Using Multiple Approaches to Evaluate the Effects of Flow Regulation on Fishes in the Tallapoosa River, Alabama.

By

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Abstract:

Dams alter many aspects of riverine environments and generally have broad effects both upstream and downstream. Despite the variety of potential benefits of dams to humans (recreation, flood control, navigation, etc.), they can also negatively affect riverine ecosystems. In particular, hydropeaking dams affect downstream fish habitats by increasing water velocity and altering water temperature regimes. Since 1983, the discharge and water temperatures in the Tallapoosa River, Alabama, have been regulated by Harris Dam. In 2005, a group of stakeholders successfully petitioned to implement an adaptive management plan (the Green Plan) on the operation of Harris Dam which set limits on the timing, schedule, and duration of water releases. To assess the effects of Harris Dam operating under the Green Plan, I collected fish from four sites on the Tallapoosa River, three of which were downstream of and regulated by Harris Dam, and one site upstream and unregulated by Harris Dam. I used multiple approaches to quantify patterns in fish assemblage structure, and diet composition and movement patterns of four recreationally important species - Channel Catfish *Ictalurus punctatus*, Redbreast Sunfish *Lepomis auritus*, Alabama Bass *Micropterus henshalli*, and Tallapoosa Bass *Micropterus tallapoosae*.

First, I quantified the fish assemblage structure using Shannon’s H, nonmetric multidimensional scaling (NMDS), a multiresponse permutation procedure (MRPP), and indicator species analysis to determine if the distribution of species varied among sites along a gradient downstream of Harris Dam. Shannon's H varied little across my sites, whereas NMDS and MRPP revealed significant assemblage differences among sites. The tailrace fish assemblage was distinct from the other downstream sites and was characterized by higher number of fluvial specialist species. This suggests that the tailrace assemblage may favor species that are able to persist through higher flows at the expense of other native species that are not. Additionally, target species' diets were quantified, revealing spatially variable diet compositions. Diets (proportion by weight) from fish collected in the tailrace were distinct from those from the other sites in that they contained some prey types (e.g., amphipods, isopods) not found in diets at the other sites.

Trace element ratios in otolith edges (i.e., recently incorporated material) and water samples were weakly positively related. In addition, water trace element ratios did not vary seasonally for any element except barium. Elemental signatures in both water and otoliths varied across sites with the largest difference occurring between the upstream unregulated site and the three downstream regulated locations. Differences between the three downstream sites were less apparent, and Sr:Ca ratios were identified as the most informative of the four elemental ratios analyzed (Sr:Ca, Ba:Ca, Mn:Ca, Mg:Ca). Using linear discriminant analysis, otoliths were correctly assigned to capture region with overall accuracy of 39.5-82.7% depending on the otolith region being considered (core, edge, or entire otolith transect). Variation in classification accuracy among otolith segments indicated potential ontogenetic shifts in site fidelity, although interpretation was limited by low variation across downstream site element ratios. Strontium ratios across the entire otolith ablation transect suggested three predominant movement patterns: 1) individuals spent their entire life at the capture location, 2) individuals recruited to the capture location from a different river section, and 3) individuals moved away from, then returned to, the capture location. Distributions of these three patterns indicated that fish in the tailrace recruited from a limited area whereas fish at downstream sites recruited from, and continued to use, a broader area of the river.

To further quantify the effects of Harris Dam on fish movement, combined acoustic and radio tags were surgically implanted into (13) Alabama Bass and (3) Tallapoosa Bass individuals. An acoustic array of 10 stationary receivers was deployed in the area immediately downstream of the Harris Dam tailrace, and fish were also manually tracked with a radio receiver. Telemetry data revealed that longitudinal movement of black basses was minimal in response to the Harris Dam operation, with a maximum net longitudinal movement over the course of the study being 6.3 km.

Overall, considering fish assemblages, diets, and trace element analyses, my results from the tailrace consistently differed from the other downstream sites. These results suggest that although the Green Plan may have mitigated some of the initial effects of peaking hydropower flows from Harris Dam, the current operation of the dam continues to affect downstream fishes.

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Background

Dams are an ecological threat to aquatic ecosystems, disrupting over half of the world’s major river systems (Nilsson et al. 2005). Although dams vary in their characteristics such as size and function, they have common effects on habitat, discharge, and energy inputs. Throughout the 18th and 19th centuries, river systems were often viewed as untapped resources that were only viewed positively when they met the needs of humanity (Billington et al. 2005). This mindset heralded a period of dam construction that resulted in flow regulation of the majority of navigable rivers in the United States (NRC 1992). All dams interrupt the movement of water and aquatic organisms and can disrupt energy and nutrient flows. Organisms in river systems rely on energy inputs from both upstream and downstream habitats, and structures interrupting river connectivity generally have watershed-scale effects (Vannote et al. 1980; Ward 1989; Schindler et al. 2013). Upstream, inundation eliminates terrestrial habitats, increases water depth, and removes turbulent flow, significantly affecting lotic specialists (Nilsson et al. 2005). Downstream, releases from the dams alter habitats, create unnatural temperature variation, alter flow variation, limit channel development/maintenance, and reduced system productivity (Nilsson et al. 2005).

Low-head dams refer to those that create a height difference generally <7.5 m between the upstream reservoir and downstream river (Stanley et al. 2002). Additionally, low-head dams have lesser impacts on flow regimes than dams with higher heads and often provide storage for various services including navigational locks, irrigation, and mill power (Stanley et al. 2002). Although these low-head dams act as barriers to aquatic organism movement, their downstream footprints are smaller compared to larger structures (Fencl et al. 2015). High-head dams create significant height differences between the reservoir's surface and the surface of the tailrace, increasing potential for energy generation. There are two main types of hydropower facilities: run-of-the-river (or diversion) plants and impoundment plants (U.S.Department of Energy 2020). Run-of-the-river plants are generally smaller and do not always require a dam/reservoir for storage; instead, they divert the river's channel through a canal and simply rely on the force of the river's flow to generate power (U.S.Department of Energy 2021). Impoundment plants are typically larger and require a dam and a reservoir for energy production. Impoundment systems can operate with many different schedules including hydropeaking operation where power is generated during periods of high electricity demand (Young et al. 2011). Despite dams having a variety of additional potential benefits, such as recreation, flood control, water supply, and navigation, operation of these dams can negatively affect downstream aquatic life (Graf 1999; Young et al. 2011; Kaunda et al. 2012). In addition to increased variation in discharge, both downstream water temperature and dissolved oxygen concentrations can be effected depending on dam operations (Ashby et al. 1999). In systems where water is drawn from the upstream reservoir hypolimnion, water released into the tailrace can be anoxic and colder than the ambient river temperature. Surface water withdrawal can release warmer water and epilimnetic organisms into the tailrace. Variable tailrace conditions alter and/or degrade pre-dam habitats and influence aquatic life, including fishes (Carolli et al. 2012; Taylor et al. 2012; Cooper et al. 2017). High flow periods can scour river substrates, modify riverbanks, and degrade littoral zones (Cooper et al.

2017). These conditions have a myriad of effects on fishes, including increased energy expenditure, decreased growth, decreased (or even eliminated) spawning success, and physical displacement (Cooper et al. 2017, Weyers et al. 2003).

R.L. Harris Dam in Alabama is an example of an impoundment hydropower dam operated on a hydropeaking schedule. Constructed in 1983, the dam initially operated with no partial flow between generation periods. Harris Dam's hypolimnetic release results in both thermal and flow modification of the downstream area and released water is potentially anoxic. The subsequent change in downstream water temperature could reach 10° C and the decrease remained measureable many kilometers downstream from the tailrace (Irwin and Freeman 2002). In 2005, stakeholders adopted an adaptive management plan, the Green Plan, that established regulations on dam operations including base flow requirements between generation periods (Irwin and Freeman 2002; Kleinschmidt Associates 2018). Although the Green Plan mitigated some of the effects of dam operation, water was still released from the hypolimnion, leading to pulses of higher flow with colder water temperatures during the summer. Downstream effects of dam operations under the Green Plan on fish populations and assemblages have not been thoroughly described.

Here I used multiple approaches to quantify the effects of dam operations on downstream fishes. First, I quantified fish assemblage structure (species richness and diversity) and diet composition of selected target species across a spatial gradient to quantify effects of dam operation on the overall fish assemblage and individual fish growth. Second, I used trace element analysis to quantify coarse-scale movement and to identify natal origins. I also used telemetry with combined acoustic and radio tags

(CART tags) in conjunction with both passive and active tracking to quantify longitudinal fish movement in the 12 km immediately downstream of Harris Dam.

Chapter 1: An update on the status of fish communities in the Tallapoosa River with descriptions of sport fish diet compositions

Introduction

Alabama is considered a hotspot of aquatic biodiversity and each river system in the state has its own assemblage of native fishes (Freeman et al. 2005). Historically as many as 126 fish species have been identified in the Tallapoosa River, and the broader Mobile drainage is one of the most diverse fish assemblages in North America (Travnichek and Maceina 1994; Freeman et al. 2005). Protecting biodiversity in Alabama rivers is increasingly important as anthropogenic impacts increase. To protect aquatic resources (ecosystems, communities, and populations), it is necessary to identify how various threats (such as a dam) affect the ecology and life history of resident organisms. The Tallapoosa River from the tailrace of Harris Dam to the headwaters of Lake Martin is diverse in both habitat and resident fishes and is considered one of the highest quality river segments in the piedmont region of Alabama (Irwin and Freeman 2002). Although little information about the Tallapoosa River fish assemblage is available from before Harris Dam was constructed, several studies of the fish assemblage have been conducted since 1983 (Kinsolving and Bain 1993; Travnichek and Maceina 1994; Irwin and Freeman 2002; Freeman et al. 2005; Irwin et al. 2019). Beyond the effects of hydropeaking dam operation discussed earlier, Travnichek and Maceina (1994) showed that the diversity and richness of fish inhabiting shallow downstream waters was decreased in regulated portions of the Tallapoosa River versus upstream/unregulated areas.

Work on popular sportfish within the Tallapoosa River has showed that age-and-growth and reproductive success were reduced by the operation of Harris Dam (Martin 2008; Goar 2013; Earley and Sammons 2018). Some literature exists quantifying the diet composition of Redbreast Sunfish in the Tallapoosa River; however, work concerning Alabama Bass has been restricted to other river systems, diet characterizations for Channel Catfish have primarily focused on lentic populations and aquaculture settings, and studies of diets of Tallapoosa Bass (and redeye basses in general) are scarce (Martin 2008; Weisburg and Janicki 2011; Braun and Phelps 2016).

Here I collected and analyzed diets from four target species from above Harris Reservoir and along a downstream gradient to test the potential impact of dam operations on foraging and to support bioenergetics analyses conducted as a part of a larger research study (DeVries et al. 2021). In addition, I quantified the fish assemblage composition across the same gradient of sites

Methods

Fish Collection

Fish were collected by pulsed-DC boat electrofishing (Midwest Lake Management, Inc. Missouri, USA) once every other month from April 2019 through February 2021 (a total of 12 bimonthly sampling events), with sampling at each site consisting of six, 600-sec transects at three sites located on the mainstem Tallapoosa River below Harris Dam (tailrace, Wadley, Horseshoe Bend). An additional site upstream of Harris Reservoir served as an unregulated reference site (Lee’s Bridge; Figure 1). Output voltage was standardized between 700-900 volts with 100-120 pulses per second, and GPS coordinates were recorded at the start and end of each transect. A tow-barge electrofisher was used at the tailrace site given that it is inaccessible by boat; barge sampling consisted of one individual with the anode and 1-2 dip-netters wading alongside, with another individual pushing the barge itself. Barge electrofishing followed the same procedures, although a slightly lower voltage (500-700 volts) was used for safety.

Target species were four recreationally important species and were selected to encompass a range of life history, diet, and habitat use/strategies: Channel Catfish *Ictalurus punctatus*, Redbreast Sunfish *Lepomis auritus*, Alabama Bass *Micropterus henshalli*, and Tallapoosa Bass *Micropterus tallapoosae.* For each sampling date, dissolved oxygen and temperature were measured just below the surface (YSI model 55). Sampled fish were either measured, weighed, and returned to their sample location or kept on ice and returned to the lab for identification. For roughly half the sample events, all collected fish were bagged and immediately placed in an ice water slurry with fish from each transect stored separately; for the remainder of the sampling events, target species individuals were kept separate by transect in an ice water slurry while non-target individuals were identified, measured (nearest mm TL), weighed (nearest g), and returned to the area from which they were collected.

Laboratory Methods

In the lab, all collected fish were identified to species and up to 10 individuals of each non-target species were weighed and measured. If more than 10 individuals of a given species were present in a transect, the remaining individuals were counted, and the group was bulk weighed. The same methods were used when the non-target species were processed and returned to their capture location in the field. Each individual of the target species was weighed, measured, and sexed. Additionally, stomach contents and sagittal otoliths (lapillar otoliths for ictalurids) were extracted from all collected individuals of each target species. Stomach contents were viewed under a dissecting microscope and all prey items were identified to the lowest taxon possible, measured to the nearest 0.1 mm along their longest axis using an ocular micrometer, and counted; a note was made if the item was not whole (e.g., a head, an otolith, etc.). In instances where large numbers of a diet type were present, a haphazard subsample of 10 individuals of that diet type was measured, the remaining items were counted, and the total number recorded.

Data Analysis: Assemblage

Shannon's diversity index (H) and total species richness were calculated for each site to allow comparison across sites and with previous studies (e.g. Travnichek and Maceina 1994; Freeman et al. 2005). Estimates were generated both with non-native species included and omitted. Additionally the proportion contribution by numbers for each fish family at each site was calculated.

Overall assemblage structure was quantified using several multivariate methods as in Kiraly et al. (2014), all of which were conducted using R statistical software using the *Vegan* and *labdsv* packages (R core team 2020; Oksanen et al. 2020; Roberts 2019). I fourth-root transformed CPE data to account for the extreme skew that was present in these data before using Bray-Curtis dissimilarity to conduct nonmetric multidimensional scaling on the transformed CPE data (NMDS; Goodsell and Connell 2002; Kiraly et al. 2014). Bray-Curtis dissimilarity typically performs better than other measures of dissimilarity for ecological datasets (Kiraly et al. 2014; Orksanen et al. 2020). Function metaMDS in R was used to perform nonmetric multidimensional scaling. Several random starts were used with 50 iterations maximum and final dimensionality was determined by considering stress reduction and interpretability (Kiraly et al. 2014; Oksanen et al. 2020). Kendall's tau correlation coefficient (*T*) was calculated to determine the magnitude and direction of species correlations and MDS axes.

To quantify differences in fish assemblages between sites, I used multiresponse permutation procedure (MRPP) based on the same Bray-Curtis dissimilarity (Kiraly et al. 2014). Only the three seasons (spring: March-May, summer: June-August, fall: September-November) in which sampling occurred at all sites were included. Additionally, hybrids were excluded from multivariate analyses. MRPP generates an *A*-statistic as well as a p-value, both of which must be considered to fully interpret results. The *A* statistic is a measure of effect size and describes within-group homogeneity compared to the random expectation; *A*=1 if all units within groups are identical and *A*=0 if heterogeneity among groups equals the expected value by chance (McCune and Grace 2002). If the null hypothesis is true, the p-value is the likelihood that the observed difference between groups is due to chance (McCune and Grace 2002). Average dissimilarities both between and within groups were calculated to create a dendrogram describing the relationships between groups and to create group blocks. Additionally, MRPP generates a test statistic, δ, which is the overall weighted mean of group mean differences (Oksanen et al. 2020). Permuting δ serves as a hypothesis test of differences between groups of sampling units where p is the probability that δ is less than the observed value. A dendrogram was generated based on the Bray-Curtis dissimilarities where the termination of each branch represented the within-group dissimilarity between seasons while the horizontal lines represented the dissimilarity between site blocks.

Indicator species values (IndVal) were calculated based on the formula given in Dufrene and Legendre (1997) and clarified by Roberts (2019) in the *labdsv* R package for each species given a significant overall MRPP result. This formula calculates the indicator values “d” of species as the product of the relative frequency and relative average abundance in clusters (Roberts 2019) as follows:

$$d_{ic} = f_{ic} * a_{ic}$$

$$f_{ic} = \frac{\sum_{j \in c} P_{ij}}{n_c}$$

$$a_{ic} = \frac{\sum_{j \in c} x_{ij}/n_c}{\sum_{k=1}^{K}(\sum_{j \in c} x_{ij}/n_k)}$$

where: P_{ij} = presence/absence (1/0) of species *i* in sample *j*, x_{ij} = abundance of species *i* in sample *j*, and n_c = number of samples in cluster *c*, for cluster $c \in K$.

IndVal analysis accounts for species site specificity and fidelity and ranges from 0-1 (Dufrene and Legendre 1997). The index equals one when a species is found in all sampling units (seasons) of a group (site). P-values were calculated for each species’ IndVal using a randomization procedure (Roberts 2019). Species were randomly reassigned to sampling units and groups 1000 times and IndVals were recalculated to create a distribution of possible IndVal values from the given data. The p-value was the proportion of randomized IndVals that were greater than the observed value, i.e., the probability that the observed value was due to chance (Dufrene and Legendre 1997).

Data Analysis: Diets.

The weight of each measured diet item was estimated either using published length-weight regressions (i.e., Benke et al. 1999; as in Purcell 2011), or from length-weight regressions that I generated as follows:

$$W = aTL^b$$

where *W* is the diet item weight, *TL* is the length of the diet item, *a* is the intercept, and *b* is the slope. Percent-by-weight of each diet item was then calculated for each target species by season and site (Bowen 1998; Garvey and Chipps 2012),

$$Percent\ by\ Weight = \frac{1}{P}\sum_{j=1}^{P}\left(\frac{W_{ij}}{\sum_{i=1}^{Q} W_{ij}}\right) * 100$$

where *P* is the number of fish with food in their stomachs, W_{ij} is the weight of prey *i* in fish *j*, and *Q* is the number of food categories.

Diet items were classified into the following categories, which contributed 85-100% of diets in all species/site/season combinations: arachnids (spiders, mites), crayfish, non-crayfish crustaceans (amphipods, isopods), fishes, insects, insect larvae, mollusks (snails, bivalves), snakes, worms, zooplankton, and other (plant matter, fish/unidentified eggs, centipedes, millipedes).

Results

Fish Assemblage

Fish species diversity was highest at Wadley (H=2.88) and lowest at Horseshoe Bend (H=2.49), although all values were generally similar across sites (range among sites was 0.39; Table 1.1). The number of species designated as fluvial specialists (by Travnichek and Maceina 1994) ranged from 11 at Horseshoe Bend to 19 in the tailrace, with a decreasing upstream-downstream gradient in the number of fluvial specialists downstream of the dam (Table 1.1). Centrarchids made up the largest proportion of collected fish across all sites with the highest contribution being in the tailrace and at Horseshoe Bend (Figure1.2). Cyprinids/leuciscids comprised the second largest proportion of fish collected from all sites except the tailrace where the second largest proportion was from percids (Figure 1.2). The proportion of the assemblage composed of percids was also relatively high at Wadley, although Lee's Bridge and Horseshoe Bend had much lower values (Figure 1.2). Catastomids made up larger proportions of the collected families at Lee's Bridge and Wadley compared to the other sites and was a very low proportion in the tailrace (Figure 1.2). At Lee's Bridge 3 of the 10 most abundant species were centrarchids and at the downstream sites it ranged from 3 of 10 in the tailrace to 4 of 10 at Horseshoe Bend (Table 1.2).

NMDS yielded stable, two-dimensional ordinations (Figure 3a, final stress = 0.09) in 40 iterations. Ordination axis MDS-1 correlated most positively with Snail Bullhead (T=0.98), Bandfin Shiner (T=0.80), Black Madtom (T=0.80), and Rough Shiner (T=0.79; Figure 4); and most negatively with White Bass (T=-0.96), Yellow Perch (T=-0.92), River

Redhorse (*T*=-0.92), and Bullhead Minnow (*T*=-0.92). Axis MDS-2 correlated most positively with Blueback Herring (*T*=0.63), Skipjack Herring (*T*=0.63), Golden Shiner (*T*=0.63), and Black Bullhead (*T*=0.63); and most negatively with Grass Carp (*T*=-0.62), Pretty Shiner (*T*=-0.62), Snail Bullhead (*T*=-0.34), and Bandfin Shiner (*T*=-0.31; Figure 1.4). Based on the habitat classification for each species, MDS-1 generally correlated positively with fluvial specialist species and negatively with habitat generalist species (Table 1.2, Figure 1.4).

Ordinations showed clear separation between sites as well as seasonal variation in the assemblages within sites (Figure 1.3a). Differences between seasons were driven by generally negative summer and positive spring values for MDS-2 (Figure 1.3a). There was separation across sites in MDS-1 that showed longitudinal variation in fish assemblage downstream of Harris Dam; additionally, with distance downstream, MDS-1 of the fish assemblage approaches that of the upstream “control” site, identifying a potential recovery gradient (Figure 1.3a). No such gradient was present in MDS-2, which was highly variable among sites and seasons (Figure 1.3a).

Multiresponse Permutation Procedure and Indicator Species Analysis

MRPP indicated a significant difference among site-specific fish assemblages (p = 0.001, *A*=0.298, δ=0.290; Figure 1.3b). Indicator species analysis identified 1 - 5 significant indicator species depending on site (Table 1.3). At Lee’s Bridge all five identified indicator species were habitat generalists (Bowfin, Bullhead Minnow, Gizzard Shad, Blue Catfish, Redear Sunfish). At both Wadley and the tailrace three indicator species were identified including one fluvial specialist and two habitat generalists

(Lipstick Darter, Yellow Bullhead, and Bluegill) at the tailrace and two fluvial specialists and one habitat generalist (Speckled Darter, Alabama Hogsucker, and Blacktail Redhorse) at Wadley (Table 1.3). One indicator species was identified at Horseshoe Bend that was a habitat generalist (Black Redhorse; Table 1.3). Average between- and within-group dissimilarities grouped Horseshoe Bend and Wadley as the site block with the least dissimilarity and Lee’s Bridge as the most dissimilar (Figure 1.5). The Tailrace fell within a block containing the downstream sites although separated from Wadley and Horseshoe Bend (Figure 1.5).

Diets

A total of 197 Channel Catfish stomachs were collected, of which 149 contained prey items that could be identified; stomachs with contents that could not be accurately identified due to advanced digestion were excluded. A total of 35 prey types were present in Channel Catfish diets, including adult insects from 10 orders and insect larvae from 5 orders, as well as crayfish, non-crayfish crustaceans, mollusks, fishes, zooplankton, and plant matter.

Channel catfish diets varied across study sites. At Lee’s Bridge, adult insects contributed the largest percentage by weight of Channel Catfish diets in the spring and summer, while larval insects contributed the most in the fall (Figure 1.6). In the tailrace, diet composition varied seasonally, although adult insect contributions remained large in all seasons (Figure 1.6). During summer and winter, Channel Catfish in the tailrace consumed a higher percentage by weight of non-crayfish crustaceans compared to other site/season combinations (Figure 1.6). At Wadley, crayfish contributed a higher percentage by weight of Channel Catfish diets than at other sites, although sample sizes were limited at this site (Figure 1.6). Channel Catfish collected from Horseshoe Bend consumed a high percentage of adult insects during all seasons (Figure 1.6).

A total of 347 Redbreast Sunfish stomachs were collected of which 318 contained prey items that could be identified. I identified 38 unique diet types in Redbreast Sunfish stomachs, including adult insects from 12 orders and insect larvae from four orders, as well as arachnids, crayfish, non-crayfish crustaceans, fishes, mollusks, and zooplankton. Diet composition of Redbreast Sunfish collected from the tailrace were distinct from those collected at other sites and were both seasonally variable and generally characterized by higher contributions from non-crayfish crustaceans (Figure 1.7). Fish collected during the summer were the only Redbreast Sunfish from the tailrace whose diet compositions were similar to those of other downstream sites (Figure 1.7). At Lee’s Bridge, Wadley, and Horseshoe Bend the primary contributors to Redbreast Sunfish diets were adult insects, insect larvae, and mollusks (Figure 1.7).

A total of 448 Alabama Bass diets were collected, of which 312 contained identifiable diet items. A total of 41 diet types were identified in Alabama Bass diets including adult insects from 11 orders and 9 genera of fishes (7 families), and arachnids, crayfish, other crustaceans, insect larvae, snakes, worms, and zooplankton. Fish, crayfish, and adult insects contributed the most to Alabama Bass diets collected from Lee’s Bridge, Wadley, and Horseshoe Bend (Figure 1.8). Adult insects contributed more to Alabama Bass diets collected from the tailrace than from other sites and the tailrace was the only site with any contribution from non-crayfish crustaceans. No single prey type dominated Alabama Bass diets at Horseshoe Bend, with all prey types contributing less than 50% (Figure 1.8). During spring, the contribution of larval insects was higher at all sites; however, at Lee's Bridge and the tailrace there was also a spring increase in contribution of fish (Figure 1.8).

A total of 76 Tallapoosa Bass diets were collected with 60 containing identifiable contents. I identified 21 diet types in Tallapoosa Bass stomachs, including eight insect orders and three fish genera (three families) as well as arachnids, crayfish, insect larvae, snakes, and worms. Insufficient sample sizes limited interpretation of data from Lee's Bridge and the tailrace where the only diet type consumed was crayfish (Figure 9). At the remaining two sites, crayfish, adult insects, and insect larvae were the primary contributors to Tallapoosa Bass diets. At Wadley, the contribution from crayfish was especially high while at Horseshoe Bend there was a seasonal increase in the contribution of adult insects from winter (lowest) to fall (highest) that corresponded with a decrease in the importance of crayfish (Figure 1.9). At Horseshoe Bend and Wadley, spring diet contributions from insect larvae were higher than during other seasons, a pattern also observed in Alabama Bass and Redbreast Sunfish (Figures 1.7,1.8,1.9).

Discussion

Assemblage

Releases of water from dams can strongly affect habitat conditions for fish and other aquatic organisms (Freeman et al. 2005; Young et al. 2011). Impacts that affect fish at the individual scale can also be manifested at the population and assemblage scales. My sampling spanned a longitudinal gradient that included a site above Harris Dam and three sites at increasing distances downstream of the dam, allowing me to examine whether patterns in fish communities are consistent with expected effects of the dam, namely a recovery gradient in the diversity or assemblage composition. Previous studies (see below) have quantified assemblage structure and responses of particular fish populations across this same reach, allowing comparisons that span a range of temporal scales.

Sunfishes and minnows were generally the most common fish families in this part of the Tallapoosa River and variation in diversity from upstream to downstream was neither dramatic nor systematic. Catostomids, centrarchids, and cyprinids were dominant in catches above Harris Dam, similar to the findings of Travnichek and Maceina (1994) who conducted a survey prior to the implementation of the water release management Green Plan in 2005 of the broader Tallapoosa River from the upper Tallapoosa near Heflin, AL to the Coastal Plain. Overall species diversity index values for this study area were slightly higher and more variable in 1994 (1.98 - 3.53) compared to my study (2.49 - 2.88), though this difference may have been driven in part by differences in sampling techniques. Additionally, Travnichek and Maceina (1994) split their sampling into deep and shallow water categories and reported the values for each; within the comparable sites, my observed values of H were always higher than those reported for shallow water species (2.05-2.27) and lower than those reported for deep water species (3.19-3.53). This suggests that my sampling procedures may have provided a compromise across depths. Trends in fish diversity upstream to downstream were similar between my findings and those of Travnichek and Maceina (1994), who found some evidence that river regulation diminished the number of obligate fluvial specialist species in the Tallapoosa River. It is important to note that centrarchids were not historically dominant in the Tallapoosa River (Irwin and Hornsby 1997). Considering that catch rates of centrarchids in both my study and in Travnichek and Maceina (1994) were high downstream of Harris Dam, this supports the idea that generalist species (such as many centrarchids) may be less affected by river regulation (Scott 1951, Swingle 1953; Kinsolving and Bain 1993; Travnichek and Maceina 1994). Travnichek and Maceina (1994) also observed an increase in species richness from upstream to downstream across a much broader study area. Their results from within the bounds of this study area did not yield such a trend, consistent with my study.

Irwin and Hornsby (1997) compared rotenone surveys conducted at Horseshoe Bend in 1951 (pre-Harris Dam) versus 1996 (post-Harris Dam) to assess the effects of river regulation due to Harris Dam on downstream fish assemblages. Differences in species composition between these rotenone studies suggested that pre- versus post-dam the fish assemblage at Horseshoe Bend had shifted from one dominated by cyprinids and ictalurids to an assemblage dominated by centrarchids (Irwin and Hornsby 1997). My findings found the relative proportion by numbers of centrarchids to be larger than the 1951 rotenone sample, but lower than the 1996 sample. The proportion of cyprinids and catostomids in my sample was higher than in the 1996 rotenone sample but similar to the 1951 findings (Irwin and Hornsby 1997). Unfortunately, some of these trends likely resulted from differences in sampling method (electrofishing versus rotenone), sampling frequency (bimonthly versus a single sample), and seasonality. The continued prevalence of centrarchids in the Tallapoosa River downstream of Harris

Dam in my study, Travnichek and Maceina (1994), and Irwin and Hornsby (1997) suggests that Harris Dam has and continues to affect the downstream fish assemblage.

The presence of significant differences in fish assemblage composition across sites in ordinal space and the formation of upstream versus downstream site blocks on the Tallapoosa River suggests that regulation by Harris Dam is responsible for much of the variation in fish assemblage structure. Although riverine fish assemblages naturally vary longitudinally, regulation of the Tallapoosa River by Harris Dam changes the quality of habitat by altering temperatures, limiting productivity, and fragmenting the river, changing natural patterns of movement, persistence, and colonization (Vannote et al. 1980; Irwin and Freeman 2002; Kiraly et al. 2014, Irwin et al. 2019). My study describes patterns that are influenced by Harris Dam and the resulting flow regulation. The first multidimensional scaling axis (MDS-1) correlated strongly with several species that are fluvial specialist (e.g., Snail Bullhead, Bandfin Shiner, Black Madtom, Rough Shiner), and interestingly, the tailrace had the highest scores of any site for MDS-1. This suggests that the tailrace is primarily occupied by specialists that are able to thrive in the variable flow conditions, potentially at the expense of other native species including minnows and suckers. The variable classifications (FS or HG) of the indicator species identified for each site further describes the myriad differences throughout the study area. For example, the three indicator species identified for the tailrace include a fluvial specialist percid and two habitat generalist centrarchids, further supporting the conclusion that this site is not compatible for some native fishes (especially suckers). Meanwhile the indicator species at the other downstream sites were primarily fluvial specialists including several native catastomids.

The MDS analysis demonstrated both spatial and temporal variation in fish assemblage structure throughout the mainstem Tallapoosa River. Several previous studies did not include seasonal variation when quantifying the Tallapoosa fish assemblage (Travnichek and Maceina 1994; Irwin et al. 2019). My results indicate that conclusions drawn from Travnichek and Maceina (1994) and Irwin et al. (2019) need to be limited to the time scale they encompassed. My inability to sample at Lee’s Bridge in winter unfortunately hampers even broader generalization given that a full model of the annual variation in fish assemblage might identify additional patterns.

It is difficult to determine exactly how much Harris Dam has affected the fish assemblage in the Tallapoosa River given the lack of literature predating its construction. Longitudinal variation in fish assemblage structure is inevitable in river systems as energy and nutrient inputs change (Vannote et al. 1980). Overinterpreting the single rotenone sample taken before Harris Dam began operation is also questionable given the habitat heterogeneity of the Tallapoosa River and the limited spatial and temporal coverage of rotenone sampling (Swingle 1953). Outside of this study area, literature indicates that flow regulation, and especially peaking flow, negatively impacts fish recruitment and spawning (Weyers et al. 2003; Rolls et al. 2013). The persistent regulation of the Tallapoosa River by Harris Dam likely resulted in a dramatic change in the fish assemblage driven by the inability of certain species to adapt their spawning and feeding habits to the rapid temperature and flow fluctuation (Rolls et al. 2013).

Diets

Spatial and temporal variation in fish diets were present for all species, and diet composition in the tailrace were particularly unique. During at least one season, all target species with a suitable sample size in the tailrace showed a distinct increase in the contribution from non-crayfish crustaceans. This pattern was particularly clear in Channel Catfish and Redbreast Sunfish diets and may result from two possible explanations. First, the base of the food web could be less diverse in the tailrace, affording fish fewer choices in prey types. Second, these prey types may have only been present in the tailrace and did not have established populations elsewhere. Additionally, Alabama Bass, the most piscivorous of the four target species, consumed far less fish in the tailrace compared to other sites, which might also point to a restricted food web that limits the productivity of the tailrace site.

Fencl et al. (2015) found substrate size immediately downstream of dams to be coarser than in the upstream reaches, potentially impacting invertebrate communities. This could help explain some of the unique patterns in diet composition observed in the Harris Dam tailrace. Additionally, Coffman et al. (2019) observed differences in the assemblage composition of aquatic invertebrates between regulated and unregulated portions of the Tallapoosa River and noted that invertebrate species richness was lower at regulated sites compared to unregulated sites. While not all diet types observed from fish in the tailrace were aquatic invertebrates, they contributed important portions of diets of all target species during every season. Additionally, results from my assemblage analysis indicated that the tailrace had proportionally fewer minnows, sucker, and shad compared to the other sites and many of the other species present are generally cryptic (e.g., darters) or have better defenses (e.g. madtoms), potentially limiting their vulnerability.

Overall, studies of diet composition of riverine sportfish are lacking compared to those conducted in lentic environments, and this is particularly true in the Southeastern U.S. where several new black bass species have been described in the last two decades (Baker et al. 2013). In general, my findings concur with published literature regarding the primary components of target species diet composition (Johnson and Johnson 1984; Johnson and Dropkin 1993; Johnson and Dropkin 1995; Scott and Angermeier 1998; Wheeler and Allen 2003; Weisberg and Janicki 2011; Sammons 2012; Helms et al. 2018).

Studies of lotic Channel Catfish populations in comparable river systems are extremely limited. Braun and Phelps (2016) reported that Channel Catfish from the Middle Mississippi River, Missouri consumed primarily plant matter, fish, crayfish, and insects, but the proportions of insects were much lower and the proportions of plant matter and fishes in diets much higher than those observed in my study. In the Susquehanna River, Maryland, Weisberg and Janicki (2011) observed higher rates of insect consumption by Channel Catfish compared to my study. Unfortunately much of the literature concerning Channel Catfish diet compositions is directed toward the aquaculture industry or lentic populations, making those studies unsuitable for comparison.

Helms et al. (2018) observed similar diet items to those in my results from Redbreast Sunfish collected from the Chattahoochee River, Georgia, including insects, zooplankton, crayfish, arachnids, and mollusks. They also found that across sample sites aquatic and terrestrial insects contributed most to Redbreast Sunfish diets, and that the number of invertebrate taxa collected from a stream was strongly correlated with the number found in Redbreast Sunfish stomach contents. This supports the earlier hypothesis that a limited forage base may relate to the unique diet patterns observed in the tailrace of Harris Dam tailrace. Other studies have indicated that Redbreast Sunfish adults feed heavily on insects and insect larvae while juveniles consume higher percentages of zooplankton (Johnson and Johnson 1984; Johnson and Dropkin 1993; Johnson and Dropkin 1995).

Scott and Angermeier (1998) found that in the New River, Virginia, Spotted Bass and Smallmouth Bass consumed high proportions of crayfish and insects as juveniles. In addition, while Smallmouth Bass underwent an ontogenetic shift to consume fish, insects, and crayfish, Spotted Bass (likely *M. punctatus*) shifted to consume crayfish, fish, and insects (listed in decreasing order of contribution percentage). Riverine black basses in the Flint River, Georgia have been better studied than those in the Tallapoosa River, and previous studies showed that introduced Alabama Bass consumed more crayfish and insects than did native Shoal Bass or Largemouth Bass (Sammons 2012). Similarly, Wheeler and Allen (2003) reported that Largemouth Bass and Shoal Bass in the Chipola River, Florida, consumed high proportions of crayfish and fish as adults, but consumed aquatic insects as juveniles. Parsons (1954) gave a brief, non-quantitative description of the diet composition of Redeye Bass from the Conasauga River (now

Micropterus coosae) stating that “Insects taken from the surface of the water appear to be the most common food.”

Overall the unique diet composition present in several species in the Harris Dam tailrace indicates that that the operation of Harris Dam and the subsequent downstream habitat modification affected fish feeding habits and diet composition. My study successfully quantified the diet composition of four sportfish in the Tallapoosa River in order to serve as a comparison for future studies, to inform bioenergetic modelling, and to equip managers with information about the effects of river regulation by the operation of Harris Dam. As Harris Dam continues to undergo management changes through the ongoing relicensing process, the Green Plan, or other stakeholder action, it will be important to continue cataloguing elements of fish life history to quantify the effects of these changes.

The effects of Harris Dam on the downstream fish communities have been well documented over the past 30 years (Travnichek and Maceina 1994; Irwin and Hornsby 1997; Freeman et al. 2005; Irwin et al. 2019); however, the lack of information about the status of fish communities in the Tallapoosa River prior to the construction of Harris Dam continues to limit such studies. Here, I showed that the fish assemblage in the tailrace of Harris Dam is restricted by the presence/operation of the dam and dominated by fluvial specialist species. Additionally, fish collected from the tailrace preyed on diet items at different proportions compared to other sections of the river and several prey items were only found in diets collected from the tailrace. Taken together, it is clear that Harris Dam continues to affect many aspects of fish ecology in the downstream area.

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Table 1.1. Total number of fish species, families, Shannon’s H diversity index values, and number of fluvial specialist (FS) species collected from four sites on the Tallapoosa River, Alabama. Fluvial specialists were defined by Travnichek and Maceina (1994) and listed in Table 1.2. All estimates were calculated excluding non-native species; the total number of species and values of Shannon’s H calculated by excluding non-native species are in parentheses.

Site	Total Species	Total Families	Shannon's H	# FS Species
LB	39 (37)	9	2.80 (2.78)	14
TR	38 (38)	7	2.59 (2.59)	19
WD	35 (35)	7	2.88 (2.88)	17
HB	33 (31)	7	2.49 (2.54)	11
All	55 (51)	9	3.06 (3.05)	24

Table 1.2. Scientific names, common names, species abbreviations, site distributions, and classifications used in this report. T&M 1994 refers to the classifications designated by Travnichek and Maceina (1994) and abbreviations are defined in Table 1.1. The ten most abundant species at each site are identified by superscripts (most abundant =1). Sites are LB=Lee's Bridge, TR=Harris tailrace, WD=Wadley, HB=Horseshoe Bend.

Scientific Name	Common Name	Abbreviation	T&M 1994	LB	TR	WD	HB
Amia calva	Bowfin	BOWF	HG	X			
Alosa aestivalis	Blueback Herring	BBHR	HG				X
Alosa chrysochloris	Skipjack Herring	SKJH	HG				X
Dorosoma cepedianum	Gizzard Shad	GIZS	HG	X^{6}		X	X
Dorosoma petenense	Threadfin Shad	THSH	HG	X		X	X^{10}
Campostoma oligolepis	Largescale Stoneroller	LSSR	FS	X	X^{7}	X^{10}	
Cyprinella callistia	Alabama Shiner	ALSH	FS		X^{3}	X^{9}	X
Cyprinella gibbsi	Tallapoosa Shiner	TPSH	FS	X	X	X	
Cyprinella venusta	Blacktail Shiner	BTSH	HG	X^{5}	X	X^{4}	X^{4}
Cyprinus carpio	Common Carp	CCAR	HG	X^{9}	X	X	X
Luxilus chrysocephalus	Striped Shiner	STSH	FS		X	X	
Luxilus zonistius	Bandfin Shiner	BAFS	FS		X		
Lythrurus bellus	Pretty Shiner	PRSH	FS	X			
Notemigonus crysoleucas	Golden Shiner	GLDA	HG				X
Ctenopharyngodon idella	Grass Carp	GCAR	HG	X			
Notropis baileyi	Rough Shiner	RSHN	FS		X		
Notropis stilbius	Silverstripe Shinner	SPSH	FS	X		X^{8}	X^{3}
Notropis texanus	Weed Shiner	WESH	HG	X	X		
Notropis xaenocephalus	Coosa Shiner	COOS	FS	X	X	X	X
Pimephales vigilax	Bullhead Minnow	BUMN	HG	X^{7}			
Semotilus thoreauianus	Dixie Chub	DXCB	FS		X		
Hypentelium nigricans	Alabama Hogsucker	AHOG	FS	X	X	X^{7}	X
Minytrema melanops	Spotted Sucker	SPSR	HG	X	X	X	X

Moxostoma carinatum	River Redhorse	RVRH	FS	X			
Moxostoma duquesnei	Black Redhorse	BREH	FS	X		X	X[9]
Moxostoma poecilurum	Blacktail Redhorse	BTRH	HG	X[1]	X	X[2]	X[5]
Ameiurus brunneus	Snail Bullhead	SNBL	FS		X		
Ameiurus melas	Black Bullhead	BLBH	HG				X
Ameiurus natalis	Yellow Bullhead	YBUL	HG	X	X[10]	X	X
Ameiurus nebulosus	Brown Bullhead	BRBH	HG		X	X	
Ictalurus furcatus	Blue Catfish	BCAT	HG	X[10]			X
Ictalurus punctatus	Channel Catfish	CCAT	HG	X[3]	X[9]	X	X[7]
Noturus funebris	Black Madtom	BLMT	FS		X		
Noturus leptacanthus	Speckled Madtom	SPMT	FS			X	
Pylodictis olivaris	Flathead Catfish	FCAT	HG	X	X		X
Fundulus olicaceus	Blackspotted Topminnow	BLTM	HG	X	X	X	X
Morone chrysops	White Bass	WHBA	HG	X			
Morone saxatilis	Striped Bass	STBA	HG	X	X		
Ambloplites ariommus	Shadow Bass	SHBA	FS	X	X[4]	X	X
Lepomis auritus	Redbreast Sunfish	RBSF	HG	X	X	X[3]	X[2]
Lepomis cyanellus	Green Sunfish	GSUN	HG		X	X	X
Lepomis gulosus	Warmouth	WARM	HG	X	X	X	X
Lepomis macrochirus	Bluegill	BLGL	HG	X[2]	X[1]	X[6]	X[6]
Lepomis microlophus	Redear Sunfish	REAR	HG	X[8]	X	X	X
Lepomis spp.	Bluegill X Green Sunfish	BGGN	HG		X	X	X
Lepomis spp.	Hybrid Redbreast	RBSX	HG			X	X
Micropterus henshalli	Alabama Bass	ALAB	HG	X[4]	X[6]	X[1]	X[1]
Micropterus salmoides	Largemouth Bass	LGMB	HG	X	X	X	X
Micropterus tallapoosae	Tallapoosa Bass	TPBA	FS	X	X	X	X[8]
Pomoxis annularis	White Crappie	WHCP	HG	X	X	X	
Pomoxis nigromaculatus	Black Crappie	BLCP	HG	X	X	X	X

Etheostoma chuckwachatte	Lipstick Darter	LIPD	FS		X[5]	X	X
Etheostoma stigmaeum	Speckled Darter	SPDT	FS	X	X	X	
Etheostoma tallapoosae	Tallapoosa Darter	TPDA	FS		X	X	
Perca flavescens	Yellow Perch	YPER	HG	X			
Percina kathae	Mobile Logperch	MLOG	FS	X	X	X	X
Percina palmaris	Bronze Darter	BRDT	FS	X	X[2]	X[5]	X
Percina smithvanizi	Muscadine Darter	MBDT	FS	X	X[8]	X	X

Table 1.3: Indicator species values and p-values for sites on the Tallapoosa River, Alabama. Abbreviations as defined in Table 1.2.

Site	Species	Indicator Value	P-value
LB	BOWF	1.0000	0.015
LB	BUMN	1.0000	0.023
LB	GIZS	0.7027	0.015
LB	BCAT	0.6636	0.024
LB	REAR	0.4619	0.023
TR	LIPD	0.5940	0.015
TR	YBUL	0.5402	0.031
TR	BLGL	0.3124	0.043
WD	SPDT	0.6591	0.032
WD	AHOG	0.4152	0.016
WD	BTRH	0.3635	0.016
HB	BREH	0.4019	0.045

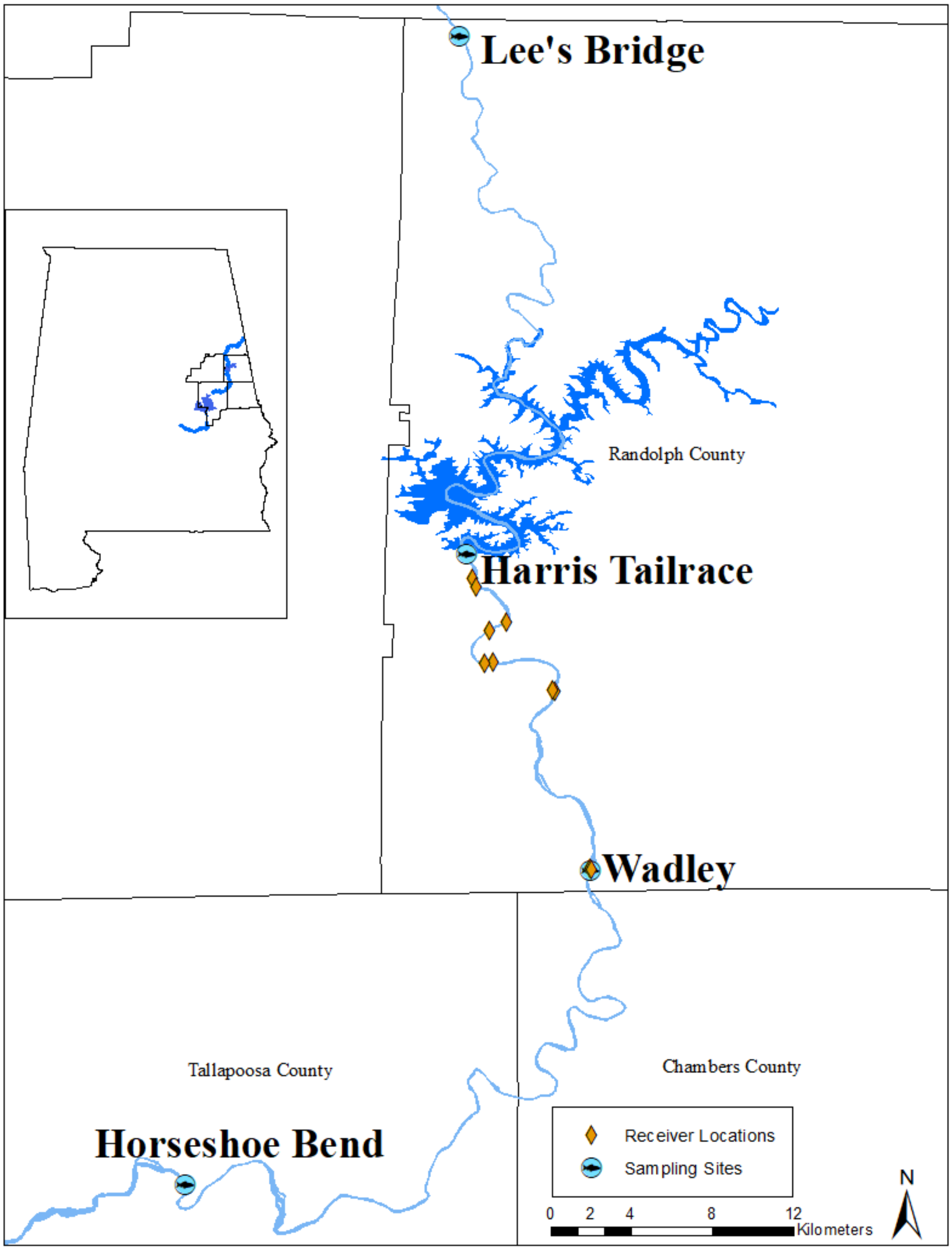

Figure 1.1: A map of the study area on the Tallapoosa River, AL. Sampling locations are indicated with a blue circle surrounding a fish and acoustic receiver locations are indicated with orange diamonds.

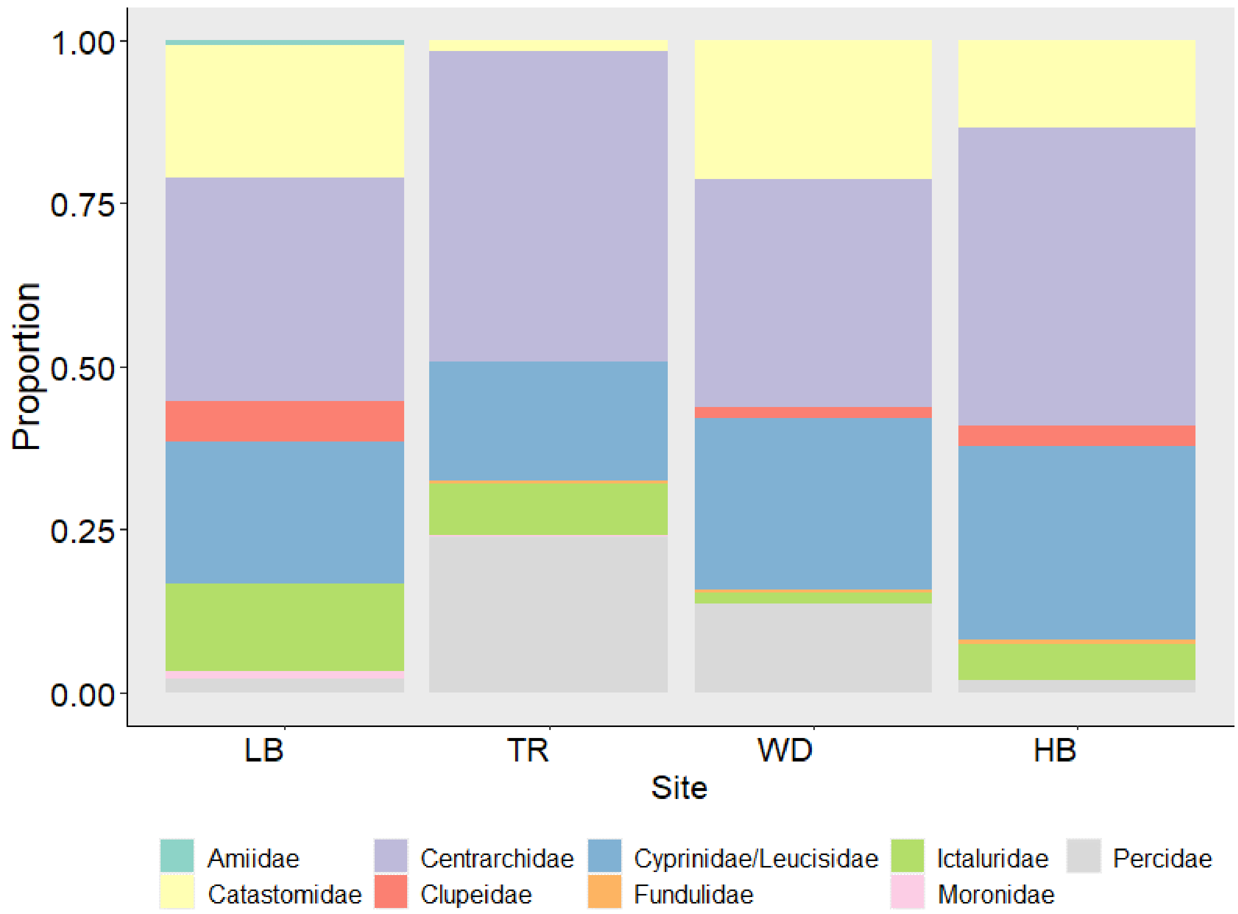

Figure 1.2: Proportion of the sampled fish assemblage belonging to each family collected from four sites on the Tallapoosa River, AL.

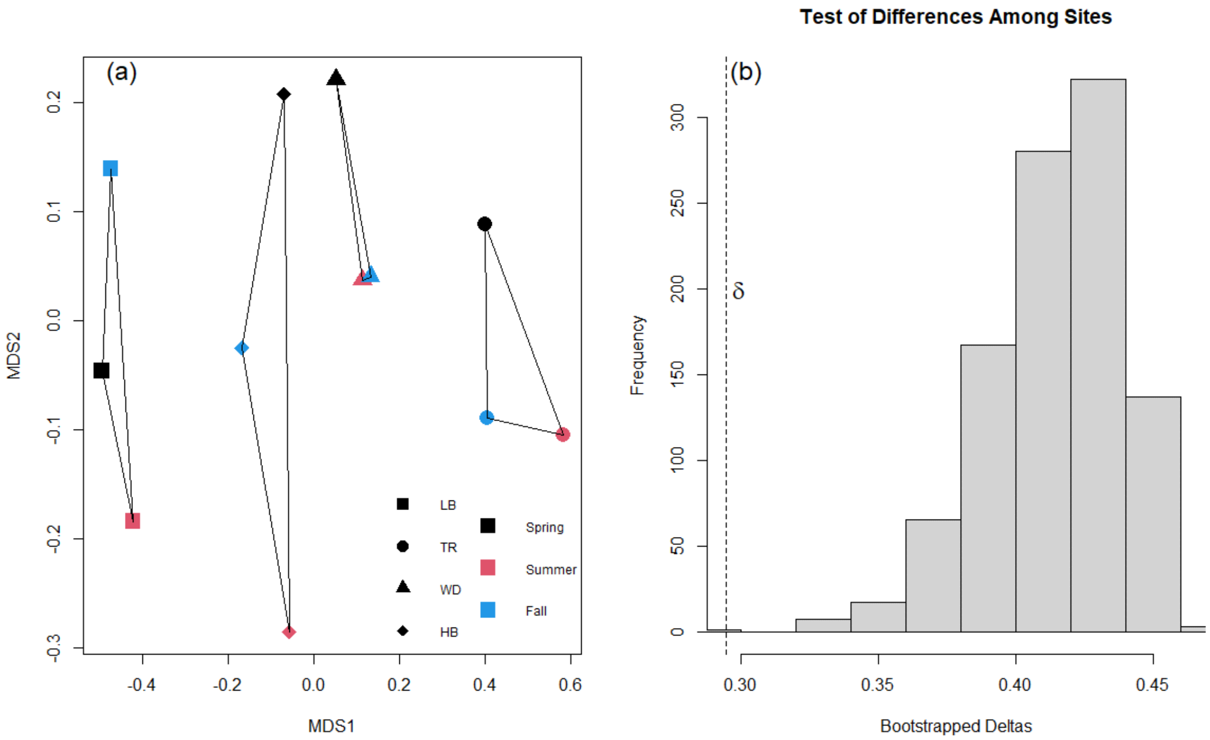

Figure 1.3: (a) Multidimensional scaling ordinations of CPE by site and season for fishes captured from four sites in the Tallapoosa River, AL. Sites are: LB=Lees Bridge, TR=tailrace, WD=Wadley, HB=Horseshoe Bend. (b) The distribution of permuted delta values from a multiresponse permutation procedure. The dashed line represents the observed delta.

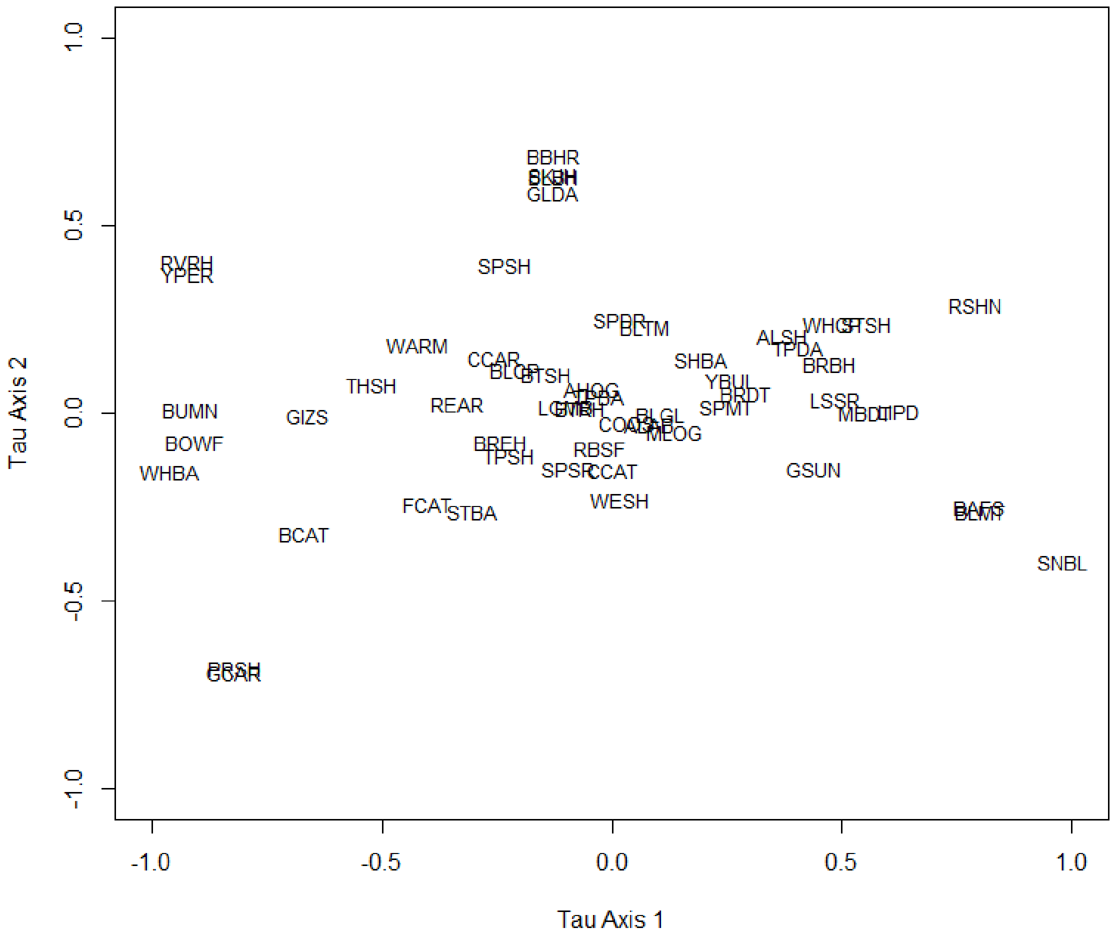

Figure 1.4: Kendall’s tau correlations with nonmetric multidimensional scaling ordination axes for individual species CPE data from the Tallapoosa River upstream and downstream of Harris Dam. Abbreviations as defined in Table 1.2.

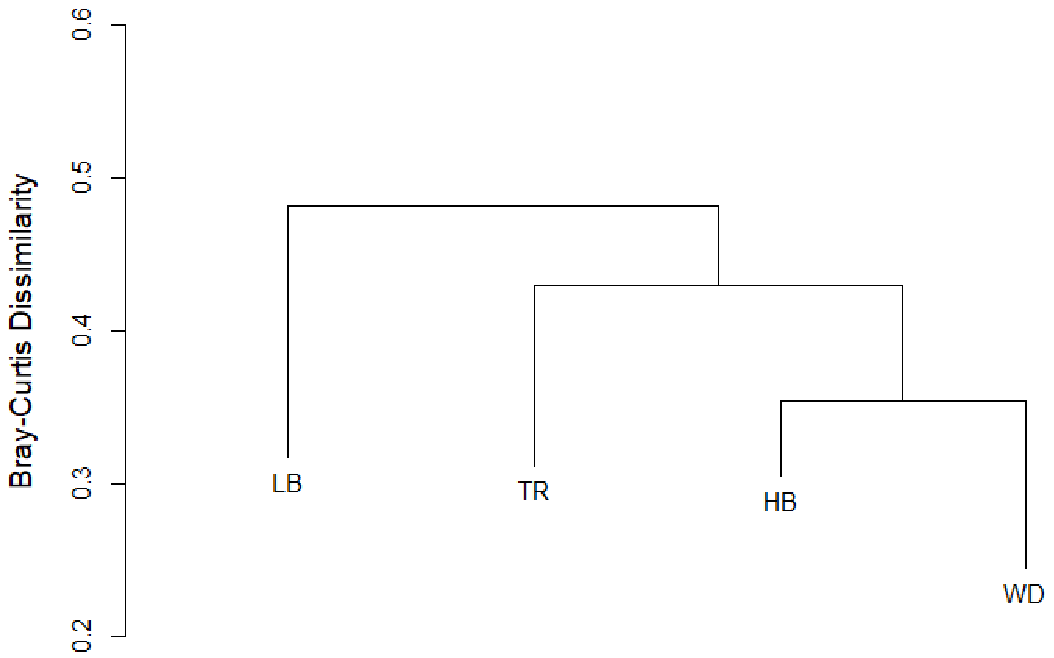

Figure 1.5: Site clusters based on between and within site dissimilarity for sites on the Tallapoosa River, AL. Branches hang to within group dissimilarity while horizontal lines are positioned at among group dissimilarities.

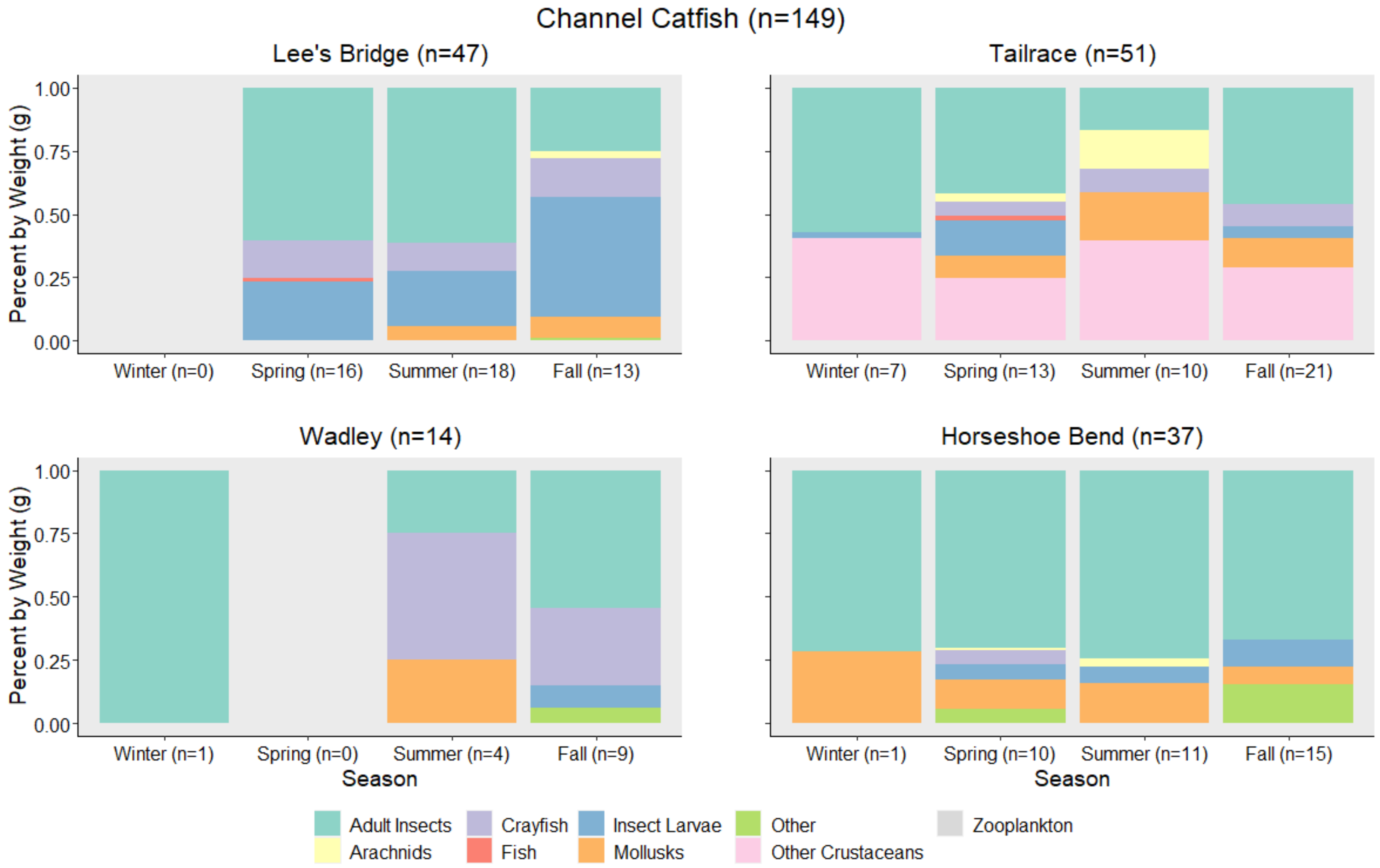

Figure 1.6: Diet composition in percent by weight for Channel Catfish collected from four sites on the Tallapoosa River during 2019-2021.

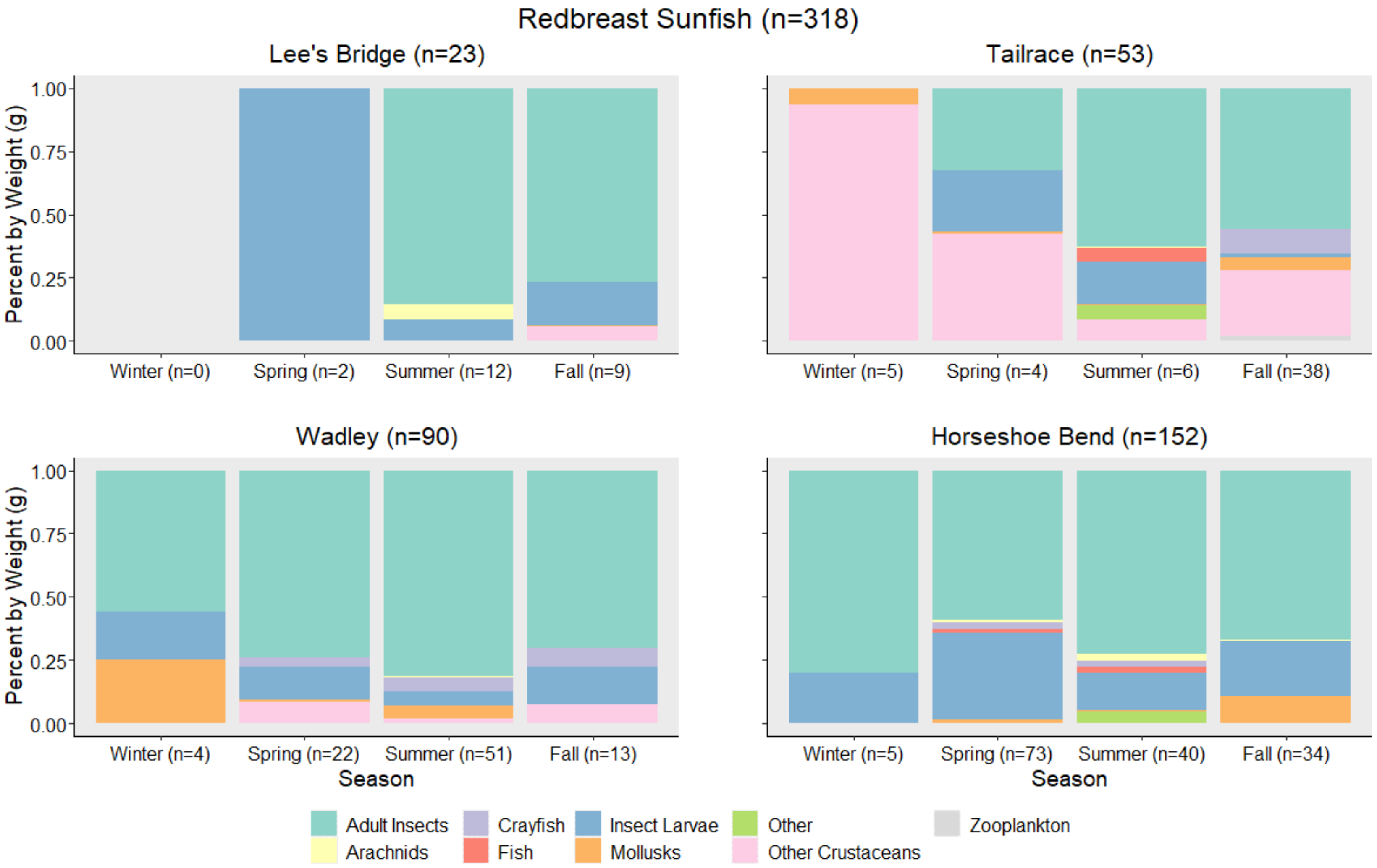

Figure 1.7: Diet composition in percent by weight for Redbreast Sunfish collected from four sites on the Tallapoosa River during 2019-2021.

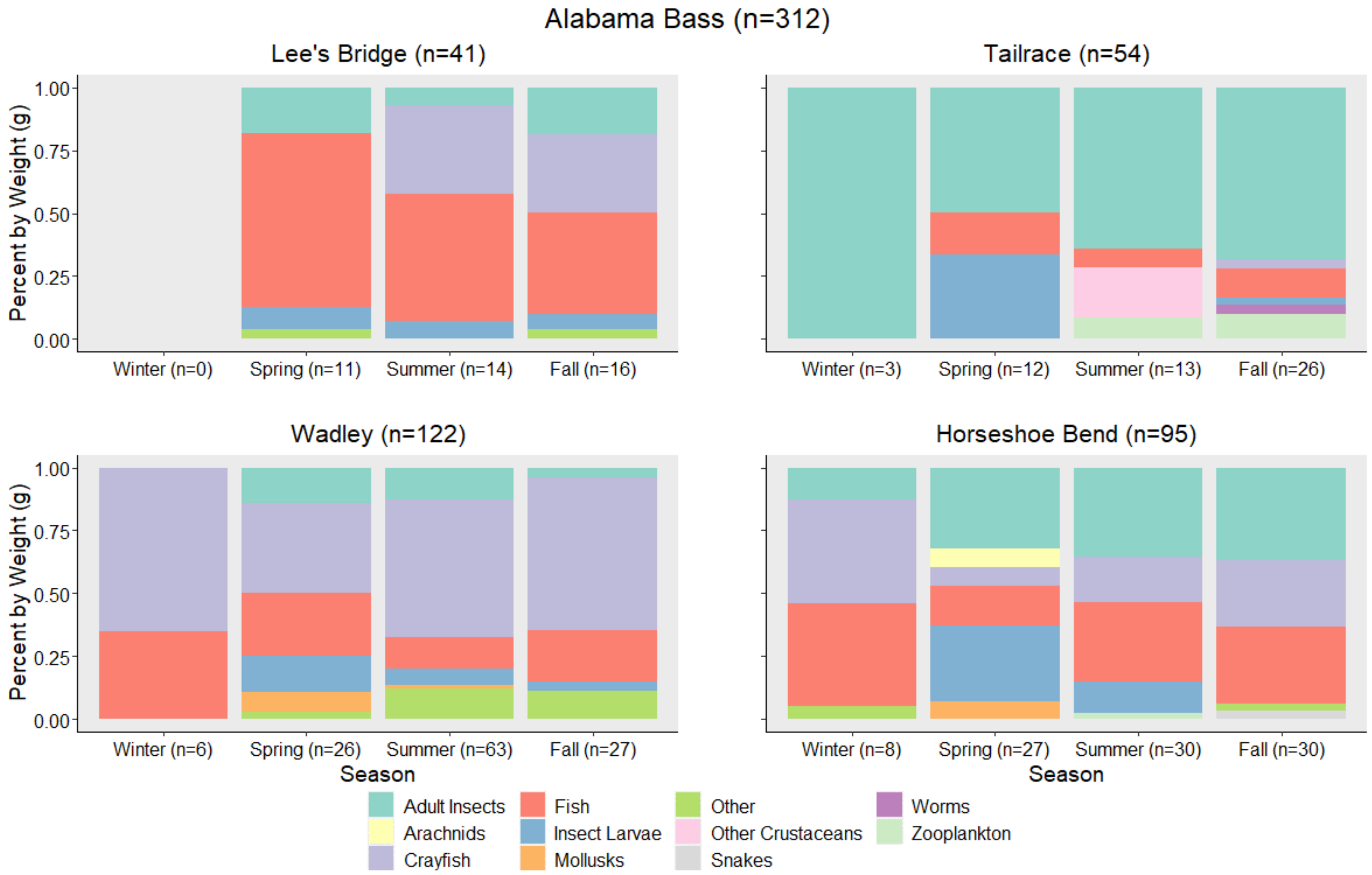

Figure 1.8: Diet composition in percent by weight for Alabama Bass collected from four sites on the Tallapoosa River during 2019-2021.

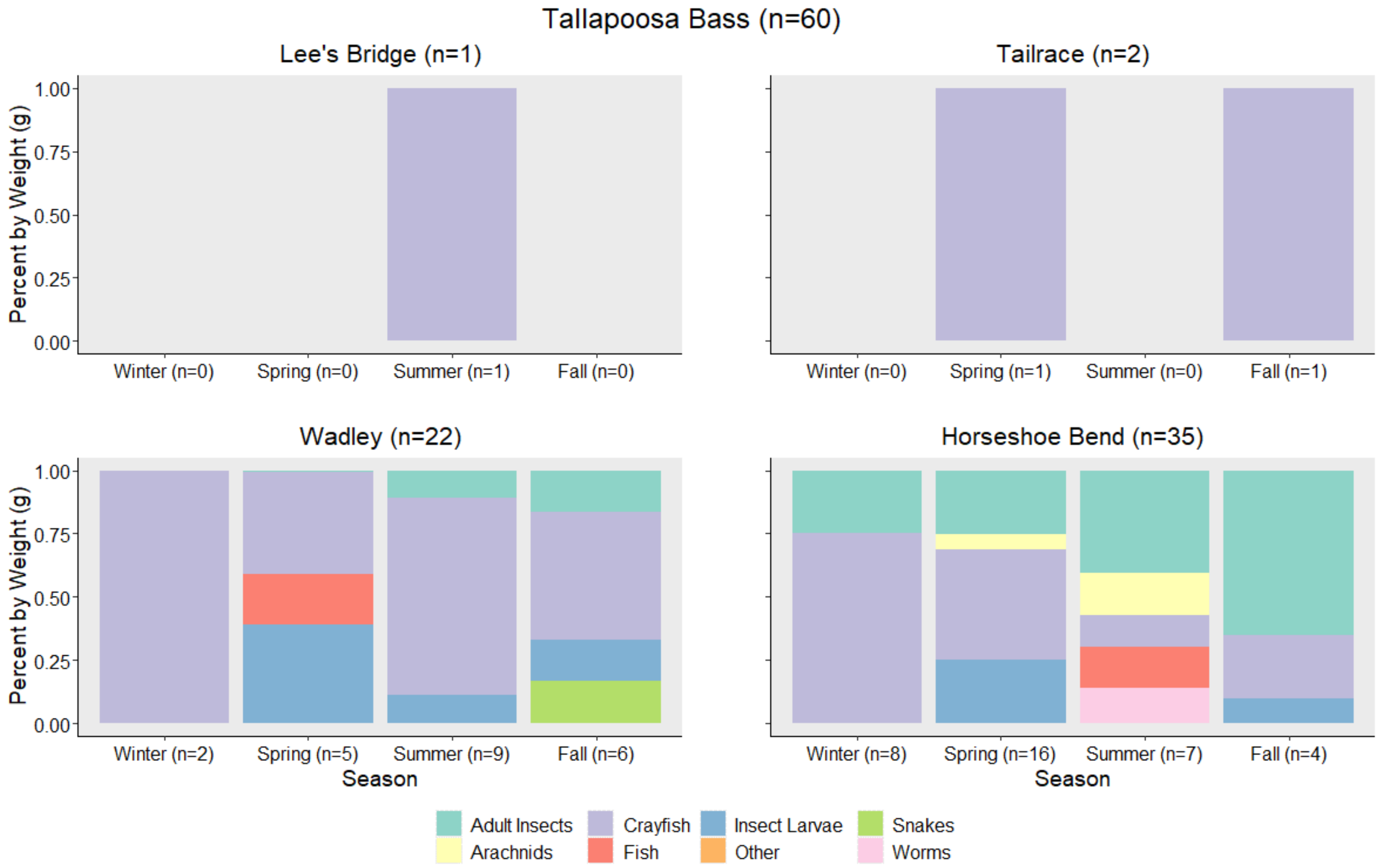

Figure 1.9: Diet composition in percent by weight for Tallapoosa Bass collected from four sites on the Tallapoosa River during 2019-2021.

Chapter 2: Assessing movement of sport fishes in the tailrace of a peaking hydropower dam: a mixed approach using natural tags and telemetry.

Introduction

High velocity, peaking releases of water from hydropower dams create changes in downstream fish habitats and have the potential to significantly affect fish movement. Altered fish movement can negatively affect fish growth, foraging, survival, and recruitment as movement allows fish to respond to environmental stressors (Kahler et al. 2001; Earley and Sammons 2015). Recent work has improved our knowledge regarding black bass movement in Southeastern US rivers, though studies relied heavily on radio telemetry and active tracking (Goclowski et al. 2010; Knight et el. 2011; Earley and Sammons 2015; Sammons and Earley 2015; Cottrell et al. 2018). In a peaking hydropower system where high flows can create unsafe river conditions; passive receivers may provide additional insight into fish movement beyond those times when lower flows allow active tracking (Cooke et al. 2008; Crossin et al. 2017; Hershey et al. 2021). Additionally, given that the temporal scale of telemetry studies is generally months to years, natural tag approaches, such as otolith trace element analyses, can provide insight into movements across a broader time scale as they typically follow the entire lifespan of the fish in question.

Trace elements in hard structures (otoliths) typically reflect the ambient water chemistry; as such, it is possible to follow changes in water chemistry that a fish experiences throughout its life (Campana et al. 1997; Kennedy et al. 2002; Farmer et al. 2013; Zeigler and Whitledge 2011; Kratina 2019). Otoliths are calcified inner ear structures present in fishes and comprise a calcium carbonate and protein matrix. Otoliths grow as the fish age from deposition of calcium carbonate which is incorporated daily into concentric rings. During this process, trace elements from the ambient water can be incorporated into otoliths sometimes replacing Ca^{++} in the matrix. Once these materials are deposited, the elemental concentrations remain temporally stable because the otolith is not reworked (Campana 1999, Campana and Thorrold 2001). In previous studies, analyses of trace elements in various hard parts have been used to identify natal origins (Zeigler and Whitledge 2011; Phelps et al. 2012), quantify movement and migration patterns (Zeigler and Whitledge 2011; Farmer et al. 2013), and describe specific life-history patterns (Helms et al. 2018). In these types of studies, gradients in water chemistry are generally created by salinity differences (Farmer et al. 2013; Chrisp 2021), differences in site types (e.g. river, tributary, lake; Zeigler and Whitledge 2011) or large spatial scales (Bronte et al. 1996; Wells et al. 2003; Kratina 2019).

Harris Dam is the first impoundment on the Tallapoosa River in Alabama, it was constructed in 1983 and has been subjected to release regulations under an adaptive management plan (the Green Plan) since 2005 in an attempt to mitigate the effects of hypolimnetic release. However, pulses of water are still released from the hypolimnion and resulting in dramatic variation in both discharge and water temperature in the tailrace which, combined with the release of cold, potentially anoxic hypolimnetic water, may result in unique trace element signatures. These signatures may vary with distance downstream of the dam, providing a gradient that could allow me to quantify general fish location or movement patterns using natural tags.

By sampling different regions on fish otoliths, each of which can be associated with temporal landmarks for that fish, I attempted to reconstruct information about fish habitat conditions during various periods of their life (Campana and Thorrold 2001; Kennedy et al. 2002, Kratina 2019). My objectives were: 1) to test the feasibility of using this otolith trace element analyses in the Tallapoosa River by classifying fish back to their capture location based on elemental signatures in otoliths and 2) if successful, identify trends in recruitment and potential movement patterns.

Although natural tags provide insight into broad-scale habitat use throughout a fish’s life, tracking individuals using acoustic and radio telemetry can provide shorter-term movement, migration, and dispersal information. Advances in telemetry technologies have made it a useful tool for studying movement and migration patterns (Cooke et al. 2008; Crossin et al. 2017; Hershey et al. 2021), as well as home range size and habitat use (Knight et al. 2011). However, little telemetry data are available on black bass species such as the Alabama Bass *Micropterus henshalli* and the Tallapoosa Bass *Micropterus tallapoosae* in lotic environments. To date, the completed studies with this group of species rely entirely on active tracking of tagged fish using a handheld antenna and receiver (Goclowski et al. 2010; Knight et el. 2011; Earley and Sammons 2015; Cottrell et al. 2018). High velocity water pulses are of concern given their potential effects on fish movement and the capacity for pulses to displace fish downstream. By deploying acoustic receivers at specific points downstream of the dam, I sought to document movement along the river both due to routine movements and those related to water releases using both acoustic and manual tracking (Heupel et al. 2006; Kessel et al. 2014; Crossin et al. 2017). This approach was selected to quantify the effects of peaking hydropower on the longitudinal movements of fish in Harris Dam tailrace.

Methods

Water chemistry

Water samples were collected every other month with a Van Dorn bottle. The sampler was submerged open at 1 m for 30 sec to flush any residual water before being closed and the sample collected (Farmer et al. 2013; Kratina 2019). Water was taken from the Van Dorn bottle using a 50-mL sterile syringe and filtered through a disposable 0.45-µm PTFE glass-fiber filter (Whatman GD/XP) directly into a 125-mL acid washed bottle containing 0.5-mL of nitric acid preservative (Lowe et al. 2011; Farmer et al. 2013; Kratina 2019). This process was repeated until the 125-mL bottle was filled, after which the bottle was sealed, placed on ice, and then refrigerated upon return to Auburn University's Ireland Center Laboratory.

Water samples were analyzed for elemental composition at the Auburn Geosciences Department using an Agilent Technologies 7900 Quadrupole ICPMS and MassHunter software (Agilent Technologies). Analysis was performed using solution-based ICPMS, for a suite of 26 different elements using a surface water standard (NIST 1640a) to correct for equipment and matrix drift effects for individual samples. Water data were recorded in ppb for ease of conversion to molar element: calcium ratios. Separate ANOVAs were used to test differences in water elemental compositions between seasons (replicated across sites) and sites (replicated across seasons) and Tukey's HSD post-hoc test was used to determine which sites exhibited distinct signatures. In instances where element: calcium ratios varied seasonally, a two-way ANOVA with an interaction term was used to determine if the differences between sites were dependent on season and vice-versa.

Fish collection

Fish were collected by boat electrofishing (Midwest Lake Management, Inc. Missouri, USA) once every other month, with sampling at each site consisting of six, 600-sec transects; a total of 12 bimonthly sampling events took place over the duration of this study. Three sites were located on the mainstem Tallapoosa River below Harris Dam (tailrace, Wadley, Horseshoe Bend), and an additional site upstream of Harris Reservoir served as an unregulated reference site (Lee’s Bridge). Output voltage was standardized between 700-900 volts with 100-120 pulses per second, and GPS coordinates were recorded at the start and end of each transect. A floating barge electrofisher was used at the tailrace site given that it is inaccessible by a regular boat; barge sampling consisted of one individual with the anode and 1-2 dip-netters wading alongside, with another individual pushing the barge itself. Barge electrofishing followed the same procedures as boat electofishing, although a lower voltage (500-700 volts) was used for safety. Target species were four recreationally important species and were selected to encompass a range of life history, diet, and habitat use/strategies: Channel Catfish *Ictalurus punctatus* (demersal, omnivore), Redbreast Sunfish *Lepomis auritus* (demersal, insectivore), Alabama Bass *Micropterus henshalli* (benthopelagic, piscivore), and Tallapoosa Bass *Micropterus tallapoosae* (benthopelagic, invertivore).

Otolith preparation

Sagittal otoliths (lapillar for ictalurids) were extracted from all target species. Otoliths were set in epoxy, sectioned with an Isomet diamond wheel low-speed saw to 0.6 mm and mounted on glass slides using thermoplastic cement. Otoliths were polished using 300-1400 grit lapping film until smooth. Next, sections were rinsed with ultrapure water and mounted on petrographic slides before being placed in a covered petri dish filled with ultrapure water. The dish was then floated in a bath within an ultrasonic cleaner and sonicated for 10 min. Following sonication, otolith sections were rinsed again with ultrapure water and allowed to air dry (Zeigler and Whitledge 2011, Lowe et al. 2011, Farmer et al. 2013, Kratina 2019). Otolith elemental analyses were conducted at the Auburn University Geosciences Department. Samples were analyzed using an Agilent 7900 ICPMS.

Otolith elemental analysis

Trace element compositions were measured using laser ablation inductively coupled plasma mass spectrometry (LA-ICPMS; NWR193 193nm ArF laser attached to an Agilent 7900 Quadrupole ICPMS). Each otolith was ablated from the edge to the center of the core. Prior to each analytical ablation, a pre-ablation cleaning burn was run with 10% energy output, 25 Hz repetition rate, 45 µm spot size, and 80 µm/sec scan speed to debride any remaining contaminants from the otolith surface in the track of the laser ablation (Campana et al. 1994; Kratina 2019). Analytical ablation settings were 35% energy output, 20 Hz repetition rate, 25 µm spot size, and 10 µm/sec scan speed. Concentrations of Ca, Sr, Ba, Mg, and Mn were quantified in counts per second (cps).

Data processing

To ensure precision and control for drift, I ablated certified reference materials before starting otolith ablations each session and after every fifth otolith. The certified reference materials were a glass standard from the National Institute of Standards and Technology (NIST-612) and a matrix carbonate standard (MACS-3). Additionally, to calculate limits of detection (LODs) the background carrier gas (Argon [Ar]) was analyzed for 30 sec prior to each ablation. I used Iolite v4 software to calculate LODs based on the formula from Longerich et al. (1996) where LOD = mean background count rate + 3 standard deviations. For an element to be included in analysis it had to exceed LOD in ≥50% of samples. Iolite was also used to convert trace metal concentrations from counts per second (cps) to concentration units (ppm; Ludden et al. 1995; Ludsin et al. 2006). Element to Ca ratios were calculated to standardize concentrations for comparison with existing literature (Nelson et al. 2018; Kratina 2019). Outliers were identified as any points beyond three standard deviations from an individual element’s mean within a single ablation and removed from the data set.

Data analysis

Otolith edges were defined as 20 µm of the ablation transect from the edge, corresponding to the most recent portion of the fish’s life (exact duration of time variable depending on fish age). The mean edge elemental ratios for each species were compared to water samples collected on the same sampling trip as that individual fish. This was done to test the relationships between the ambient water element concentration and the recent element concentration in fish otoliths which provides a measure of how directly trace elements assimilate from the ambient water into fish otoliths. A linear model was used to establish the relationship between otolith edge elemental ratios and water elemental ratios (Farmer et al. 2013; Kratina 2019).

Multivariate analysis of variance (MANOVA) was used to test differences in Sr:Ca, Ba:Ca, Mn:Ca, and Mg:Ca ratios among the four sites based on (1) mean elemental ratios calculated for the length of the entire otolith transect, (2) the inner 20 µm of the otolith (otolith core), and (3) the outer 20 µm of the otolith (otolith edge). For species where MANOVA indicated significant ($\alpha \leq 0.05$) differences among sites, univariate analysis of variance (ANOVA) and Tukey's HSD pairwise comparison tests were used to determine which elements and sites drove observed differences.

Linear discriminant analysis (LDA) was used to determine classification accuracy to the sites of capture for each fish based on elemental signatures. LDA was used for each section of the ablation to determine which region of the otolith most accurately classified the fish to site of capture. LDA scores were calculated for each individual and plotted to visualize multivariate elemental signatures (Zeigler and Whitledge 2011). Non-parametric, nonlinear regression using locally estimated scatterplot smoothing (LOESS) was used to visually assess data for trends in elemental compositions. Based on observed variation in the elemental concentration displayed within otolith transects, deviations from sites of capture were identified as points outside of the 95% confidence interval calculated using the edge data of each element for all fish at a specific site.

Telemetry

During July 2020, I surgically implanted 16 combined acoustic and radio transmitter tags (CART tags, Lotek MM-MC-8-SO) in 13 Alabama Bass and 3 Tallapoosa Bass (tag weight was always <2% of individual's body weight; Winter et al. 1996). A 1.5 – 2 -cm incision was made on the left ventral side of the fish, through which a CART tag was inserted, the tags antenna was passed through a second hole made using a 14-gauge needle 1-cm anterior to the initial incision. Incisions were closed with simple interrupted polydioxanone dissolvable sutures and were additionally secured with veterinary-grade (VetBond) surgical adhesive. Collection took place between the Harris tailrace and the Randolph County Road 15 bridge in Malone, Alabama using both boat electrofishing (~7.5 RKM downstream of Harris Dam) and angling (continuous between Harris Dam tailrace and CR-15). Fish were sedated with MS-222 (approximate concentration = 150 ppm) prior to surgery and aerated water was pumped across the fish's gills during tag implantation. Implantation followed the procedures outlined in Cooke et al. (2012). Fish were held in a tank or live cage to ensure recovery after surgery before being released at their capture sites. After release, manual radio tracking efforts occurred at weekly intervals during low flows starting three weeks post-tagging via a canoe paddled from the tailrace to the CR 15 bridge. Manual tracking was conducted using a Lotek VHF Receiver GPS positioning and a handheld Yagi antenna. Fish position was determined by paddling downstream until a radio signal was detected and then wading or paddling until signal strength was highest when the antenna was pointed at the water.

In addition, eight stationary acoustic receivers were deployed in pairs to provide four gates between the R.L. Harris tailrace and CR 15 in Malone, with each gate consisting of an upstream receiver and a downstream receiver (receivers were located 20.54, 20.14, 16.90, 17.74, 14.69, and 14.31 upstream of the Wadley site). Receivers were attached to concrete anchors cabled to the bank with steel cable and deployed in water exceeding 1.5 m in depth during non-generation flows. The upstream-downstream configuration was an attempt to identify any directional movement should a fish pass both receivers within a gate. An additional four receivers formed gates at the CR-15 bridge in Malone and the AL-77 bridge in Wadley (for a total of 10 receivers) to detect any further downstream movement. A test tag towed through the receiver array was detected at all receivers, supporting that the array of receivers was functioning properly.

The river-km positional location of each tag detected was recorded from the beginning of August 2020 until the end of September 2020. Instances where receivers detected other receivers were identified and eliminated from the dataset. Graphs of each detected fish's location over time were constructed to visually assess movement. To examine fish movement with relation to dam operation, the plots of the four most frequently detected fish's locations were constructed with hydrograph data measured at the CR 15 bridge in Malone and downloaded from the USGS National Water Information System Web Interface (U.S. Geological Survey, 2021).

Results

Water chemistry

All five elements (Ca, Sr, Mn, Mg, Ba) met the limits for inclusion (i.e. above LOD in >50% of samples) for all water samples and were therefore included in analyses. Sample values are reported in element (µmol) per Ca (mol) molar ratios. Analysis of variance indicated no statistically significant differences among seasons for Sr:Ca, Mn:Ca, and Mg:Ca ratios, but significant differences among seasons in Ba:Ca ratios ($F_{3, 23} = 7.469$, $p = 0.001$; Figure 2.1). Lack of significant seasonal differences in water element to calcium ratios for Sr, Mg, and Mn indicated stability in water trace element signatures across time. Among sites, Sr:Ca and Mn:Ca ratios varied significantly ($F_{3, 23}$ = 6.36 and 4.14, p = 0.003 and 0.02 respectively) with Sr:Ca ratios increasing from upstream to downstream and a distinct Mn:Ca signature in the tailrace (Figure 2.2). Because seasonal Ba:Ca ratios varied significantly, a two-way ANOVA with an interaction term was used to test differences in Ba:Ca ratios between sites and seasons. This revealed that Ba:Ca ratios varied by site ($F_{3, 13} = 4.823$, $p = 0.01$) and season ($F_{3, 13} = 11.408$, $p = 0.001$) but the interaction term was non-significant ($F_{8, 13} = 0.665$, $p = 0.713$).

Otolith Microchemistry

Across all species, 278 otoliths were ablated using LA-ICPMS including 62 Channel Catfish, 86 Redbreast Sunfish, 75 Alabama Bass, and 55 Tallapoosa Bass. All elements quantified with LA-ICPMS (Ca, Sr, Ba, Mg, Mn) exceeded the limits of detection in >50% of samples and were therefore for included in analyses.

Water-Otolith Edge Correlation

Channel Catfish otolith edge mean elemental ratios for Sr:Ca and Ba:Ca were positively correlated with the corresponding water elemental ratios (R^2 = 0.21 and 0.15, p = 0.01 and 0.02 respectively; Figure 2.3). However, regressions of otolith edge and water Mg:Ca and Mn:Ca were not statistically significant (R^2 = 0.02 and 0.01, p = 0.51 and 0.54 respectively; Figure 2.3).

Redbreast Sunfish otolith edge element concentration was not significantly related to water element concentrations across sites for any element (R^2 = 0.06, 0.001, 0.02, 0.0001; p= 0.09, 0.84, 0.45, 0.95; Figure 2.4).

Mean Alabama Bass otolith edge Sr:Ca ratios were positively correlated with water Sr:Ca ratios (R^2 = 0.30, p = 0.002; Figure 2.5). Otolith Ba:Ca, Mg:Ca, and Mn:Ca were not significantly related to water chemistry (R^2 = 0.11, 0.0002, 0.001; p= 0.08, 0.95, 0.87; Figure 2.5)

Tallapoosa Bass otolith edge elemental compositions were not significantly correlated with water compositions for any elements (R^2 = 0.006, 0.001, 0.001, 0.02; p= 0.63, 0.86, 0.87, 0.42; Figure 2.6).

Multivariate analysis of variance (MANOVA) indicated that Channel Catfish otolith edge elemental signatures varied among sites (MANOVA: $F_{3,58}$ = 1.90, p = 0.03). Subsequent univariate analyses of variance (ANOVA) showed Sr:Ca (ANOVA: $F_{3,58}$ = 5.006, p = 0.004) and Ba:Ca (ANOVA: $F_{3,58}$ = 6.51, p = 0.001) were the primary differentiators between sites (Figure 2.7). Edge elemental ratios for Sr:Ca and Ba:Ca tended to increase with distance downstream of Harris Dam and reached their highest mean elemental ratios at Horseshoe Bend (Figure 2.7).

Otolith elemental signatures across the entire ablation transect varied significantly among sites (MANOVA: $F_{3,58} = 3.70$, $p < 0.001$). Sr:Ca (ANOVA: $F_{3,58} = 4.87$, $p = 0.004$) and Ba:Ca (ANOVA: $F_{3,58} = 5.30$, $p = 0.003$) were again the primary differentiators between sites (Figure 2.8). As in the edge data, there was a longitudinal gradient present in the whole transect Sr:Ca and Ba:Ca mean elemental ratios (Figure 2.8).

Channel Catfish otolith core elemental ratios were also significantly different among sites (MANOVA: $F_{3,58} = 2.26$, $p = 0.006$). Unlike in other portions of Channel Catfish otoliths, Ba:Ca (ANOVA:$F_{3,58} = 1.92$, $p = 0.14$) was not a primary driver of differences between sites; however, Sr:Ca (ANOVA: $F_{3,58} = 6.22$, $p = 0.001$) and Mn:Ca (ANOVA: $F_{3,58} = 4.50$, $p = 0.007$) did relate to site differences (Figure 2.9). Both Sr:Ca and Mn:Ca tended to increase with distance downstream of the Harris Dam tailrace and Mn elemental ratios at Wadley were similar to those in the tailrace and upstream of Harris Reservoir (Figure 9).

Linear discriminant analysis including all available element ratios (Sr:Ca, Ba:Ca, Mg:Ca, Mn:Ca) was most accurate for Channel Catfish whole transect otolith means and lowest for otolith edge means, although all classifications were generally similar (Table 2.1). Classification accuracy was highest at Lee's Bridge for Channel Catfish whole transect otolith means and otolith edge means but highest in the tailrace for otolith core means (Table 2.1). Ba:Ca contributed most to discrimination for all otolith sections with minimal contributions from the other elements (Table 2.2). Overall classification accuracies were limited by lack of consistent differences in water trace element compositions among downstream sites and weak correlations between water and otolith trace element concentrations (Figures 2.2, 2.3). The first two linear discriminant axes accounted for 52.2-87.6% and 6.7-41.3% of the variation in the data depending on otolith region, and plots of the first two linear discriminant axes indicated poor separation among sites for all otolith sections (Figure 2.10).

Visual evaluation of smoothed Sr:Ca whole transect time series plots revealed three distinct patterns. Pattern one was characterized by an overall homogenous Sr:Ca ratios over the entire transect, falling within the 95% confidence interval of ratios for a specific site from core to edge (Figure 2.11). Pattern two was characterized by variation in the Sr:Ca value during the early life (core; right side of Figure 12) portion of an ablation transect before stabilizing within the 95% confidence interval of Sr:Ca ratios for the specific site. Pattern three was characterized by variation throughout the life of the fish with Sr:Ca ratios deviating from, and returning to, the 95% confidence interval for a site at least once (Figure 2.13). Distributions of each pattern type were substantially different among sites, with pattern one occurring most frequently in the tailrace and decreasing downstream (Table 2.3). Pattern two was most frequent for Channel Catfish collected at Lee’s Bridge and Horseshoe Bend and relatively rare at the tailrace and Wadley (Table 2.3). Finally, pattern three was absent in ablation profiles from Channel Catfish collected in the tailrace, most frequent at Lee’s Bridge, and increased with distance downstream of Harris Dam (Table 2.3).

Redbreast Sunfish otolith edge elemental ratio means differed significantly between sites (MANOVA: $F_{3,82} = 4.30$, $p < 0.001$) and subsequent analyses of variance indicated that Sr:Ca (ANOVA: $F_{3,82} = 16.06$, $p < 0.001$), Ba:Ca (ANOVA: $F_{3,82} = 8.29$, $p < 0.001$), and Mn:Ca (ANOVA: $F_{3,82} = 2.92$, $p = 0.04$) were the primary drivers of differences among sites (Figure 2.14). Sr:Ca ratios were clearly delineated into upstream (Lee's Bridge and Harris Tailrace) and downstream groups (Wadley and Horseshoe Bend) while Ba:Ca and Mn:Ca ratios were elevated at Wadley and overall less informative (Figure 2.14).

Mean elemental ratios across the entire otolith ablation transect for Redbreast Sunfish also differed significantly among sites (MANOVA: $F_{3,82} = 4.56$, $p < 0.001$) with the primary drivers being Sr:Ca (ANOVA: $F_{3,82} = 27.37$, $p < 0.001$) and Ba:Ca (ANOVA: $F_{3,82} = 5.45$, $p = 0.001$). Both Sr:Ca and Ba:Ca ratios increased from upstream (Lee's Bridge) to downstream (Horseshoe Bend; Figure 2.15).

My MANOVA results indicated that Redbreast Sunfish otolith core elemental ratios varied significantly between sites (MANOVA: $F_{3,82} = 15.00$, $p < 0.001$) and Sr:Ca (ANOVA: $F_{3,82} = 19.62$, $p < 0.001$) and Ba:Ca (ANOVA: $F_{3,82} = 6.53$, $p < 0.001$) were again the most informative elements. As with the whole ablation transect mean ratios, core mean ratios for Sr:Ca and Ba:Ca increased from upstream to downstream (Figure 2.16).

As with Channel Catfish, LDA was most accurate classifying Redbreast Sunfish back to capture location when using whole otolith ablation transect mean elemental rations and least accurate when using otolith core means (Table 2.4). Classification accuracy was highest at Horseshoe Bend and Wadley when considering otolith edge means and whole transect mean element ratios and highest for Horseshoe Bend and the tailrace when considering otolith core means (Table 2.4). No single element was more informative for classifying Redbreast Sunfish to site of capture for any otolith portion and all contributed similar amounts (Table 2.5). Weak correlation between water trace element composition and Redbreast Sunfish otolith trace element composition likely limited classification accuracies (Figure 2.4). The first two linear discriminant axes accounted for 73.9-93.1% and 5.1-25.6% of the variation in the data depending on the portion of the otolith analyzed and plots of the first two linear discriminant axes showed little separation among sites in Redbreast Sunfish trace element signatures (Figure 2.17).

Redbreast Sunfish patterns in smoothed Sr:Ca ratios were similar to Channel Catfish. Redbreast Sunfish exhibiting pattern one were most prevalent at Wadley and the tailrace and less prevalent at Lee’s Bridge and Horseshoe Bend (Table 2.3). There was an increase in the frequency of Redbreast Sunfish with pattern two and three with distance downstream (Table 2.3).

Alabama Bass trace element signatures for otolith edge (MANOVA: $F_{3,71} = 7.88$, $p < 0.001$), whole transect otolith (MANOVA: $F_{3,71} = 13.53$, $p < 0.001$), and otolith core (MANOVA: $F_{3,71} = 3.38$, $p < 0.001$) means varied among sites.

For otolith edge means elemental ratios, Sr:Ca (ANOVA: $F_{3,71} = 36.69$, $p < 0.001$), Ba:Ca (ANOVA: $F_{3,71} = 11.97$, $p < 0.001$), and Mg:Ca (ANOVA: $F_{3,71} = 3.89$, $p = 0.01$) were the primary separators among sites. Sr:Ca ratios for Alabama Bass edge means were elevated at Wadley and Horseshoe Bend compared to Lee’s Bridge and the Tailrace. Ba:Ca ratios increased from upstream (Lee’s Bridge) to Wadley but decreased at Horseshoe Bend (Figure 2.18).

For whole ablation transect mean elemental ratios, Sr:Ca (ANOVA: $F_{3,71} = 42.94$, $p < 0.001$), Mg:Ca (ANOVA: $F_{3,71} = 16.86$, $p < 0.001$), Ba:Ca (ANOVA: $F_{3,71} = 16.26$, $p < 0.001$), and Mn:Ca (ANOVA: $F_{3,71} = 2.83$, $p = 0.04$) were significantly different among sites (Figure 2.19). Alabama Bass whole transect otolith Sr:Ca ratios increased from upstream to downstream while Ba:Ca were separated into two groupings with the ratios from the upstream site significantly lower than those from the three downstream sites (Figure 2.19). Mg:Ca levels were significantly elevated in the tailrace compared to the other sites (Figure 2.19).

Alabama Bass otolith core elemental ratios indicated that Sr:Ca ($F_{3,71} = 10.81$, $p < 0.001$), Mn:Ca ($F_{3,71} = 4.88$, $p = 0.004$), and Ba:Ca ($F_{3,71} = 4.79$, $p = 0.004$) drove differences between sites. As with Alabama Bass whole transect elemental ratios, Alabama Bass core Sr:Ca ratios increased from upstream to downstream (Figure 20). Additionally, Alabama Bass Ba:Ca and Mn:Ca varied by site with Ba:Ca elevated in the tailrace and at Horseshoe Bend and Mn:Ca elevated at Horseshoe Bend (Figure 20).

Reclassification using LDA was most accurate when classifying Alabama Bass to capture location using whole transect otolith mean elemental ratios, similar to results from Channel Catfish and Redbreast Sunfish (Table 2.6). Ba:Ca and Mn:Ca were identified as the most important elements for Alabama Bass classification based on whole transect otolith elemental ratios (Table 2.7). Classification accuracy based on

Alabama Bass core elemental ratios was less successful, primarily driven by strong contributions from Mn:Ca, Ba:Ca, and Mg:Ca (Table 2.6, 2.7). The first two linear discriminant functions accounted for 67.8-82.9% and 15.8-23.3% of the variation in the data depending on otolith region and plots of the whole transect elemental ratios showed reasonable groupings by site, likely showing why classification accuracy was so high for these data (Figure 2.21).

Trends in Alabama Bass Sr:Ca transect patterns were different than those in Channel Catfish and Redbreast Sunfish. Fish from Lee's Bridge made up the majority of the fish identified as pattern one, and downstream of the dam elemental ratios were similar among sites (Table 2.3). The tailrace had slightly more pattern two Alabama Bass than the other sites though the differences were minimal (Table 2.3). Pattern three was variable between sites with an increasing gradient from upstream to downstream (Table 2.3).

MANOVA did not indicate significant differences among sites in Tallapoosa Bass otolith edge ($F_{3,51} = 1.39$, $p = 0.18$), whole transect otolith ($F_{3,51} = 1.23$, $p = 0.27$), or otolith core ($F_{3,51} = 0.65$, $p = 0.79$) means; further univariate testing was not conducted as it was unlikely to identify significant differences given its lower power compared to multivariate techniques. Without significant differences for any portion of Tallapoosa Bass otoliths, LDA classification accuracy was poor as expected, with the majority of fish grouped incorrectly captured at Horseshoe Bend (Table 2.8). High total accuracy percentages were inflated by uneven sample sizes and should not be over interpreted as LDA failed to correctly classify fish to capture location for most site x otolith section combinations (Table 2.8).

Trends in Tallapoosa Bass otolith Sr:Ca plots revealed that the frequency of occurrence of pattern one increased with distance downstream of the dam, as did pattern three (Table 2.3). Because of limited sample sizes in the tailrace (n=3) and at Lee's Bridge (n=2), interpreting these trends should be done with caution.

Telemetry

Of the 16 tags deployed, 12 were detected by the stationary acoustic receiver array and 10 were detected during at least one manual tracking trip (Table 2.9). Smaller CART tags implanted in fish <600 g had a battery life of ~30 days and were not active beyond the second manual tracking effort. Nine of the remaining 10 active tags were detected in at least one subsequent manual tracking event (Figure 2.25). Of the 12 tags detected by the stationary acoustic receiver array, 8 were detected only at a single location (i.e., their locations never changed) and maximum displacement detected over the course of the study by the acoustic array was 6.23 RKM (Figures 2.22, 2.23). The remaining four tags were detected at more than one receiver in the array (Figure 2.23).

Plots of the four most frequently detected fish's positions alongside data from the USGS gage at Malone, AL showed that these four fish were not displaced by the normal peaking operation of Harris Dam (Figure 2.24). During the one major flow event that occurred during my study, detections decreased which was especially clear when reviewing data for tag number 29392 (Figure 2.24). When operation of Harris Dam resumed its normal schedule, fish position was the same as before the high flow event (Figure 2.24).

Discussion

Otolith trace element analysis has the potential to be highly effective at recreating habitats used by fishes during various portions of their life depending on chemical differences across sites. Previous studies have been highly successful employing these techniques at broader scales (e.g., Bronte et al. 1996; Wells et al. 2003; Kratina 2019), among various site types (e.g., river, tributary, floodplain lakes; Zeigler and Whitledge 2011; Carleson et al 2016), and across salinity gradients (Farmer et al. 2013; Gahagan et al. 2012; Walther and Limburg 2012) but few have worked in such a restricted spatial scale within the mainstem of a single river.

Here, I expected to see strong delineation between the upstream, unregulated site (Lee’s Bridge) and the sites on the downstream, regulated portion of the river due to the presence of Harris Reservoir and the operation of Harris Dam. These differences were present in several elements within water samples, with Sr:Ca ratios elevated at the three sites below the dam and Mn:Ca ratios were elevated in the Harris Dam tailrace. These observations were important because they suggested the potential for differences in otolith trace element compositions across sites given that water chemistry is the primary driver of otolith chemistry (Campana 1999; Eldson et al. 2008; Walther and Limburg 2012). Additionally, the lack of seasonal variation in water elemental composition was important, indicating that interpretation of elemental signatures in fish otoliths was not complicated by seasonal fluctuation in water chemistry. Demonstrating spatial variation but temporal stability in water sample chemistry is often considered a vital first step in trace element analyses (Campana 1999; Walther and Limburg 2012).

Unfortunately, the correlation between water chemistry and otolith edge chemistry was weak for two of my target species, Alabama Bass and Channel Catfish, and not significant for the other two. This suggested that interpreting otolith elemental signatures was limited, the fish recently moved, or that incorporation of ambient trace elemental signatures did not occur with similar ratios to what was present in the ambient water. Campana (1999) outlined several of the barriers regulating incorporation of elements into otoliths, stating that the primary path was from water into fish blood plasma and then into the otolith. It is feasible that along this pathway, some disruption could occur, or the incorporation of elements from other sources (i.e., diet) may have been greater than from water in my study (Campana 1999). Despite this, otolith elemental signatures still varied among sites for three of my four target species, most often similarly to longitudinal trends in water sample trace element composition. Overall Sr:Ca and Ba:Ca were the most informative elements, as has been found in several other studies where Sr:Ca ratios were used to describe fish movements via otolith ablation profiles (Zeigler and Whitledge 2011; Gahagan et al. 2012; Carlson et al 2016; Kratina 2019; Chrisp 2020).

When using the entire suite of elements I quantified through LA-ICPMS, Alabama Bass was the only species that could be reclassified back to site of capture with a high level of accuracy (>75%). Redbreast Sunfish and Channel Catfish were reclassified to site at approximately twice the accuracy of a random guess (approximately 50% for four sites); these accuracy levels are greater than or similar to previous studies using trace element analyses in freshwater (Bronte et al. 1996; Wells et al. 2003; Brazner et al. 2004; Zeigler and Whitledge 2011; Kratina 2019). Classification accuracy for Tallapoosa Bass was limited due to extremely unbalanced and low sample sizes across sites.

Whole ablation mean elemental ratios were more accurate for classification than were otolith edge mean elemental ratios, which was unexpected. Otolith edge element concentration should correlate with the most recent habitats experienced by the fish and therefore be the better predictor. This indicated that data important for classification of the fish to their capture location occurred outside of the 20-µm edge section of the otolith, or that the fish recently moved to the capture site. The accuracy of classification to their capture location based on whole transect otolith data also suggests that these fish generally have high site fidelity as adults.

Otolith core mean elemental ratios were the least accurate for classifying fish to capture location, for which there are two potential explanations. First, fish may not recruit to or use the same habitat that they do as adults. Second, inherent elevated trace element ratios in otolith cores may potentially limit classification accuracy (Ruttenberg et al. 2005). However, examination of Sr:Ca time series data for whole transect ablations revealed that otolith core elemental ratios did not appear to be elevated in all fish as would be expected if the second scenario above were occurring. Taken together, this may indicate that fish in the Tallapoosa River do not recruit to or use the same habitat during their entire life.

Three patterns were identified in otolith Sr:Ca time series data, all of which were present in at least some individuals within all target species. These patterns were largely identified by first defining the “expected” range of otolith Sr:Ca ratios for a specific site for a given species (calculated as the 95% confidence interval around the mean of otolith edge ratios from that site). This calculation was done on a site and species level to eliminate potential species effects. The first pattern was characterized by consistent Sr:Ca ratios throughout the entire ablation profile. This likely indicated that the fish was largely stationary and remained in a portion of river small enough to not experience a broad range of water elemental compositions. The second pattern was defined by deviation from the site’s 95% confidence interval during a fish’s early life history (i.e., the portion of the ablation profile corresponding with the otolith core), before the Sr:Ca ratios eventually fell within the expected interval. This pattern likely indicated that a fish recruited from a site different than its capture location. The third pattern was typified as variation outside of the expected range at least once during a fish’s life (not necessarily during early life) and indicated movement away from and back to the site of capture.

The distributions of these patterns indicated that fish captured in the tailrace were more likely to have recruited from the tailrace or in the general proximity within the same water chemistry. Weyers et al. (2003) documented strong negative effects of pulsed, high velocity water on fish recruitment; as such, it is unlikely that reproduction occurred in the tailrace. Instead, the fish likely recruited from nearby areas where the water elemental composition was not sufficiently distinct from the site of capture to create variation in otolith Sr:Ca ratios, though data supporting this observation are not presented here. With distance downstream, the proportion of Channel Catfish and Redbreast Sunfish that recruited from their site of capture (pattern one) decreased, while the number that recruited from a site with differing water chemistry increased (pattern two). This indicated that the tailrace did not receive any recruitment subsidy of Channel Catfish and Redbreast Sunfish from these further downstream sites. Additionally, the frequency of fish leaving and returning to the site of capture (pattern three) increased with distance downstream of Harris Dam for all target species, potentially indicating that fish leaving the tailrace did not return. Interestingly, classification accuracy of Redbreast Sunfish and Channel Catfish in the tailrace, based on core mean elemental ratios, was reasonable, indicating that fish collected in the tailrace likely spent a significant portion of their early life there and/or recruited from areas with similar water chemistry.

Trends in the prevalence of pattern three suggest that with distance downstream, fish were more likely to leave and return to the capture site. These movements could indicate that the fish rely on a broader habitat and/or resource base, as would be expected following the river continuum concept (Vannote et al. 1980). Interestingly, pattern three was most common in Alabama Bass collected from the two downstream-most sites. My study and previous telemetry studies conducted on Alabama Bass and Tallapoosa Bass at these sites failed to identify large scale movement within the river and found that tagged Alabama Bass generally remained within 6-8 river km of their tagging location and moved laterally to the river margins during high flows (Earley and Sammons 2015). However, Earley and Sammons (2015) did observe several fish that made considerable (≈20 km) movements from their tagging location during the spawning season which could explain why many of the Alabama and Tallapoosa Bass exhibiting pattern three showed multiple deviations from their sites of capture.

Overall movement of fish as quantified via telemetry was very low, with most fish occupying a small stretch of river for the majority of the time they were detected in the study area. Redeye Bass home range size was previously estimated by Knight et al. (2011) in tributaries of the Tallapoosa River, with the conclusion that home range size decreased with increased fish size. Given that all tagged Tallapoosa Bass were near the maximum size for the species, I expected little movement based on the findings of Knight et al. (2011). The fish tagged in Knight et al. (2011) were smaller (81-200 g) than the Tallapoosa Bass tagged in this study (380-400 g). A more recent study of Alabama Bass movement by Earley and Sammons (2015) found similar results, stating that Alabama Bass remained within the 8 km river reach where they were tagged. The maximum movement detected by my acoustic array was for tag numbers 28688 and 28692, which both made maximum movements of only approximately 6.2 RKM. Based on the evidence in the literature, combined with my telemetry data, it is clear that high flow from peaking hydropower operation is not displacing Tallapoosa or Alabama Bass downstream. Manual tracking data further support this claim as most fish were detected within a few hundred meters of where they were detected during the previous trip. By examining the manual tracking detections that occurred closest to the tailrace versus those further downstream, it appears that movement may increase with distance from the dam (although additional data would be required to confirm such a conclusion). This could indicate that fish closer to the tailrace are restricted to smaller pockets of suitable habitat. Further work comparing available habitat to finer scale positional location/movement is needed to elucidate such a pattern.

Summary

My goals included determining whether trace element analyses were suitable for use within the Tallapoosa River, and more broadly, if these types of analyses could be successful over a relatively restricted spatial scale. Based on the non-migratory nature of the selected target species, the underlying geology of the Tallapoosa River, and fluctuations in flow and temperature caused by Harris Dam, I expected to be able to identify a longitudinal gradient in water chemistry, to accurately classify fish to capture location, and to identify fish natal origins and movement patterns.

Despite poor relationships between otolith edge trace element composition and water sample trace element composition, my study was successful at classifying target species to their capture location based on otolith trace element compositions. However, relating these compositions to the water chemistry was less successful. In instances where water chemistry was the primary predictor of otolith chemistry, investigators were able to draw conclusions about the scale and direction of fish movements, or chronological habitat use based on observed fluctuation in otolith elemental signatures (Kennedy et al. 2002; Farmer et al. 2008; Zeigler and Whitledge 2011; Kratina 2019; Chrisp 2020). In my study, water chemistry was not a strong predictor of otolith trace element compositions; as such, determining scale and directionality of movements was not possible given that otolith signatures could not be directly related to a gradient or threshold in water chemistry. Instead, movements were identified within ablation transects independently of water chemistry and were only informative regarding an individual fish's movement with relation to its site of capture.

Sr:Ca was the most informative elemental ratio due to strong differences between sites in water chemistry and otolith chemistry for all target species; however, adjacent sites rarely differed significantly which limited the interpretation of movement data. Examining Sr:Ca ablation profiles was previously used to identify dam passage events (Kratina 2019), assign lifetime habitat residence (Farmer et al. 2013; Chrisp 2020), quantify migrations (Gahagan et al. 2012; Walther and Limburg 2012), and define natal origins (Eldson et al. 2008; Gahagan et al. 2012; Carlson et al. 2016). Sr:Ca profiles revealed three general patterns of movement in fish collected from the Tallapoosa River and the distribution of these patterns among sites indicated that river regulation by Harris Dam potentially influenced fish recruitment and dispersal in the downstream areas.

Finally, tagging and tracking efforts failed to detect any substantial movements by Alabama Bass or Tallapoosa Bass in the Harris Tailrace. Other studies of riverine bass have reached similar conclusions; however, it does appear that the bulk of black bass movement is linked to spring spawning seasons (Knight et al. 2011; Earley and Sammons 2015; Sammons and Earley 2015). These springtime spawning movements may have been responsible for much of the variation in Alabama Bass and Tallapoosa Bass Sr:Ca ablation profiles exhibiting pattern three. During the only high flow event that occurred during the tracking portion of my study, detections did decrease. This is likely due to interference from turbulent water on tag acoustic signals although it is possible that fish were displaced during this event. Immediately following the high flows, fish were detected in the same reach as before the high flow event, again supporting that the decrease in detections was due to interference rather than displacement.

My study demonstrated that trace element analysis is useful even on a restricted spatial scale; however, it appears that not all species assimilate trace elements into their otoliths at a similar ratio to the ambient water. Thus, it is important to carefully quantify water trace element composition at a greater spatial resolution to adequately relate fish otolith elemental concentrations to fish movements. In studies focused on broader spatial scales or across stronger elemental gradients, relationships between otolith chemistry and water samples have been far stronger than observed in my study (Zeigler and Whitledge 2011; Gahagan et al. 2012; Walther and Limburg 2012; Farmer et al. 2013; Chrisp 2021). More specifically, in estuarine studies such as Chrisp (2021), researchers were able to identify habitat thresholds in water chemistry and directly apply those thresholds to patterns of fish residency and ontogeny (Lowe et al. 2009; Farmer et al. 2013). Such results were impossible for my work as defining site-specific thresholds was difficult and many sites’ trace element ratio values overlapped.

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Table 2.1: Results of Linear Discriminant Analysis (LDA) for Channel Catfish collected from four sites on the Tallapoosa River, Alabama. Correct classifications are indicated in bold; columns correspond to capture location while rows correspond to LDA classification.

Otolith Section	**LDA Classification**	**Capture Location**				**Total Accuracy (%)**
		LB	**TR**	**WD**	**HB**	
Mean Edge	**LB**	**15**	7	4	2	51.6
	TR	3	**8**	2	4	
	WD	0	0	**0**	1	
	HB	1	3	3	**9**	
Mean Transect	**LB**	**14**	4	2	3	58.1
	TR	3	**10**	2	4	
	WD	0	0	**3**	0	
	HB	2	1	2	**9**	
Mean Core	**LB**	**12**	4	2	3	55.7
	TR	6	**13**	5	4	
	WD	0	0	**0**	0	
	HB	0	1	2	**9**	

Table 2.2: Linear discriminant coefficients generated by LDA for Channel Catfish with sites as groups, the importance of each element for classifying to each site, and the proportion of trace. Larger absolute values indicate stronger effects of a specific element on a given linear discriminant. Mean edge refers to the outer 20-µm of the otolith ablation transect, mean core refers to inner 20-µm of the otolith ablation transect, and whole transect refers to the entire otolith ablation transect. Sites were: LB = Lee's Bridge, TR = Harris Dam tailrace, WD = Wadley, HB = Horseshoe Bend.

Mean Edge	**Linear Discriminants**		**LD-1**	**LD-2**	**LD-3**
	Proportion of Trace (%)		87.6	0.07	0.06
	Elements used in LDA	**Sr_{88}**	-0.002	0.010	-0.001
		Ba_{137}	-0.180	-0.240	-0.090
		Mg_{24}	-0.001	0.001	-0.010
		Mn_{55}	-0.040	-0.080	-0.003
Whole Transect	**Linear Discriminants**		**LD-1**	**LD-2**	**LD-3**
	Proportion of Trace (%)		57.8	37	0.05
	Elements used in LDA	**Sr_{88}**	-0.010	0.003	0.001
		Ba_{137}	0.350	0.110	-0.160
		Mg_{24}	-0.010	0.050	0.040
		Mn_{55}	-0.060	-0.080	-0.160
Mean Core	**Linear Discriminants**		**LD-1**	**LD-2**	**LD-3**
	Proportion of Trace (%)		74.2	22.5	0.03
	Elements used in LDA	**Sr_{88}**	-0.004	0.001	-0.001
		Ba_{137}	0.080	0.100	0.010
		Mg_{24}	-0.004	-0.010	-0.030
		Mn_{55}	-0.030	-0.020	0.030

Table 2.3: Frequency of occurrence by site of three patterns identified in otolith Sr:Ca ratios from four fish species collected in the Tallapoosa River, Alabama. Patterns were: 1) individuals spent their entire life at the capture location, 2) individuals recruited to the capture location from a different river section, and 3) individuals moved away from, then returned to, the capture location.

Channel Catfish

Site	Pattern 1	Pattern 2	Pattern 3
LB	5	7	7
TR	17	1	0
WD	6	1	2
HB	5	7	4
All	33	16	13

Redbreast Sunfish

Site	Pattern 1	Pattern 2	Pattern 3
LB	7	3	4
TR	19	0	1
WD	20	3	3
HB	7	10	9
All	53	16	17

Alabama Bass

Site	Pattern 1	Pattern 2	Pattern 3
LB	16	3	1
TR	4	6	3
WD	4	3	13
HB	5	3	13
All	29	15	30

Tallapoosa Bass

Site	Pattern 1	Pattern 2	Pattern 3
LB	2	0	0
TR	2	0	1
WD	8	4	5
HB	14	5	14
All	26	9	20

Table 2.4: Results of Linear Discriminant Analysis (LDA) for Redbreast Sunfish collected from four sites on the Tallapoosa River, Alabama. Correct classifications are indicated in bold; columns correspond to capture location while rows correspond to LDA classification. Mean edge refers to the outer 20-µm of the otolith ablation transect, mean core refers to inner 20-µm of the otolith ablation transect, and whole transect refers to the entire otolith ablation transect. Sites were: LB = Lee's Bridge, TR = Harris Dam tailrace, WD = Wadley, HB = Horseshoe Bend.

Otolith Section	**LDA Classification**	**Capture Location**				**Total Accuracy (%)**
		LB	**TR**	**WD**	**HB**	
Mean Edge	**LB**	**3**	3	0	0	51.1
	TR	9	**7**	3	8	
	WD	2	3	**18**	2	
	HB	0	7	5	**16**	
Mean Transect	**LB**	**9**	1	2	0	62.8
	TR	4	**8**	5	2	
	WD	0	9	**16**	3	
	HB	1	2	3	**21**	
Mean Core	**LB**	**0**	0	0	0	39.5
	TR	9	**14**	4	8	
	WD	5	3	**6**	4	
	HB	0	3	16	**14**	

Table 2.5: Linear discriminant coefficients generated by LDA for Redbreast Sunfish with sites as groups, the importance of each element for classifying to each site, and the proportion of trace. Larger absolute values indicate stronger effects of a specific element on a given linear discriminant.

	Linear Discriminants		**LD-1**	**LD-2**	**LD-3**
Mean Edge	**Proportion of Trace (%)**		73.9	25.6	0.005
	Elements used in LDA	$\mathbf{Sr}_{88}$	-0.005	0.004	0.000
		$\mathbf{Ba}_{137}$	-0.026	-0.091	0.070
		$\mathbf{Mg}_{24}$	0.005	0.029	-0.001
		$\mathbf{Mn}_{55}$	-0.030	-0.050	0.121
	Linear Discriminants		**LD-1**	**LD-2**	**LD-3**
Whole Transect	**Proportion of Trace (%)**		92.6	5.5	1.9
	Elements used in LDA	$\mathbf{Sr}_{88}$	-0.010	-0.003	0.001
		$\mathbf{Ba}_{137}$	0.028	0.114	-0.010
		$\mathbf{Mg}_{24}$	0.010	-0.047	0.041
		$\mathbf{Mn}_{55}$	0.031	0.031	0.057
	Linear Discriminants		**LD-1**	**LD-2**	**LD-3**
Mean Core	**Proportion of Trace (%)**		93.1	5.1	1.8
	Elements used in LDA	$\mathbf{Sr}_{88}$	-0.004	0.001	-0.001
		$\mathbf{Ba}_{137}$	0.009	-0.049	0.009
		$\mathbf{Mg}_{24}$	0.001	0.004	-0.004
		$\mathbf{Mn}_{55}$	0.013	-0.026	-0.025

Table 2.6: Results of Linear Discriminant Analysis (LDA) for Alabama Bass collected from four sites on the Tallapoosa River, Alabama. Correct classifications are indicated in bold; columns correspond to capture location while rows correspond to LDA classification. Mean edge refers to the outer 20-µm of the otolith ablation transect, mean core refers to inner 20-µm of the otolith ablation transect, and whole transect refers to the entire otolith ablation transect. Sites were: LB = Lee’s Bridge, TR = Harris Dam tailrace, WD = Wadley, HB = Horseshoe Bend.

Otolith Section	**LDA Classification**	**Capture Location**				**Total Accuracy (%)**
		LB	**TR**	**WD**	**HB**	
Mean Edge	**LB**	**16**	6	3	4	63.5
	TR	2	**6**	0	0	
	WD	0	1	**12**	4	
	HB	2	0	5	**13**	
Mean Transect	**LB**	**17**	2	1	1	82.7
	TR	1	**11**	0	0	
	WD	2	1	**17**	3	
	HB	0	0	2	**17**	
Mean Core	**LB**	**17**	12	2	3	48.6
	TR	0	**0**	0	0	
	WD	3	1	**18**	17	
	HB	0	0	0	**1**	

Table 2.7: Linear discriminant coefficients generated by LDA for Alabama Bass with sites as groups, the importance of each element for classifying to each site, and the proportion of trace. Larger absolute values indicate stronger effects of a specific element on a given linear discriminant.

Mean Edge	**Linear Discriminants**		**LD-1**	**LD-2**	**LD-3**
	Proportion of Trace (%)		82.9	15.8	1.3
	Elements used in LDA	**Sr_{88}**	-0.008	-0.004	0.000
		Ba_{137}	0.026	0.317	0.014
		Mg_{24}	0.009	-0.001	-0.006
		Mn_{55}	0.032	-0.144	-0.447
Whole Transect	**Linear Discriminants**		**LD-1**	**LD-2**	**LD-3**
	Proportion of Trace (%)		79.7	16.2	4.1
	Elements used in LDA	**Sr_{88}**	-0.012	-0.001	-0.005
		Ba_{137}	0.188	0.367	0.242
		Mg_{24}	0.034	0.007	-0.045
		Mn_{55}	-0.010	-0.040	-0.079
Mean Core	**Linear Discriminants**		**LD-1**	**LD-2**	**LD-3**
	Proportion of Trace (%)		67.8	23.3	8.9
	Elements used in LDA	**Sr_{88}**	-0.005	-2.183	0.003
		Ba_{137}	0.037	1.715	0.037
		Mg_{24}	0.007	-1.282	-0.025
		Mn_{55}	-0.013	-8.404	-0.032

Table 2.8: Results of Linear Discriminant Analysis (LDA) for Tallapoosa Bass collected from four sites on the Tallapoosa River, Alabama. Correct classifications are indicated in bold; columns correspond to capture location while rows correspond to LDA classification. Mean edge refers to the outer 20-µm of the otolith ablation transect, mean core refers to inner 20-µm of the otolith ablation transect, and whole transect refers to the entire otolith ablation transect. Sites were: LB = Lee's Bridge, TR = Harris Dam tailrace, WD = Wadley, HB = Horseshoe Bend.

Otolith Section	**LDA Classification**	**Capture Location**				**Total Accuracy (%)**
		LB	**TR**	**WD**	**HB**	
Mean Edge	**LB**	**0**	0	0	0	61.8
	TR	0	**1**	0	0	
	WD	1	0	**6**	6	
	HB	1	2	11	**27**	
Mean Transect	**LB**	**0**	0	0	0	72.7
	TR	0	**1**	0	0	
	WD	1	0	**9**	3	
	HB	1	2	8	**30**	
Mean Core	**LB**	**0**	1	0	1	38.2
	TR	0	**0**	0	0	
	WD	2	2	**15**	26	
	HB	0	0	2	**6**	

Table 2.9: Metadata for fish tagged with combined acoustic and radio tags in the Tallapoosa River, Alabama. Species are: ALAB = Alabama Bass, TPBA = Tallapoosa Bass. Weight NAs were due to a scale malfunction.

Radio ID	Acoustic ID	Detections	Species	TL (mm)	WT (g)	External Tag	Release Timestamp
20	28688	42	ALAB	344	490	1917	06/30/2020 12:30
21	28690	0	ALAB	358	550	1918	06/30/2020 12:30
22	28692	59991	ALAB	365	572	1919	06/30/2020 10:43
23	28604	0	TPBA	312	410	N	07/03/2020 08:32
24	28696	0	TPBA	310	380	N	07/03/2020 11:30
25	28698	1642	TPBA	295	380	1914	07/09/2020 10:10
160	28388	96854	ALAB	472	1100	1922	06/30/2020 10:43
161	29390	665	ALAB	418	860	1921	06/30/2020 10:43
162	29392	43367	ALAB	418	806	1920	06/30/2020 10:43
163	29394	0	ALAB	442	900	1916	06/30/2020 12:30
165	29398	419	ALAB	474	1140	1915	06/30/2020 12:30
193	29454	869	ALAB	451	NA	1913	07/09/2020 10:10
196	29460	67	ALAB	432	NA	1911	07/09/2020 10:10
199	29466	115325	ALAB	432	870	N	07/03/2020 14:11
202	29472	476	ALAB	432	870	N	07/03/2020 11:30
204	29476	6123	ALAB	489	NA	1912	07/09/2020 10:10

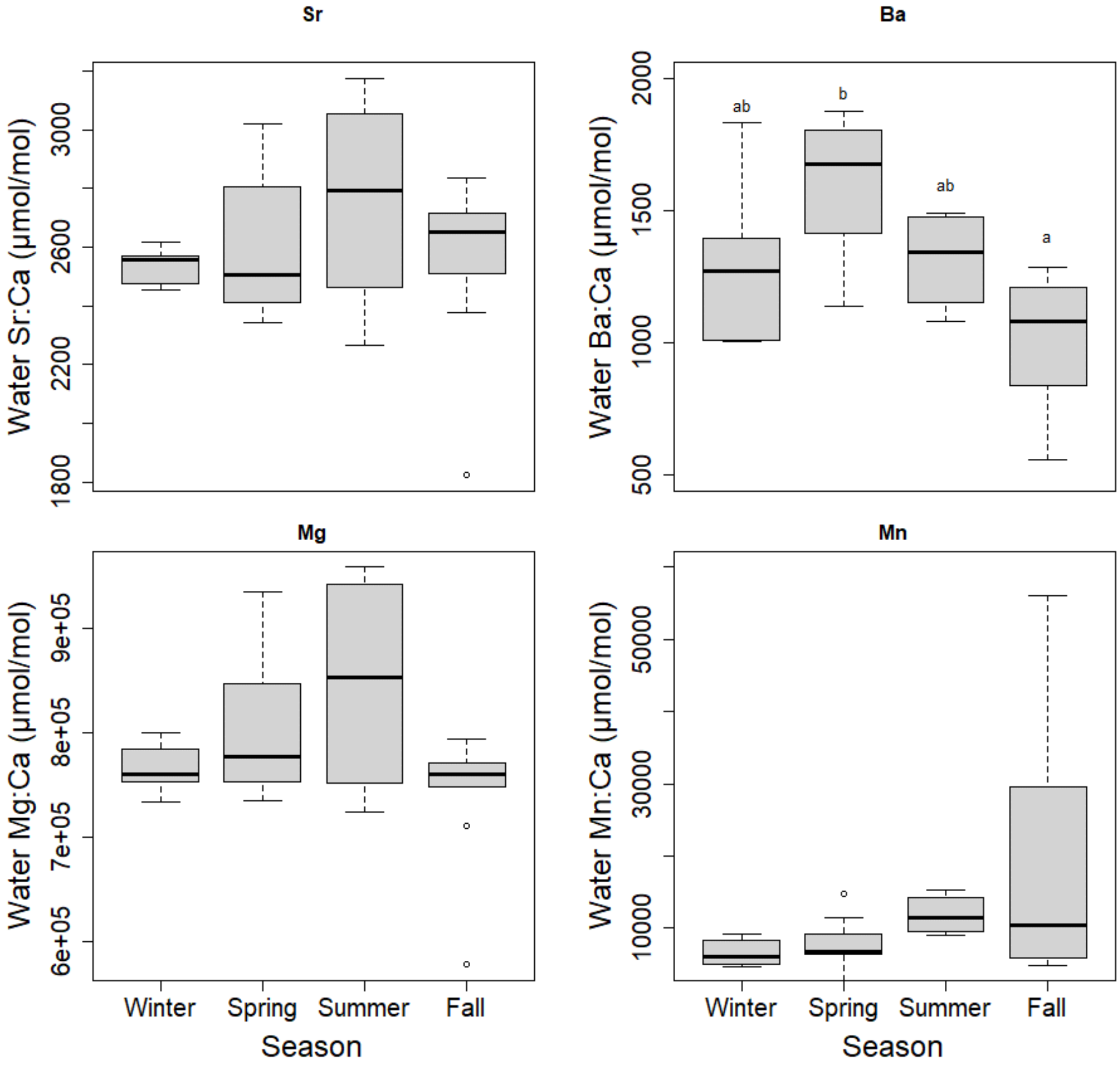

Figure 2.1: Seasonal element (µmol) to calcium (mol) ratios for water samples collected in the Tallapoosa River, Alabama. Boxes represent the median, interquartile range, and outlier limits (1.5 * IQR).

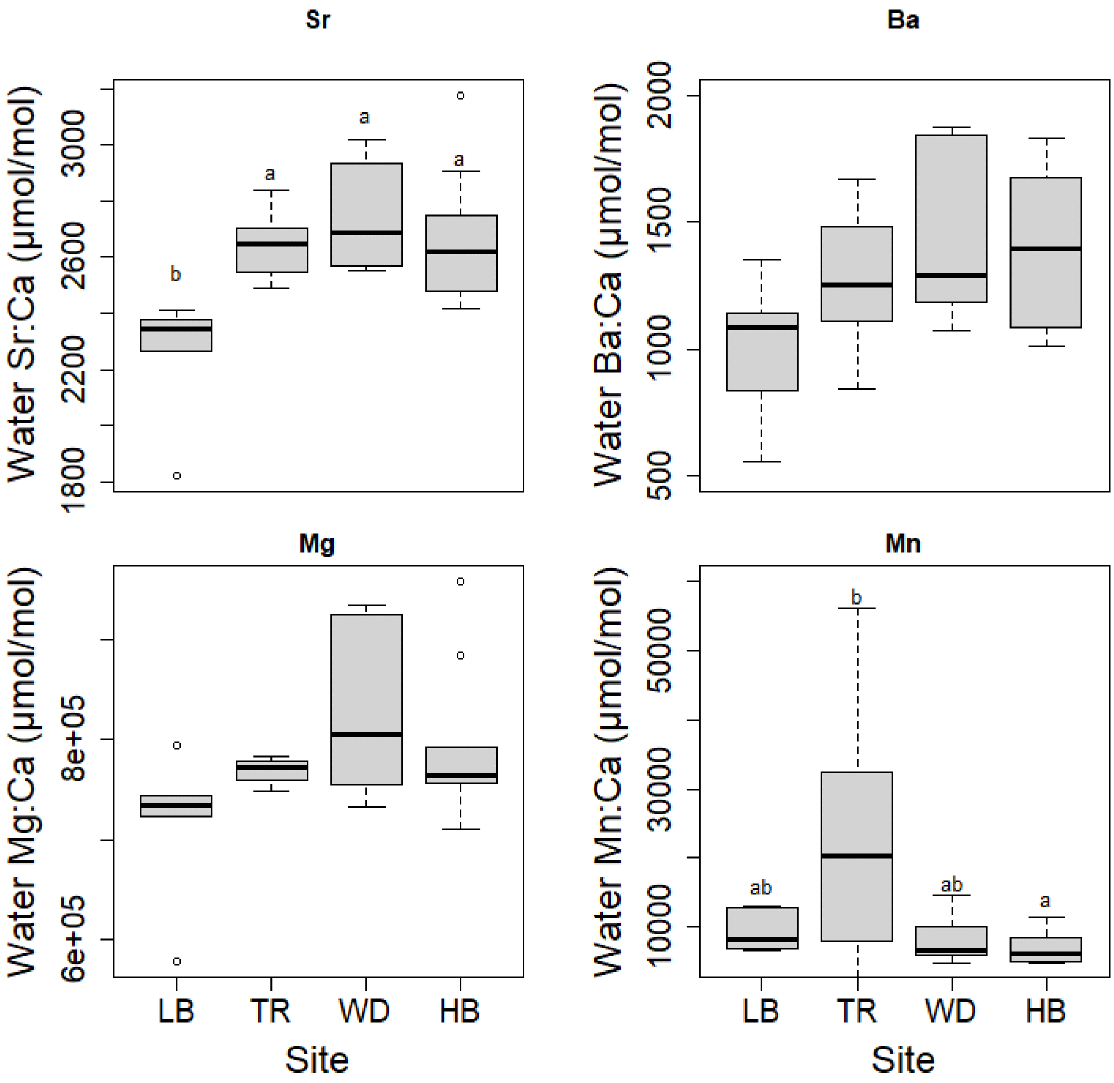

Figure 2.2: Site specific element (µmol) to calcium (mol) ratios for water samples collected in the Tallapoosa River. Sites are: LB = Lee's Bridge, TR = Harris Tailrace, WD = Wadley, HB = Horseshoe Bend. Boxes represent the median, interquartile range, and outlier limits (1.5 * IQR).

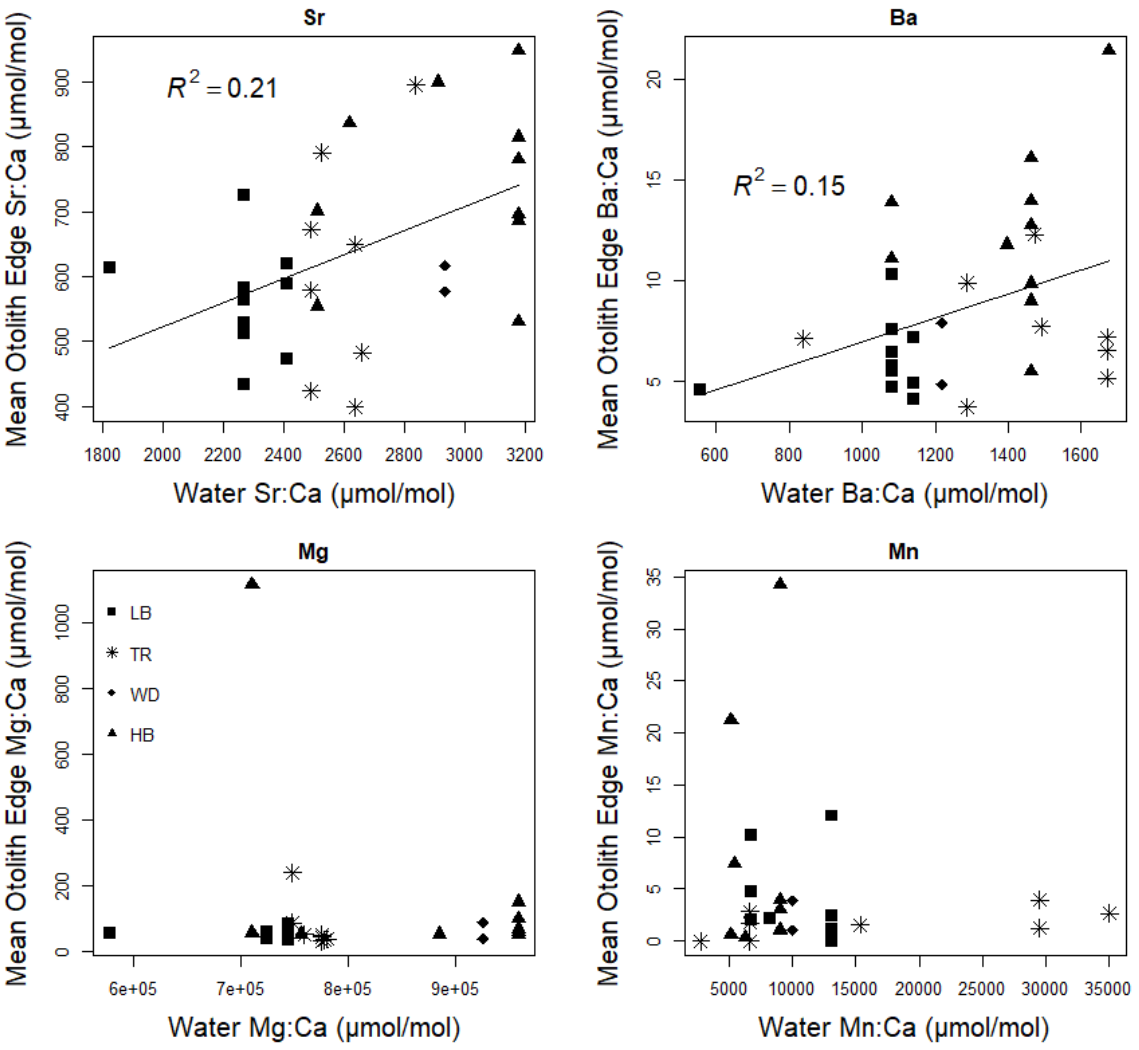

Figure 2.3: Water element (μmol) to calcium (mol) correlations with mean otolith edge elemental ratios for Channel Catfish collected from the Tallapoosa River, Alabama. Sites are: LB Lee’s Bridge, TR = Harris Tailrace, WD = Wadley, HB = Horseshoe Bend.

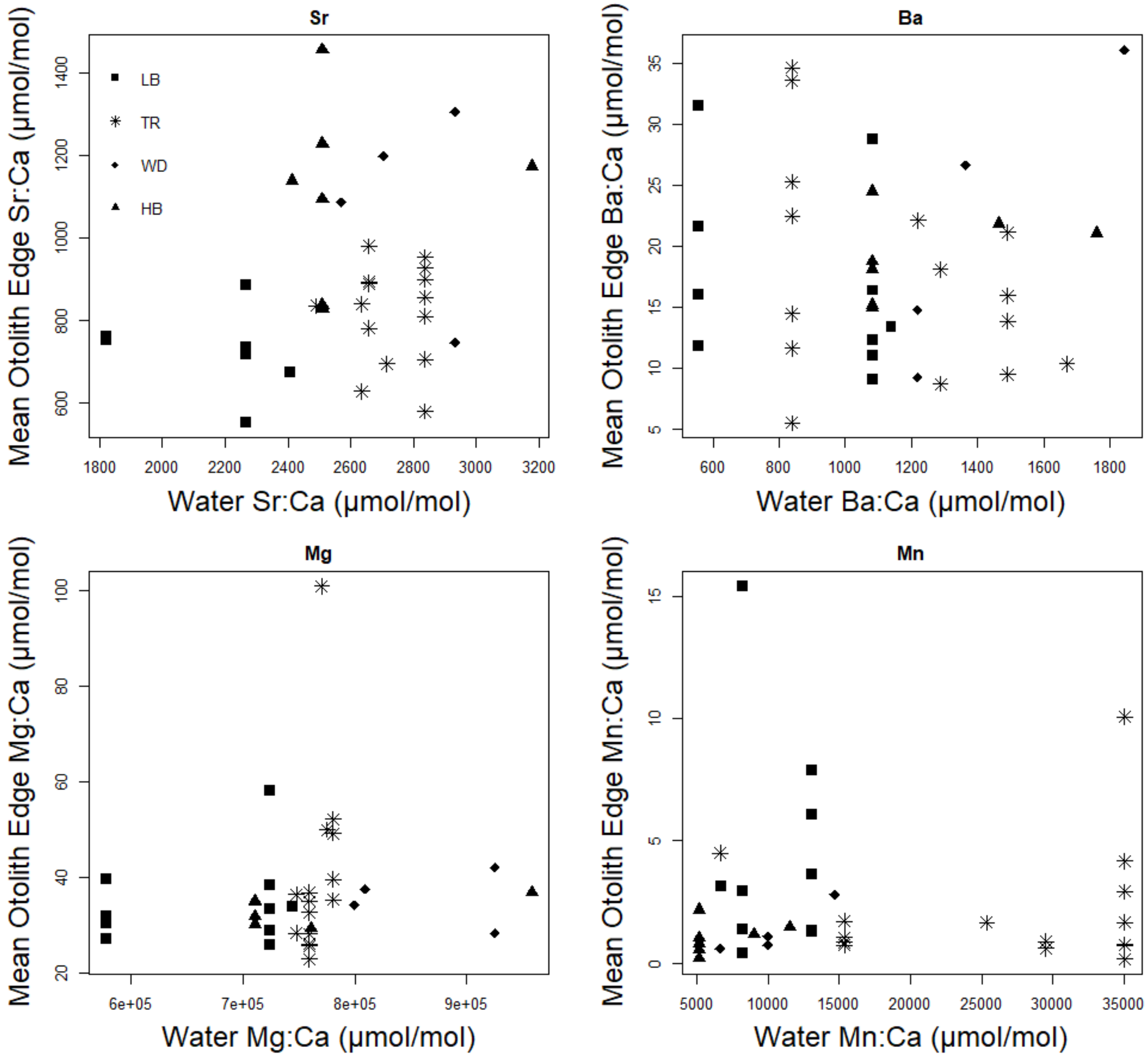

Figure 2.4: Water element (µmol) to calcium (mol) correlations with mean otolith edge elemental ratios for Redbreast Sunfish collected from the Tallapoosa River, Alabama. Sites are: LB = Lee's Bridge, TR = Harris Tailrace, WD = Wadley, HB = Horseshoe Bend.

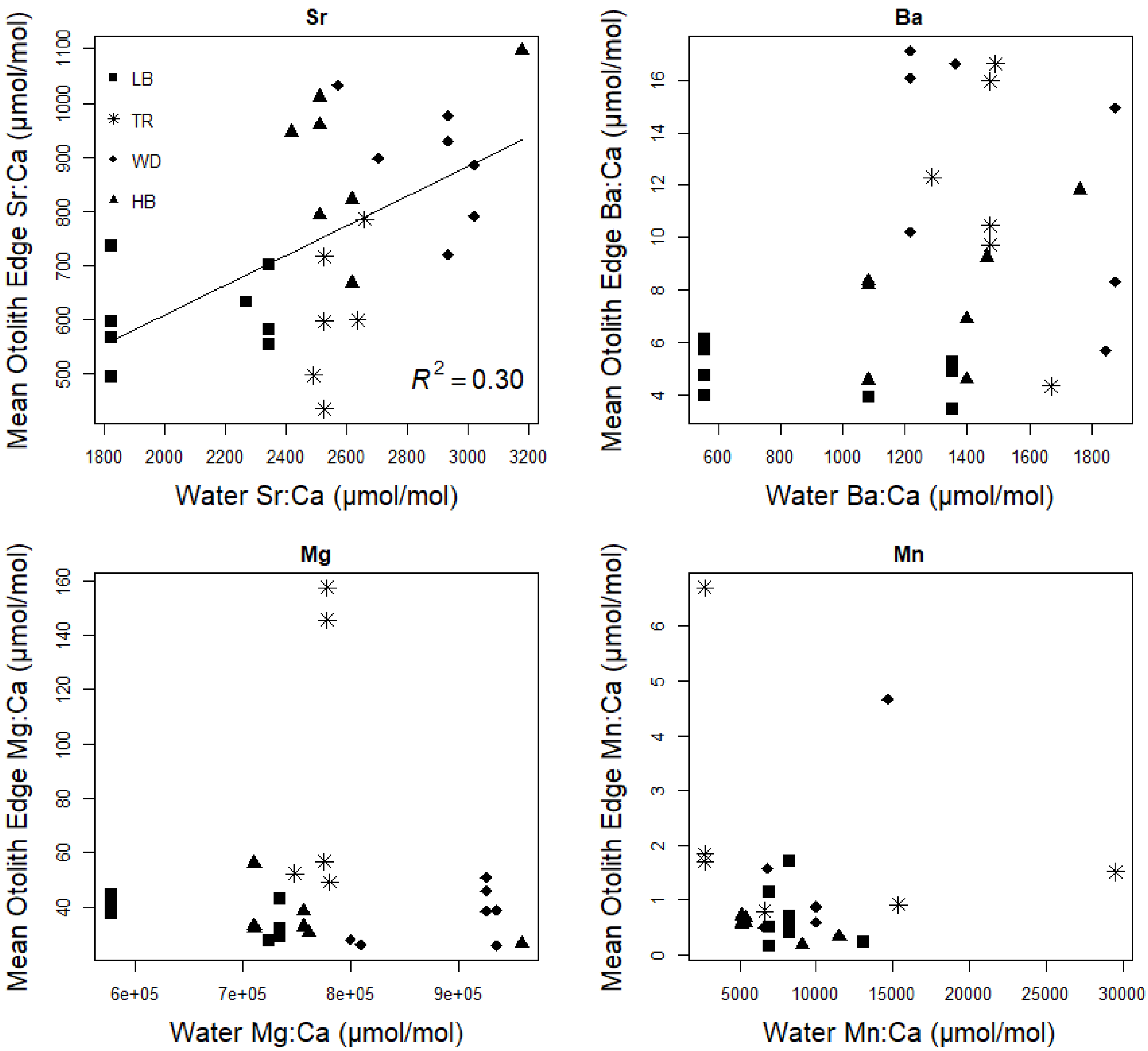

Figure 2.5: Water element (µmol) to calcium (mol) correlations with mean otolith edge elemental ratios for Alabama Bass collected from the Tallapoosa River, Alabama. Sites are: LB = Lee's Bridge, TR = Harris Tailrace, WD = Wadley, HB = Horseshoe Bend.

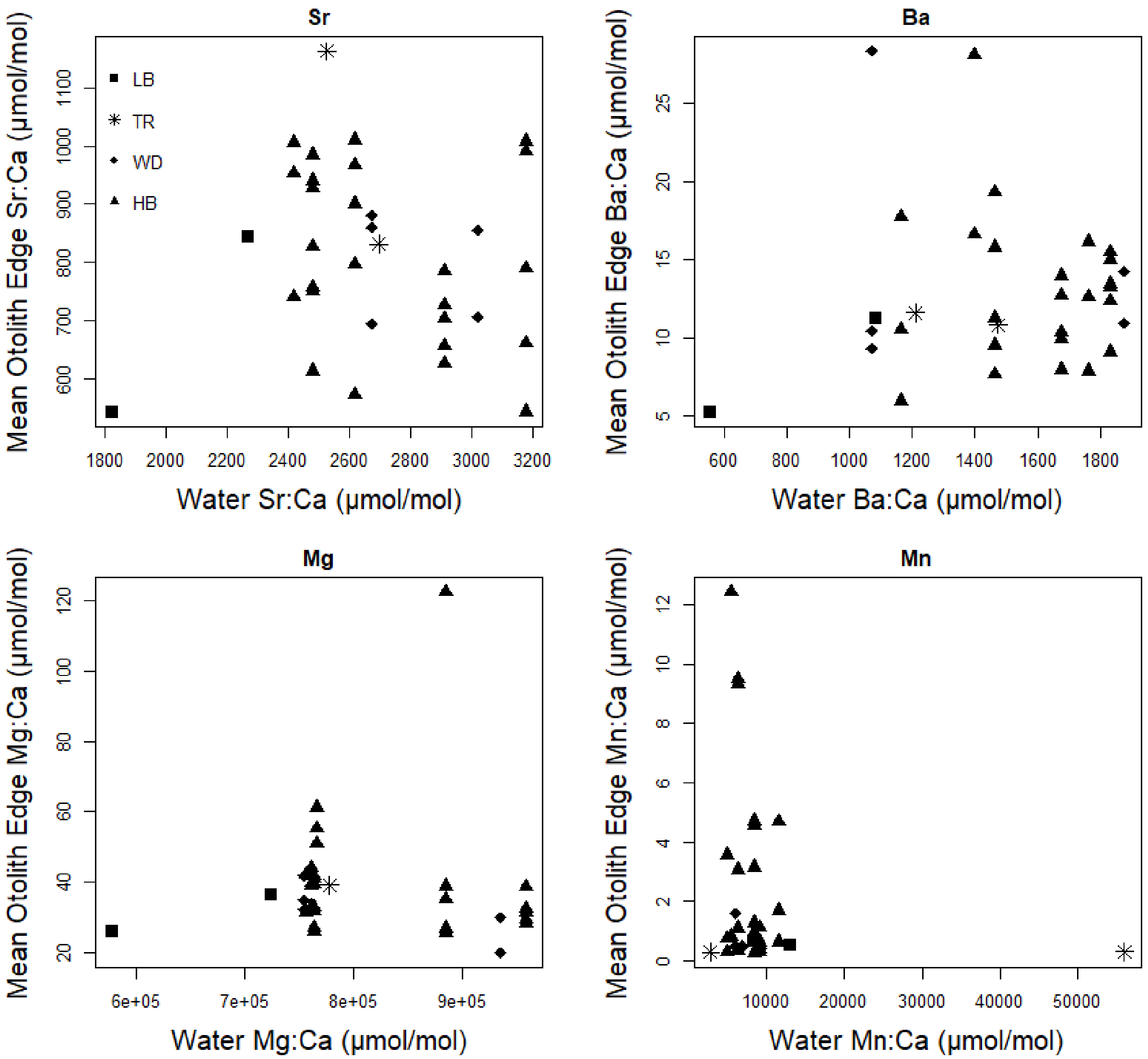

Figure 2.6: Water element (μmol) to calcium (mol) correlations with mean otolith edge elemental ratios for Tallapoosa Bass collected from the Tallapoosa River, Alabama. Sites are: LB = Lee's Bridge, TR = Harris Tailrace, WD = Wadley, HB = Horseshoe Bend.

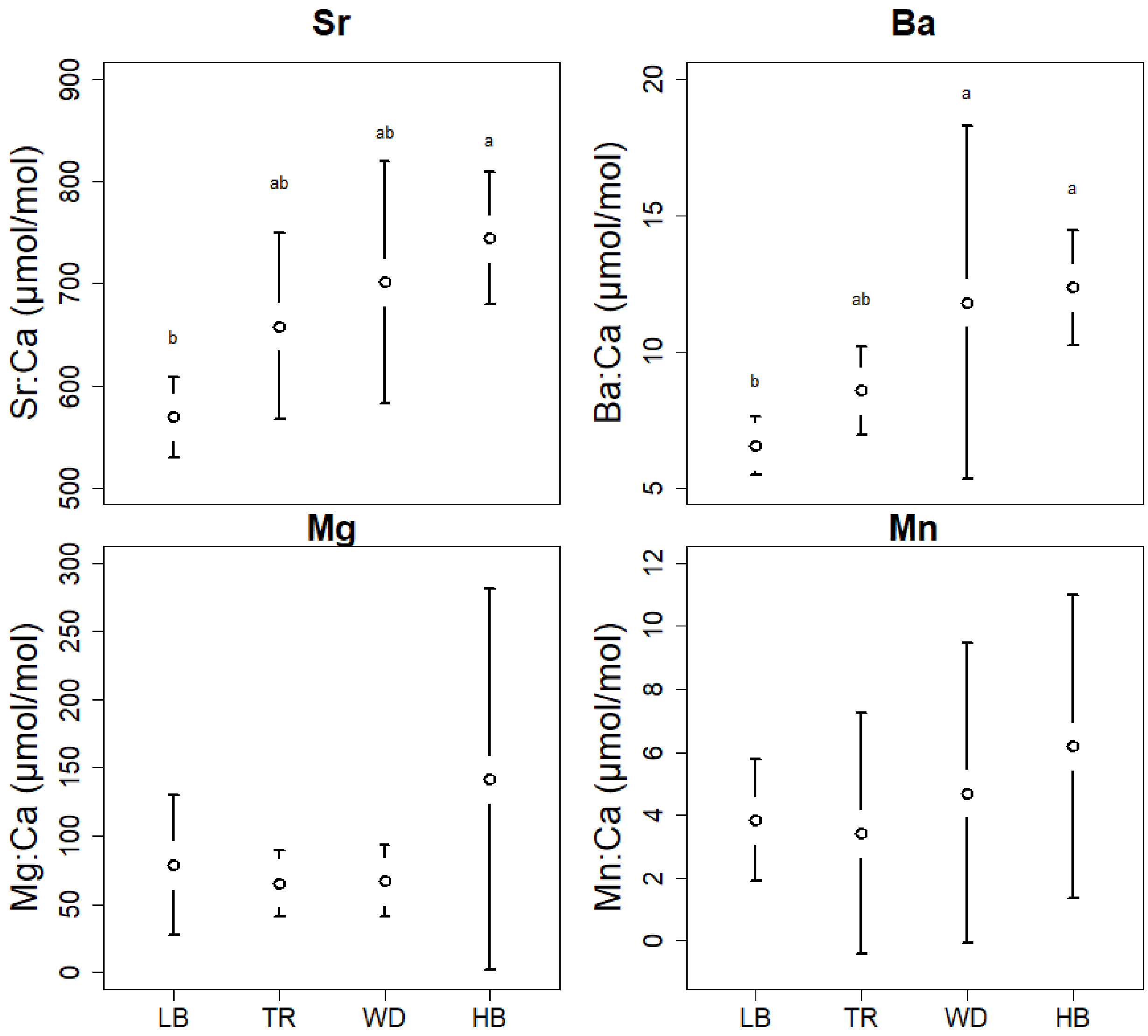

Figure 2.7: Channel Catfish otolith edge trace element means and confidence intervals from fish collected at four sites on the Tallapoosa River, Alabama. Sites are: LB = Lee’s Bridge, TR = Harris Tailrace, WD = Wadley, HB = Horseshoe Bend.

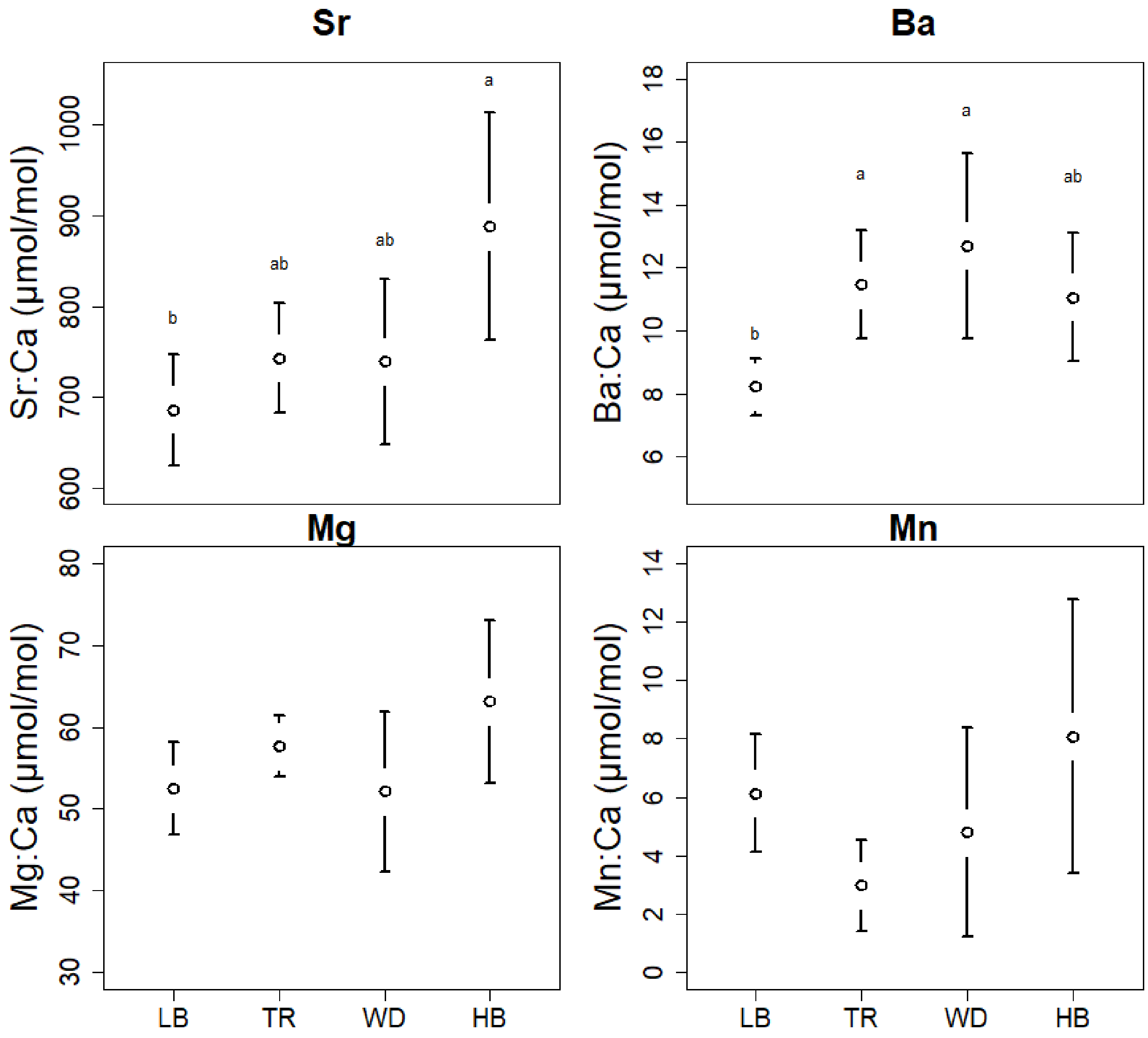

Figure 2.8: Channel Catfish whole transect otolith trace element means and confidence intervals from fish collected at four sites on the Tallapoosa River, Alabama. Sites are: LB = Lee’s Bridge, TR = Harris Tailrace, WD = Wadley, HB = Horseshoe Bend.

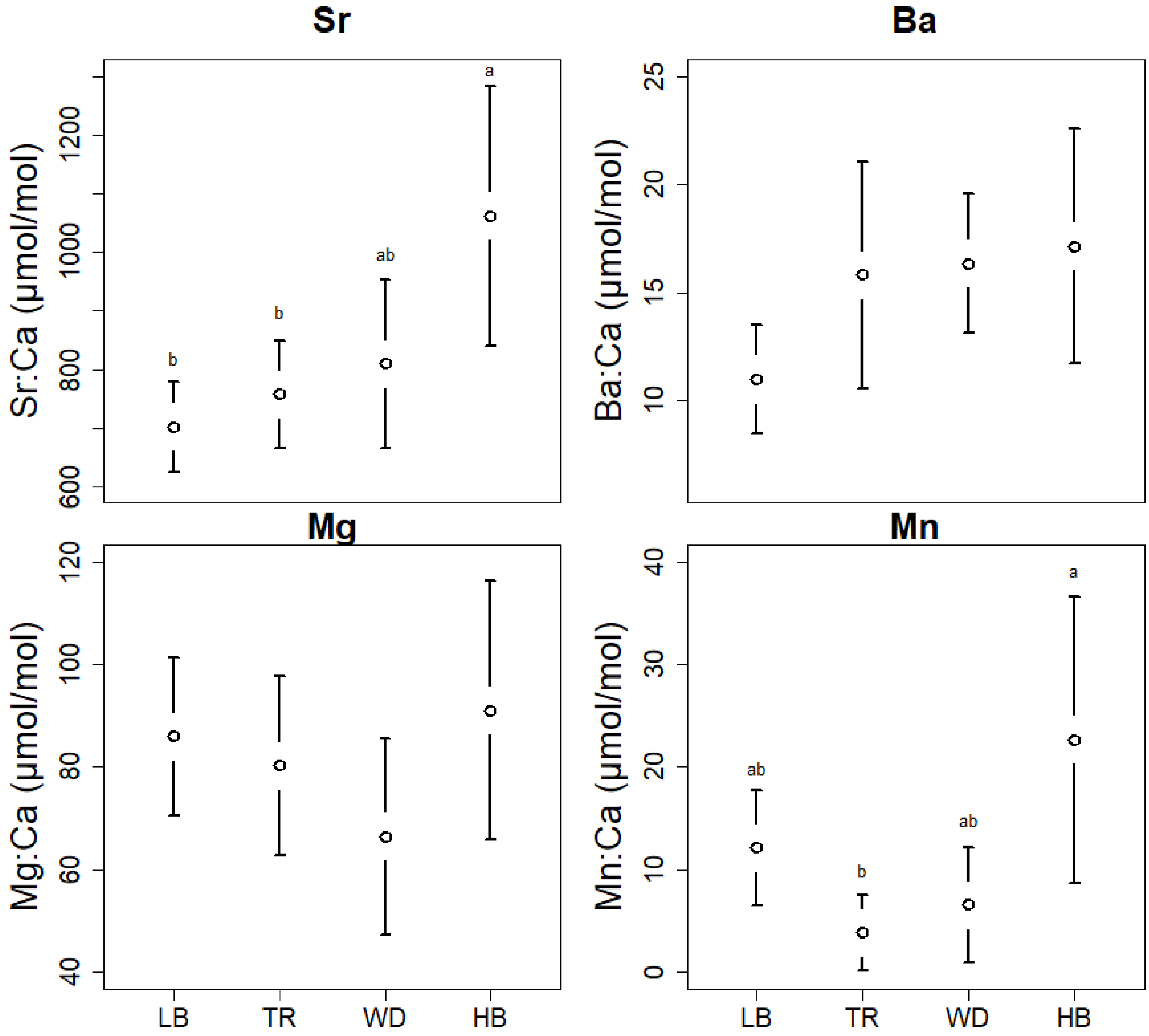

Figure 2.9: Channel Catfish otolith core trace element means and confidence intervals from fish collected at four sites on the Tallapoosa River, Alabama. Sites are: LB = Lee's Bridge, TR = Harris Tailrace, WD = Wadley, HB = Horseshoe Bend.

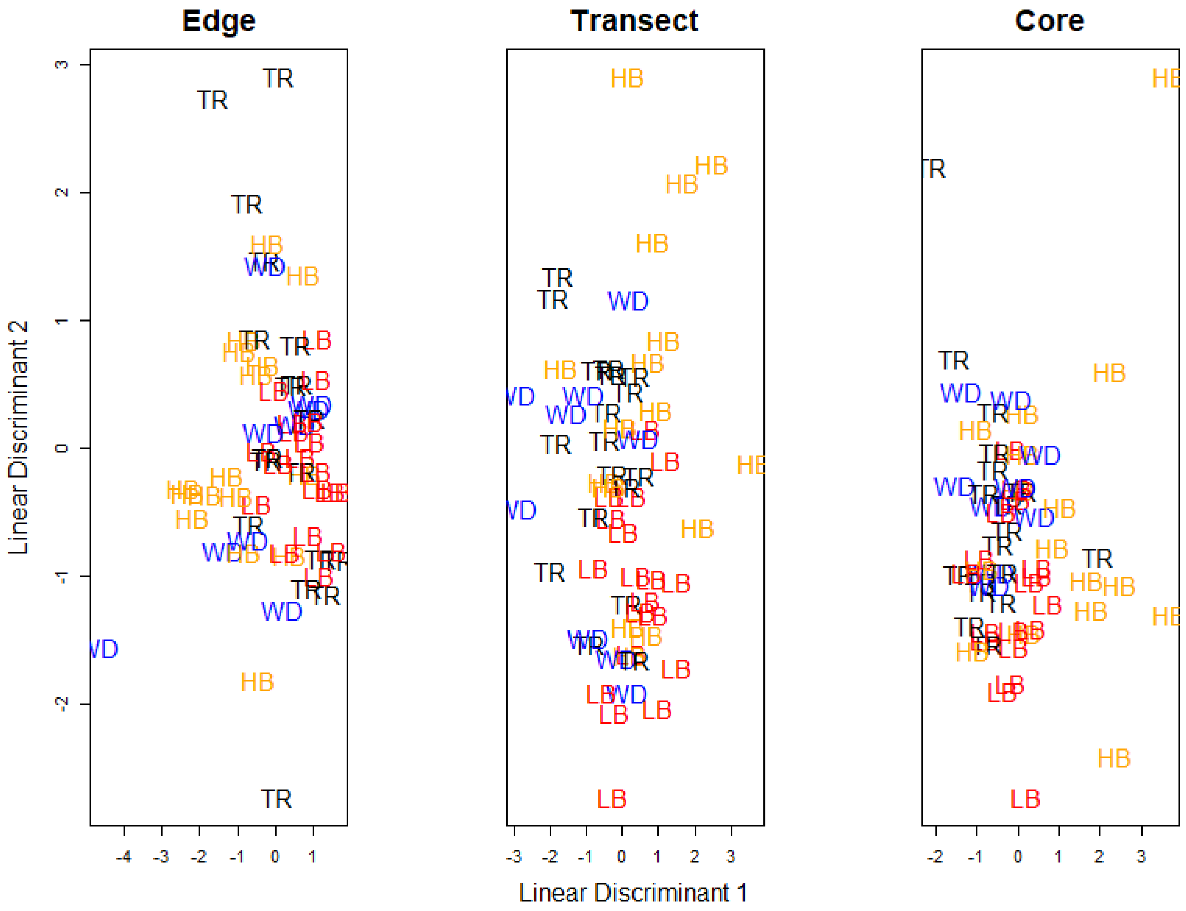

Figure 2.10: Multivariate otolith trace element signatures for Channel Catfish collected from four sites on the Tallapoosa River, AL based on the first two linear discriminant axes.

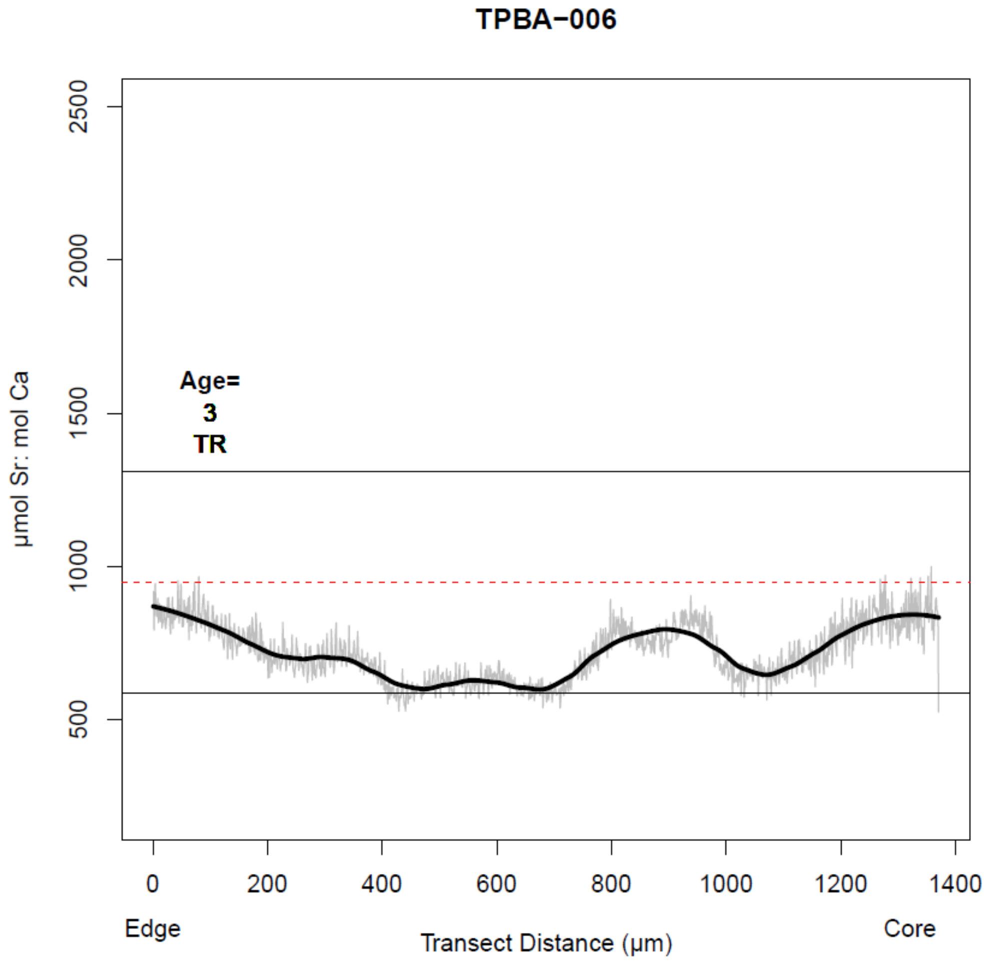

Figure 2.11: Example time series Sr:Ca ratios for an age-3 Tallapoosa Bass collected from the Harris Tailrace exhibiting a consistent Sr:Ca ratio across the entire transect deemed “pattern one”. This potentially indicated lifetime residence at the capture location. The red dashed line represents the mean edge Sr:Ca value for Tallapoosa Bass collected from this site and the solid lines represent the 95% confidence interval around the mean.

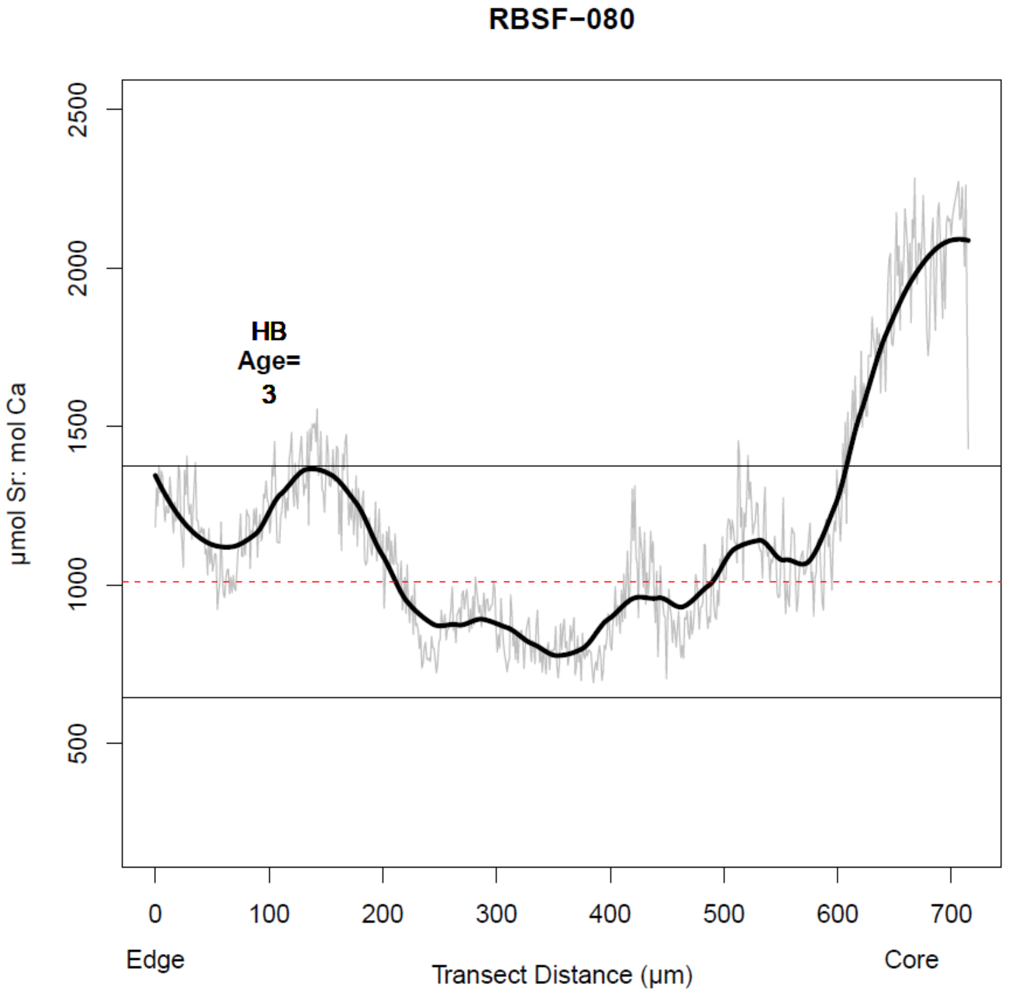

Figure 2.12: Example time series Sr:Ca ratios for an age-3 Redbreast Sunfish collected from Horseshoe Bend exhibiting Sr:Ca ratios outside of the expected range based on the site of capture during the early portion of its life deemed “pattern two”. This potentially indicated that this individual recruited to the site of capture from a different area of the river. The red dashed line represents the mean edge Sr:Ca value for Tallapoosa Bass collected from this site and the solid lines represent the 95% confidence interval around the mean.

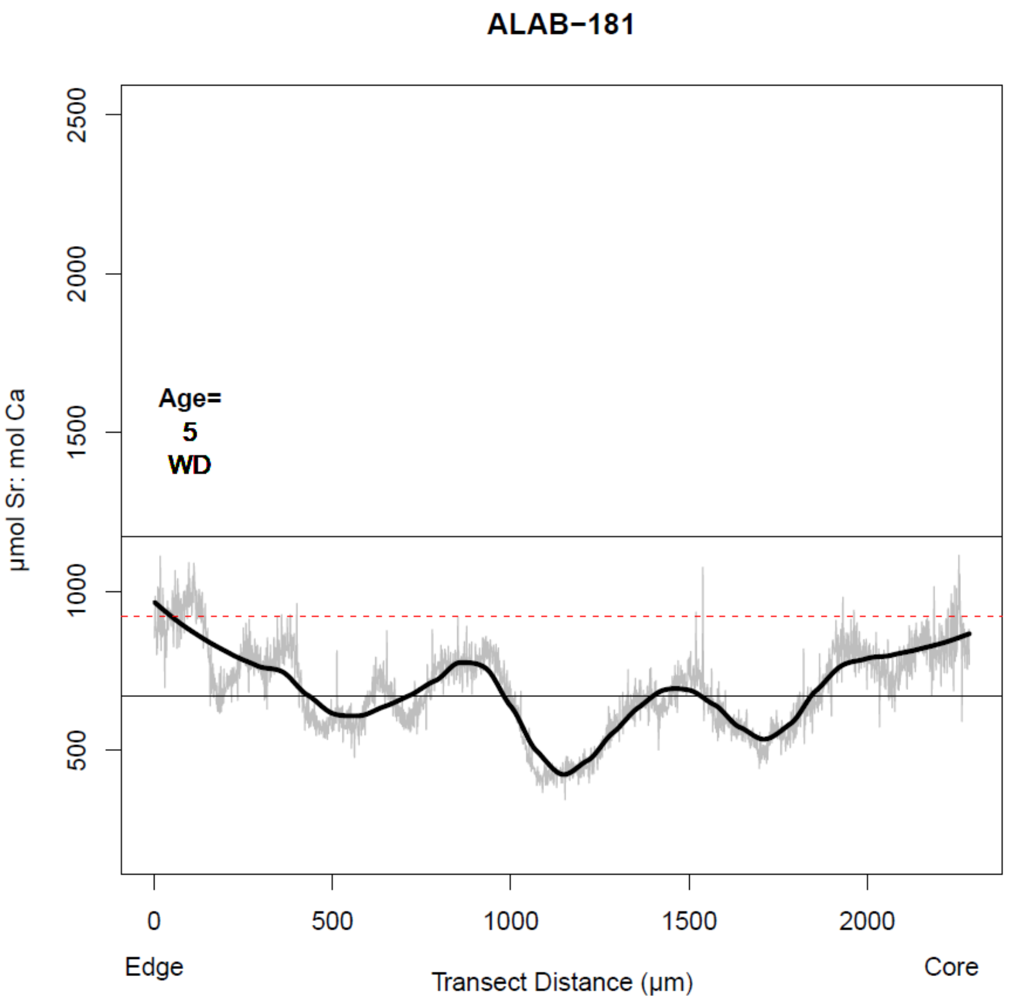

Figure 2.13: Example time series Sr:Ca elemental ratios for an age-5 Alabama Bass collected from Wadley exhibiting multiple deviations from the expected range of Sr:Ca ratios throughout its life deemed “pattern three”. This potentially indicated multiple movements away from, and returning to, the site of capture. The red dashed line represents the mean edge Sr:Ca value for Tallapoosa Bass collected from this site and the solid lines represent the 95% confidence interval around the mean.

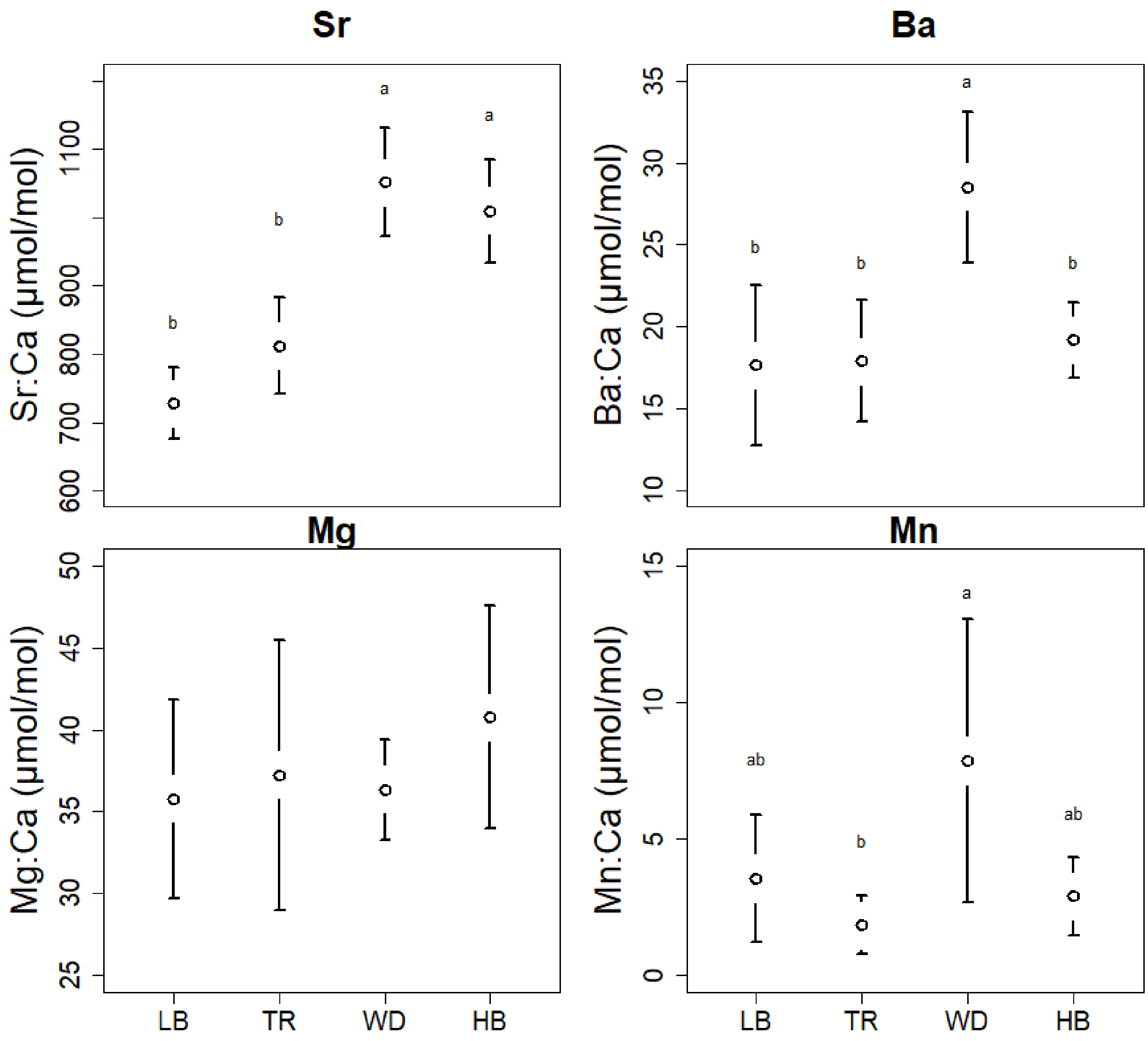

Figure 2.14: Redbreast Sunfish otolith edge trace element means and confidence intervals from fish collected at four sites on the Tallapoosa River, Alabama. Sites are: LB = Lee's Bridge, TR = Harris Tailrace, WD = Wadley, HB = Horseshoe Bend.

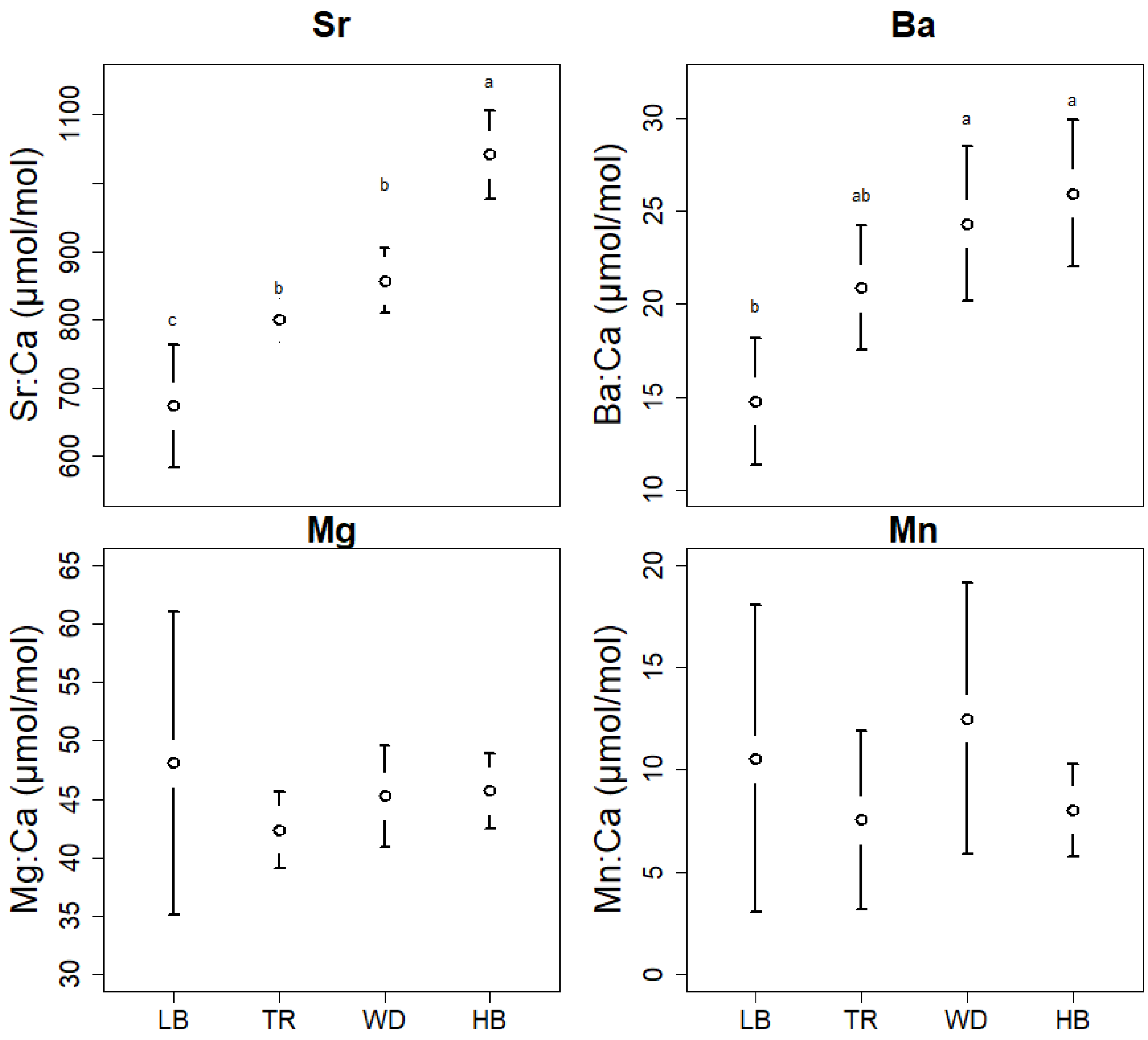

Figure 2.15: Redbreast Sunfish whole transect otolith trace element means and confidence intervals from fish collected at four sites on the Tallapoosa River, Alabama. Sites are: LB = Lee’s Bridge, TR = Harris Tailrace, WD = Wadley, HB = Horseshoe Bend.

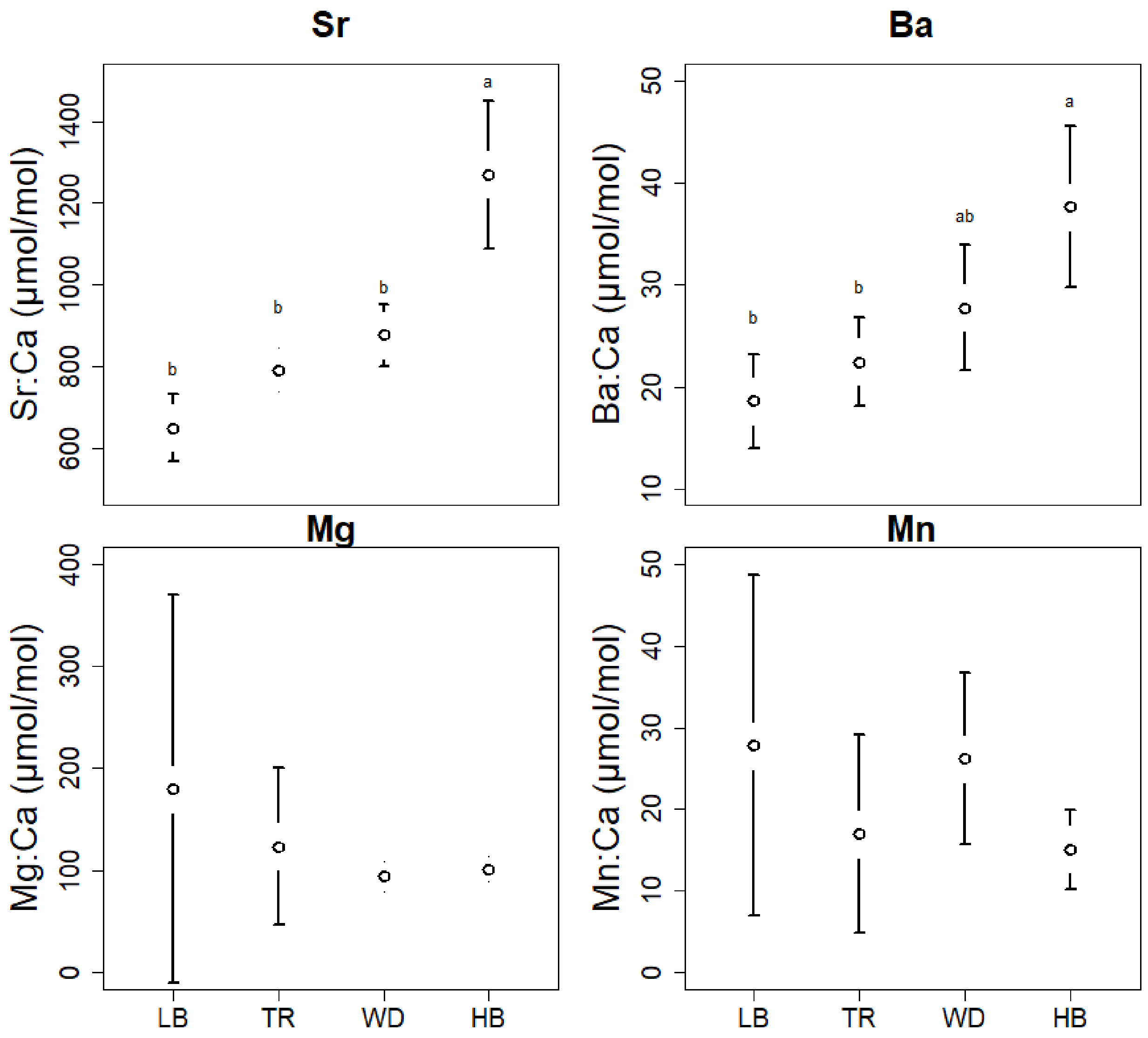

Figure 2.16: Redbreast Sunfish otolith core trace element means and confidence intervals from fish collected at four sites on the Tallapoosa River, Alabama. Sites are: LB = Lee’s Bridge, TR = Harris Tailrace, WD = Wadley, HB = Horseshoe Bend.

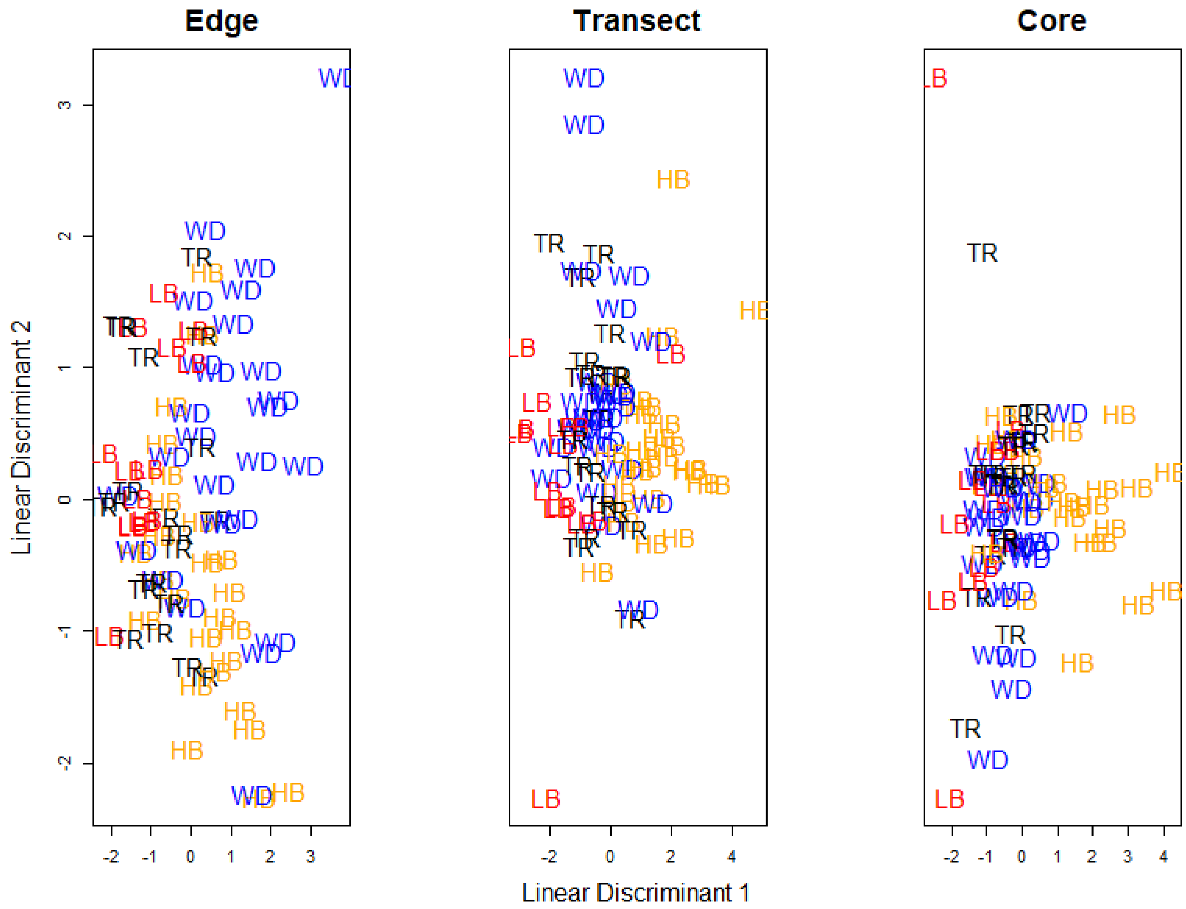

Figure 2.17: Multivariate otolith trace element signatures for Redbreast Sunfish collected from four sites on the Tallapoosa River, AL based on the first two linear discriminant axes.

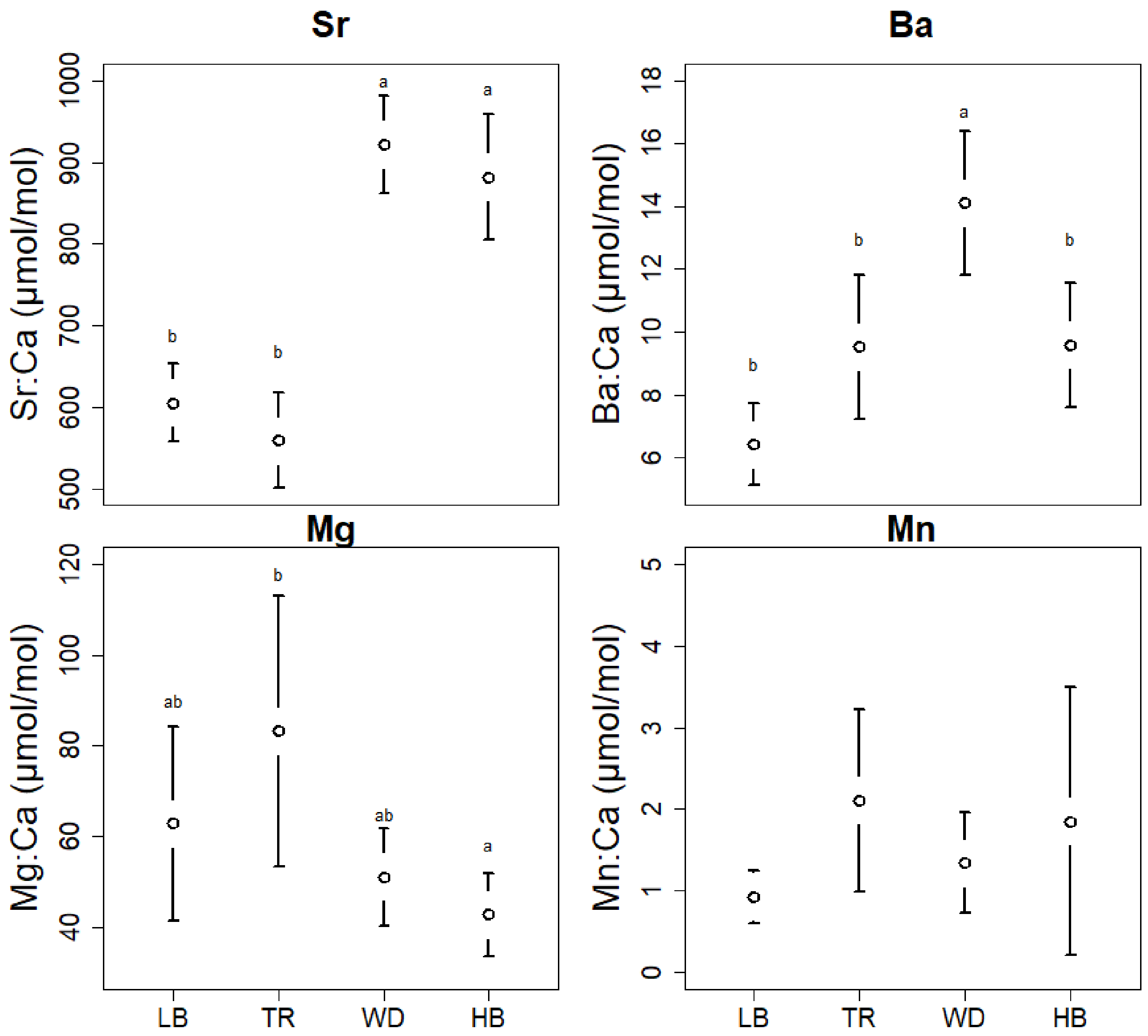

Figure 2.18: Alabama Bass otolith edge trace element means and confidence intervals from fish collected at four sites on the Tallapoosa River, Alabama. Sites are: LB = Lee's Bridge, TR = Harris Tailrace, WD = Wadley, HB = Horseshoe Bend.

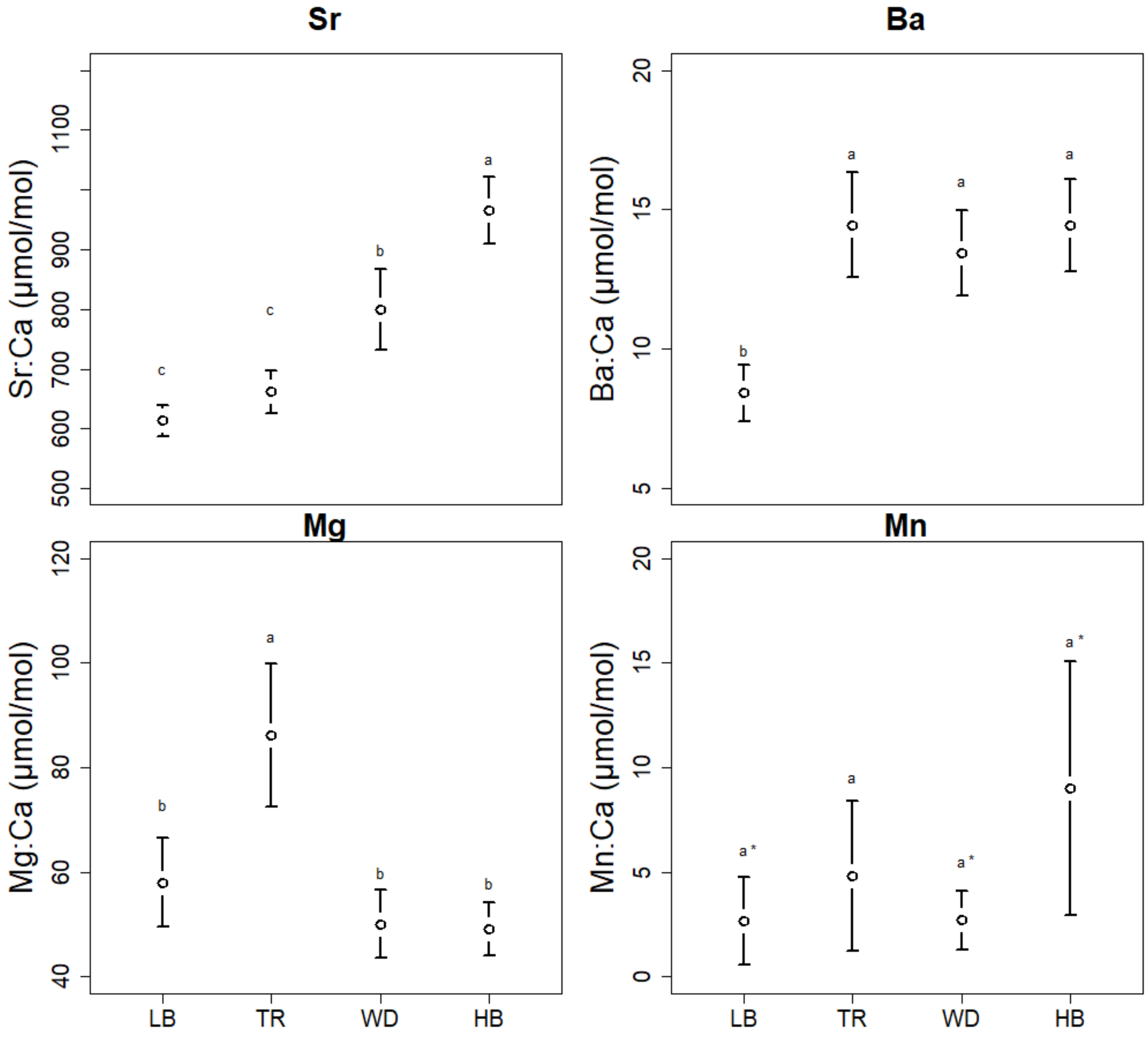

Figure 2.19: Alabama Bass whole transect otolith trace element means and confidence intervals from fish collected at four sites on the Tallapoosa River, Alabama. Sites are: LB = Lee's Bridge, TR = Harris Tailrace, WD = Wadley, HB = Horseshoe Bend.

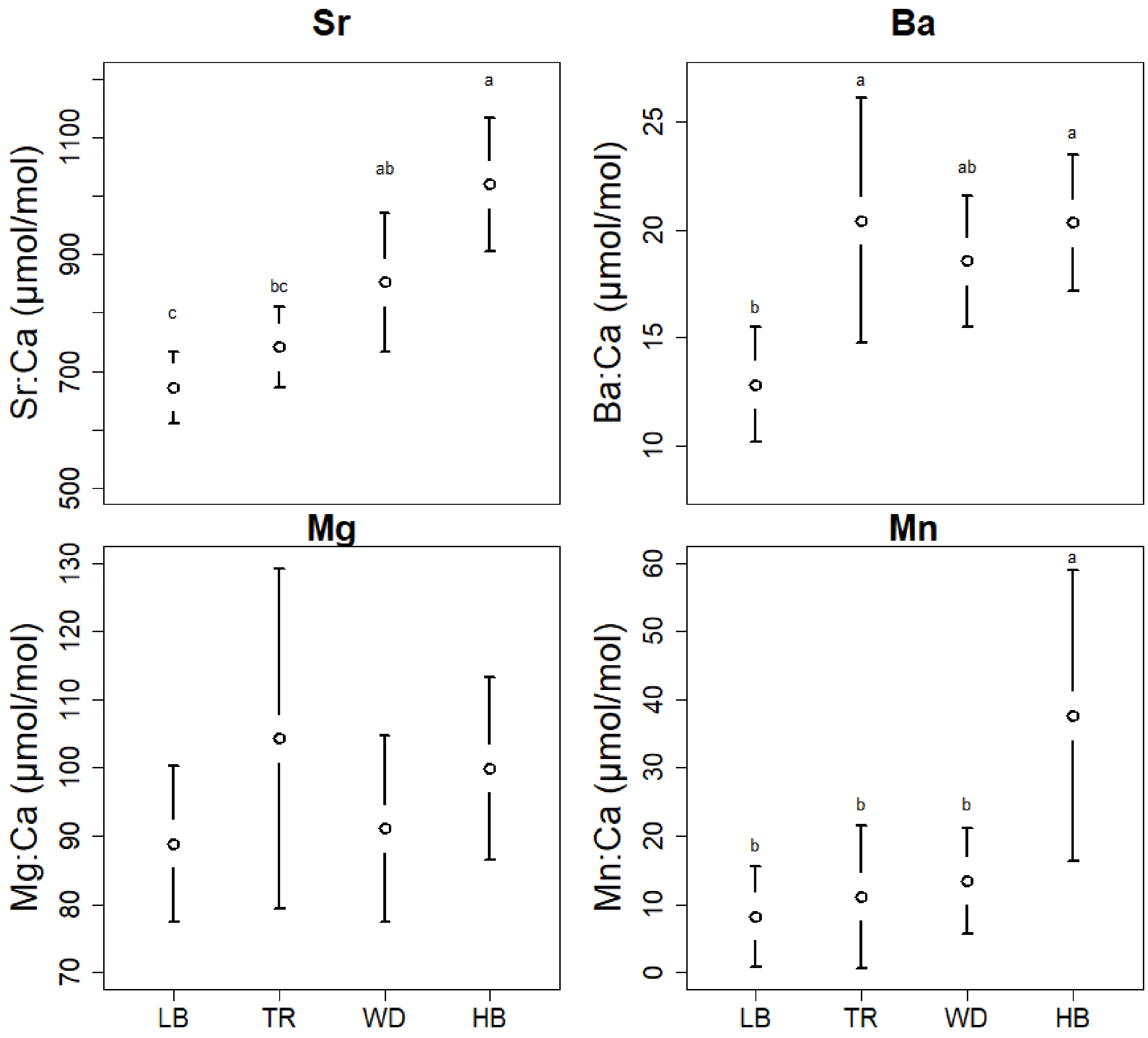

Figure 2.20: Alabama Bass otolith core trace element means and confidence intervals from fish collected at four sites on the Tallapoosa River, Alabama. Sites are: LB = Lee's Bridge, TR = Harris Tailrace, WD = Wadley, HB = Horseshoe Bend.

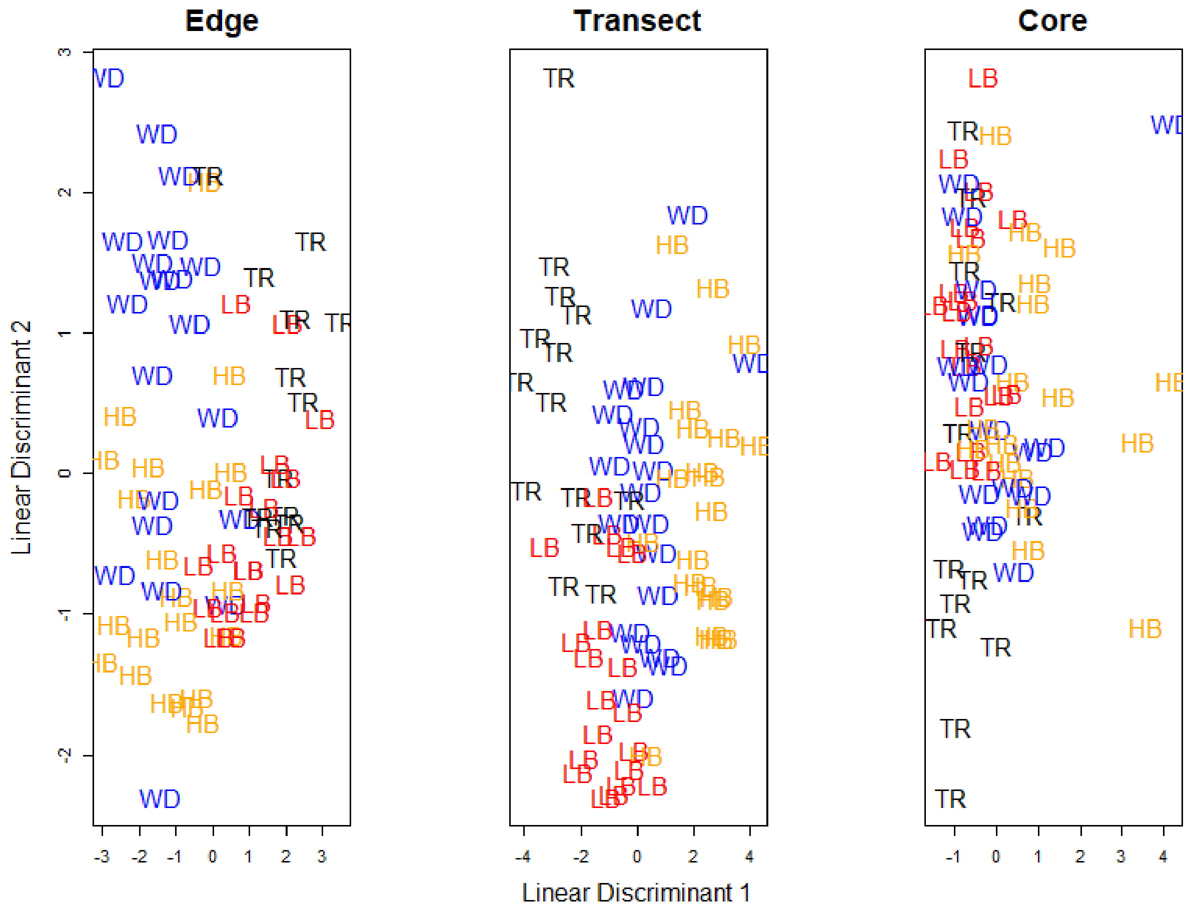

Figure 2.21: Multivariate otolith trace element signatures for Alabama Bass collected from four sites on the Tallapoosa River, AL based on the first two linear discriminant axes.

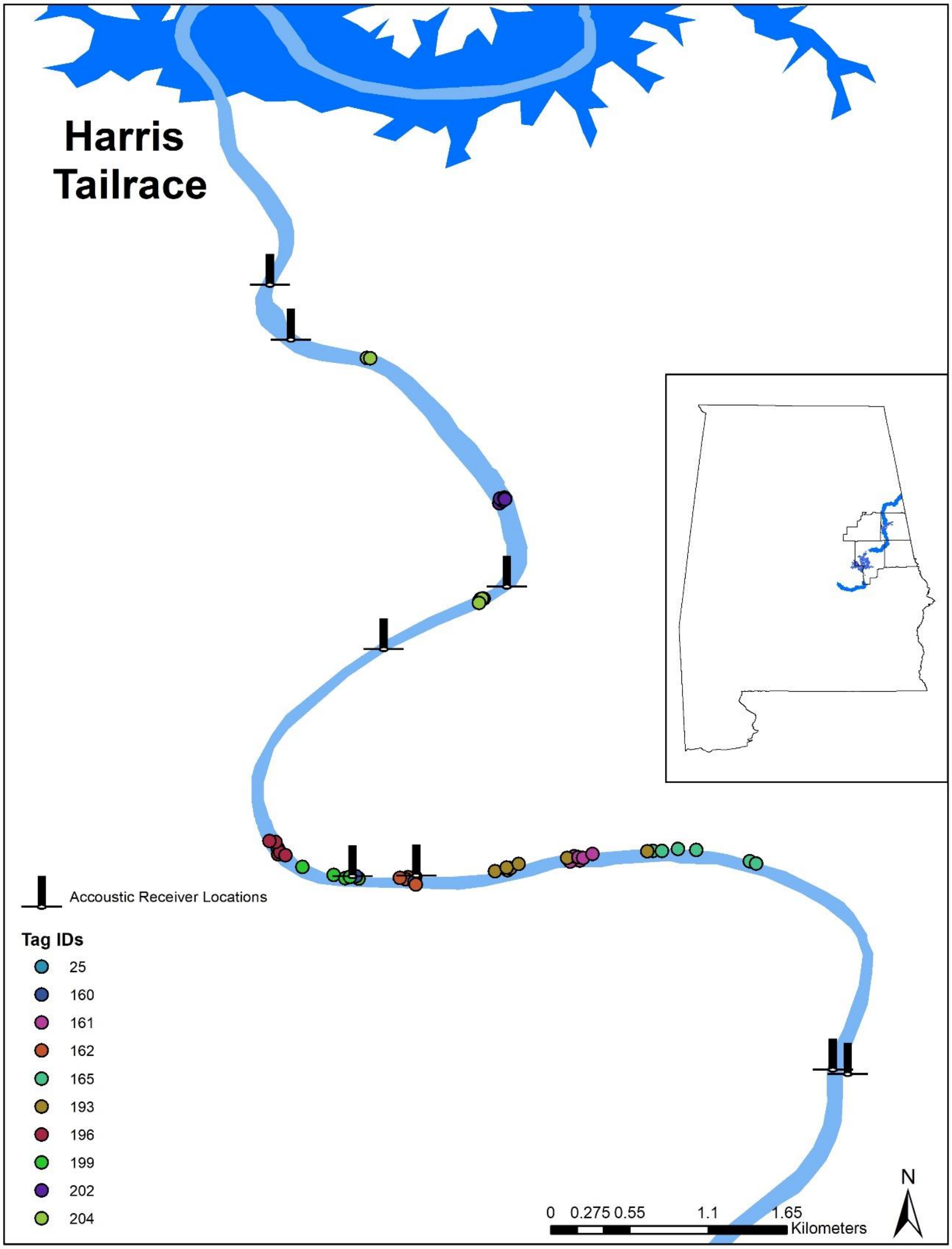

Figure 2.22: Map of each detected fish’s position (maximum signal strength) during each manual tracking effort. Tag IDs are described in Table 2.9.

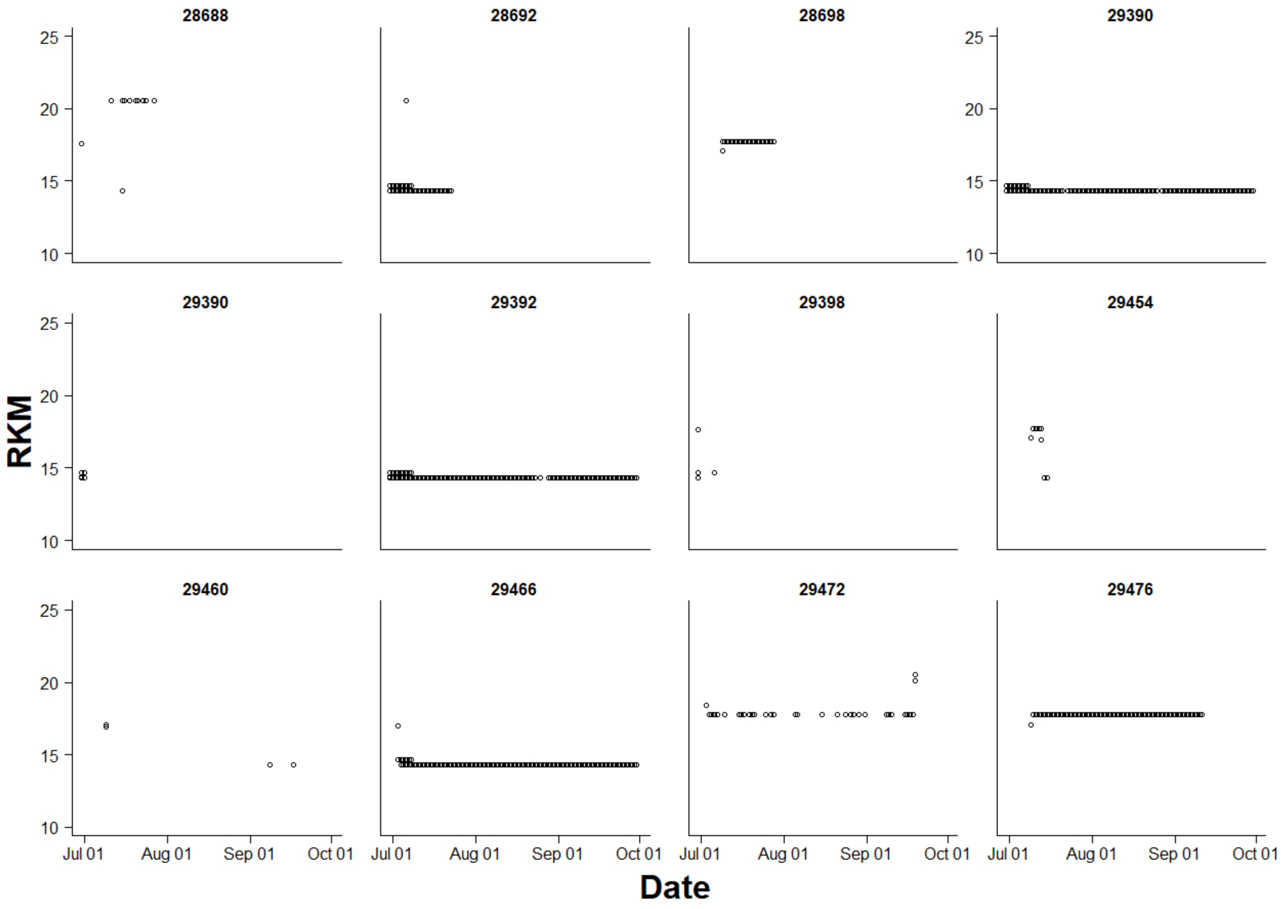

Figure 2.23: Graph of fish position (RKM upstream of AL-77 in Wadley, AL) by date for each fish detected by a stationary acoustic receiver array in the Tallapoosa River, Alabama.

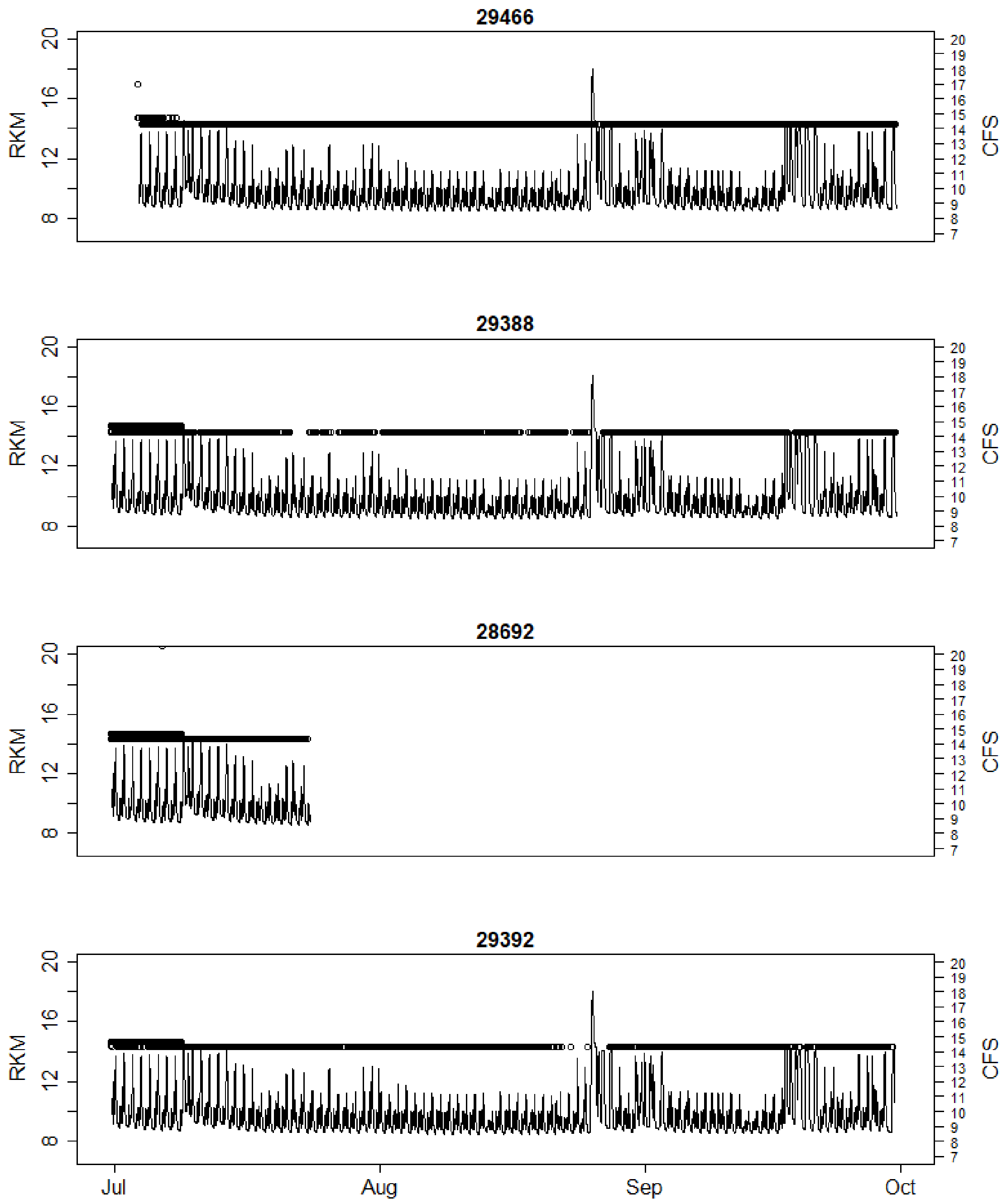

Figure 2.24: Plots of fish position (RKM upstream of the state road 77 bridge in Wadley, AL) with relation to flow (cubic feet per sec) readings from the USGS Gage (02414300) located at county road 15 in Malone, AL for four Alabama Bass tagged with combined acoustic radio tags in the Tallapoosa River, Alabama.

Conclusions

Variation is natural in river systems and there are several conceptual descriptions regarding how energy inputs, nutrient inputs, biodiversity, etc. change along a river continuum (Vannote et al. 1980; Ward and Stanford 1983; Ward and Stanford 1995). Regardless of these longitudinal variations, some overlaps in species assemblage are still expected among sites that are not separated by a natural or anthropogenic barrier (Ward and Stanford 1983). In my study, we observed little overlap in the fish assemblages among sites within ordinal space (i.e., groups of species determined using multivariate ordination); indicating that these assemblages were spatially variable in composition. This, combined with the upstream-downstream site blocks present within my data, suggest that the effects of Harris Dam, on the downstream habitat and/or discharge/temperature regime, play a role in the observed differences.

Large dams can nearly or, in some cases, completely eliminate fish passage and in instances where flow and temperature regulation is present, dams can also affect the movements of fish in the surrounding river reaches. In these scenarios, from a conservation perspective, mitigation is likely necessary in the form of fish passage structures or dam operation changes. While I observed little movement within my telemetry data, trace element analyses suggested Harris Dam affects the dispersal of juvenile/larval fishes in the downstream reach. Additionally, based on trace element signatures movement by adult fish increased with distance from the dam for all species. In this work, I observed effects of Harris Dam at both the assemblage level (chapter one) and the individual level (chapter two); whether this is acceptable under the Green Plan may need to be evaluated.

Numerous other studies on the Tallapoosa River have attempted to quantify the effects of Harris Dam on fish assemblages in the downstream area with varying success (Kinsolving and Bain 1993; Travnichek and Maceina 1994; Irwin and Hornsby 1997; Freeman et al. 2005; Irwin et al. 2019). These studies span variable temporal scales ranging from one day (Irwin and Hornsby 1997) to 12 years (Irwin et al. 2019) and were largely limited to summer and fall sampling. These studies also used variable sampling methods to collect fishes including rotenone (Irwin and Hornsby 1997), prepositioned electrofishing grids (Kinsolving and Bain 1993; Travnichek and Maceina 1994; Bowen et al. 1998; Irwin et al. 2019), and boat electrofishing (Travnichek and Maceina 1994; this study) the biases of which all must be considered (Dunn an Paukert 2020). Comparing across these studies is difficult due to the different portions of the year considered and the different gear types used (and their corresponding biases). To better understand how Tallapoosa River fish assemblages change with respect to regulation by Harris Dam over time, longer term studies such as Irwin et al. (2019) should be conducted following more robust sampling protocols such as those identified in Dunn and Paukert (2020).