NO_3^- and NH_4^+ , respectively. These values are below the levels at which fractionation has been shown to occur (Liu et al. 2013).

The second possible reason for the increase in $\delta^{15}\text{N}$ values is that upgrades of wastewater systems to tertiary treatment can increase in the $\delta^{15}\text{N}$ of the NO_3^- released by the facility due to the fractionation associated with the microbial denitrification employed in the tertiary treatment process (Savage 2005; Schmidt 2014). In fact, Gaston et al. (2004) reported that eastern hula fish (*Trachinops taeniatus*) collected near secondary and tertiary WWTFs had higher $\delta^{15}\text{N}$ values when compared with those collected near primary facilities. After the upgrade of several WWTFs within the Narragansett Bay watershed to tertiary treatment, Schmidt et al. (2016) reported that there was a 7–8 ‰ increase in sewage NO_3^- - $\delta^{15}\text{N}$ values and a 2 ‰ increase in the $\delta^{15}\text{N}$ value of NO_3^- in the Providence River region of Narragansett Bay. Therefore, this may have contributed to the increase observed in the $\delta^{15}\text{N}$ values of winter flounder collected at the PCM station. At the present time it is not possible to determine if one or both of these processes are responsible for the large change observed at this station in upper Narragansett Bay. In either case, these observations indicate that changing nutrient practices in the watershed are now being reflected in the biota of Narragansett Bay.

Conclusions

Large differences in the $\delta^{15}\text{N}$ values of muscle tissues of juvenile winter flounder were found among estuarine waterbodies studied along the coast of Rhode Island. Some reductions in winter flounder $\delta^{15}\text{N}$ values were found between the 2002–2004 and 2012–2014 sampling periods in the systems (particularly Green Hill Pond, Point Judith Pond and Narrow River). However, in general the differences among systems persisted over the six

years of the study and were consistent with the extent of anthropogenic influence as indicated by human population density in the watershed in these estuaries.

The nitrogen isotope ratios observed in the highly nutrient-enriched upper portion of Narragansett Bay suggest that fractionation at the base of the food web due to high nitrogen concentrations may have resulted in lower than expected isotopic ratios at the northernmost station in Narragansett Bay (PCM) during our 2002–2004 sampling period. However, fish collected at PCM between 2012 and 2014 were found to have $\delta^{15}\text{N}$ values that had increased by about 2–3 ‰. This highly significant increase may have resulted from two possible factors related to wastewater treatment upgrades in the watershed. Reductions in nutrient concentrations in the upper Bay region may have reduced the concentration-dependent fractionation in primary producers. Also, upgrades to tertiary treatment may have increased the $\delta^{15}\text{N}$ values of NO_3^- and NH_4^+ released to the estuary.

These findings indicate that $\delta^{15}\text{N}$ in estuarine biota can be used to infer general trends in the relative importance of anthropogenically derived nitrogen in estuarine systems; however, because of complex nitrogen biogeochemistry caution is needed when interpreting these trends in aquatic ecosystems with high anthropogenic nitrogen loadings. Nitrogen inputs to several Rhode Island watersheds have been reduced due to improved management practices in the watersheds over the past decade. Based on our results it appears that these efforts are now being reflected in the biota of these system.

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Figure 1. Sampling locations for juvenile winter flounder along the coast of Rhode Island, USA. The sampling sites included three coastal ponds (Ninigret Pond, Green Hill Pond and Pt. Judith Pond), an estuarine river system (Narrow River) and Narragansett Bay. The station identifications are listed in Table 1. Locations and annual effluent flows for the wastewater treatment facilities in the area averaged over 2012 – 2014 are also shown.

Figure 2. Mean $\delta^{15}\text{N}$ values ($\pm$ standard deviation) of juvenile winter flounder collected from the five Rhode Island coastal waterbodies for each of the three sampling years (2012, 2013 and 2014).

Figure 3. Change in mean $\delta^{15}\text{N}$ values ($\pm$ standard error) of juvenile winter flounder collected from the five Rhode Island coastal waterbodies between the 2002–2004 sampling period and each of the three more recent sampling years (2012, 2013 and 2014). * - indicates significant difference ($P < 0.05$) compared to 2002–2004 mean values.

Figure 4.

Correlation between 2010 human population density in the watershed (RIGIS 2013) and mean $\delta^{15}\text{N}$ values of the muscle tissue of juvenile winter flounder collected from each of the systems during 2012, 2013 and 2014 (2012: $r = 0.97$; $P < 0.01$, 4 df; 2013: $r = 0.95$; $P < 0.01$, 4 df; 2014: $r = 1.00$; $P < 0.01$, 4 df).

Figure 5.

(a) Mean $\delta^{15}\text{N}$ values ($\pm$ standard deviation) of juvenile winter flounder collected from the eight sampling stations within Narragansett Bay for each of the three sampling years (2012, 2013 and 2014). (b) Mean $\delta^{15}\text{N}$ values of juvenile winter flounder collected from the eight sampling stations within Narragansett Bay for each of the three sampling years (2012, 2013 and 2014). * - indicates significant difference ($P < 0.05$) between the sampling periods.

Table 1.

Stable nitrogen isotope values of juvenile winter flounder muscle tissue during 2, 3-year periods (2002–2004 and 2012–2014). Fish were collected from three habitat types (U = unvegetated, E = eelgrass and M = macroalgae) along the Rhode Island coastline (n = 10 for each value).

Location	Station Code	$\delta^{15}\text{N} \pm \text{S.D. (‰)}$				$\delta^{15}\text{N} \pm \text{S.D. (‰)}$			
		2002	2003	2004	2012	2013	2014		
<u>Narragansett Bay</u>									
Passaconquis Cove-M	PCM	13.7 ± 0.4	12.7 ± 0.4	13.4 ± 0.4	15.9 ± 0.4	15.7 ± 0.5	17.0 ± 0.3		
Barrington Beach-U	BBU	14.6 ± 0.5	14.7 ± 0.5	14.1 ± 0.7	14.4 ± 0.4	NA	15.2 ± 0.6		
Greenwich Bay-M	GBM	14.3 ± 0.3	14.4 ± 0.4	14.5 ± 0.2	14.2 ± 0.2	13.6 ± 0.4	14.3 ± 0.3		
Prudence Island-E	PIE	14.9 ± 0.7	14.7 ± 0.5	14.1 ± 0.5	14.6 ± 0.6	14.4 ± 0.3	14.6 ± 0.2		
Wickford Beach-U	WBU	14.8 ± 0.4	15.0 ± 0.3	14.9 ± 0.5	15.1 ± 0.2	14.7 ± 0.3	14.7 ± 0.4		
East Ferry-E	EFE	15.0 ± 0.4	15.2 ± 0.4	14.0 ± 0.5	14.4 ± 0.4	13.9 ± 0.3	13.5 ± 0.4		
Fox Hill-M	FHM	13.0 ± 0.6	14.5 ± 0.4	13.6 ± 0.3	12.8 ± 0.5	12.7 ± 0.4	13.5 ± 0.6		
Sheffield Cove-U	SCU	13.0 ± 0.3	14.0 ± 0.4	13.3 ± 0.3	12.4 ± 0.5	13.0 ± 0.3	13.2 ± 0.3		
<u>Narrow River</u>									
Narrow River-U	NRU	12.9 ± 0.3	13.1 ± 0.3	13.0 ± 0.3	12.2 ± 0.3	11.5 ± 0.3	12.3 ± 0.4		
Narrow River-E	NRE	13.7 ± 0.2	13.7 ± 0.4	13.6 ± 0.4	13.4 ± 0.3	12.1 ± 0.1	12.3 ± 0.4		
Narrow River-M	NRM	13.2 ± 0.4	13.8 ± 0.7	13.8 ± 0.2	12.6 ± 0.3	12.6 ± 0.2	12.4 ± 0.3		
<u>Pt. Judith Pond</u>									
Pt. Judith-M	PJM	12.8 ± 0.3	13.1 ± 0.4	12.8 ± 0.4	12.3 ± 0.4	11.3 ± 0.4	12.1 ± 0.2		
Pt. Judith-E	PJE	12.9 ± 0.4	13.2 ± 0.4	13.2 ± 0.4	12.6 ± 0.2	12.8 ± 0.4	12.0 ± 0.4		
Pt. Judith-U	PJU	12.5 ± 0.3	12.8 ± 0.2	12.6 ± 0.3	12.8 ± 0.1	12.5 ± 0.4	11.9 ± 0.3		
<u>Green Hill Pond</u>									
Green Hill-M	GHM	NA	11.4 ± 0.5	11.6 ± 0.5	10.6 ± 0.4	10.1 ± 0.5	10.5 ± 0.3		
Green Hill-E	GHE	NA	10.5 ± 0.4	NA	10.5 ± 0.5	9.6 ± 0.3	10.3 ± 0.4		
Green Hill-U	GHU	11.4 ± 0.4	10.7 ± 0.3	11.7 ± 0.2	10.7 ± 0.5	9.7 ± 0.1	10.7 ± 0.4		
<u>Ninigret Pond</u>									
Ninigret-M	NGM	9.3 ± 0.5	9.7 ± 0.5	9.4 ± 0.5	NA	9.2 ± 0.4	9.0 ± 0.3		
Ninigret-E	NGE	9.2 ± 0.8	9.1 ± 0.5	9.0 ± 0.4	9.7 ± 0.4	9.1 ± 0.2	8.7 ± 0.3		
Ninigret-U	NGU	11.2 ± 0.2	10.1 ± 0.7	11.1 ± 0.9	10.9 ± 0.4	10.8 ± 0.3	9.5 ± 0.5		

NA - fish not available