**The Relationship Between Shoreline Development and Resident
Fish Communities in Lake Martin, Alabama**

by

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Abstract

Many reservoirs in the southeastern U.S. have experienced an increase in human development along its shorelines. This development often includes armoring the shoreline with either bulkheads or riprap to reduce erosion and protect property. My objectives were to examine how human shoreline development affects the fish community in Lake Martin by quantifying fish abundance and species composition, as well as abundance and species composition of larval fish, aquatic macroinvertebrates, zooplankton, and to quantify water quality characteristics at four different shoreline habitat types (bulkhead, riprap, bulkhead with riprap at the base, and undeveloped). I found no differences in abiotic factors or lower trophic levels that I measured. However, fish abundance and community structure did differ significantly among shoreline habitat types. Fish abundances were highest at sites containing riprap while species richness and species diversity both tended to be highest at undeveloped sites.

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Introduction

Over the past several decades, residential growth has increased rapidly on lakes and reservoirs throughout the U.S. This development in watersheds and riparian zones has the potential to affect littoral zones in our aquatic systems (Schindler et al. 2000; Radomski and Goeman 2001; Scheuerell and Schindler 2004; Francis and Schindler 2006; Rosenberger et al. 2008). Littoral areas provide valuable habitat for many fish species, providing spawning habitat (Ehlinger 1997), nursery areas for young-of-year fish (Meals and Miranda 1991), refuge from predators (Mittelbach 1981; Savino and Stein 1982; Johnson et al. 1988; Radomski and Goeman 2001), and important foraging areas (Mittelbach 1981; Conrow et al. 1990; Winfield 2004). Human impacts on littoral zones can include nutrient loading (Dillon and Rigler 1975; Dillon et al. 1994), removal of trees and therefore woody debris (Christensen et al. 1996), removal of aquatic vegetation (Radomski and Goeman 2001), construction of piers and boat ramps (Radomski et al. 2010), and construction of erosion control structures (e.g., bulkheads, riprap). When left undisturbed, littoral zones often provide habitat diversity that is beneficial to the aquatic community (Beauchamp et al. 1994; Sass et al. 2006a; Smokorowski and Pratt 2007; Ahrenstorff et al. 2009).

Tree removal from the riparian zone decreases coarse woody debris deposition to littoral zones (Christensen et al. 1996; Francis and Schindler 2006). Coarse woody debris has been shown to provide habitat complexity to near shore areas for many fish (Schindler and Sheuerell 2002). Francis et al. (2007) found that coarse woody debris may retain important organic matter in littoral zones, where it can enter the detrital energy pathway, suggesting that a loss of this material may have significant consequences to the entire aquatic food web. Coarse woody debris density was found to be positively correlated with growth rates of the dominant fish species in

lakes of the U.S. upper Midwest (Christensen et al. 1996; Schindler et al. 2000), as well as with fish species richness and abundance (Newbrey et al. 2005). The presence or absence of coarse woody debris may result in behavioral changes for certain fish species (Scheuerell and Schindler 2004). Ahrenstorff et al. (2009) suggested that largemouth bass *Micropterus salmoides* changed foraging behavior depending on abundance of coarse woody debris, switching from a sit-and-wait approach when coarse woody debris was abundant to actively seeking prey when coarse woody debris was rare. This added energetic cost, combined with fewer prey species present when coarse woody debris was rare, may reduce largemouth bass growth potential. Sass et al. (2006b) also found slower growth in largemouth bass after a 75% removal of coarse woody debris, as well as a decline in abundance of yellow perch *Perca flavescens*, presumably due to increased predation.

Loss of littoral aquatic vegetation is another effect that can be associated with anthropogenic development (Radomski and Goeman 2001). Similar to coarse woody debris, aquatic macrophytes provide habitat complexity that can be important to the aquatic community (Savino and Stein 1982; Durocher et al. 1984; Conrow et al. 1990; Bryan and Scarnecchia 1992). Higher species richness and total fish abundance occurs in naturally vegetated areas compared to areas devoid of vegetation (Bryan and Scarnecchia 1992; Trial et al. 2001). In addition, vegetated areas can be important nursery habitat for a lake fish community (Conrow et al. 1990). Savino and Stein (1982) demonstrated that predation by largemouth bass on bluegill decreased as density of submerged vegetation increased. Durocher et al. (1984) found Texas reservoirs with less than 20% submerged vegetation had reduced recruitment and a lower standing stock of largemouth bass than reservoirs with 20% submerged vegetation. When submersed aquatic vegetation in Lake Conroe, TX was eliminated, Bettoli et al. (1993) found some fish species

significantly decreased in abundance and biomass (e.g., dollar sunfish *Lepomis marginatus*, warmouth *L. gulosis*, spotted sunfish *L. punctatus*), while others increased (e.g., bullhead minnow *Pimephales vigilax*, blacktail shiner *Cyprinella venusta*, channel catfish *Ictalurus punctatus*) or showed no change (e.g., common carp *Cyprinus carpio*, gizzard shad *Dorosoma cepedianum*, threadfin shad *D. petenense*).

Developed shorelines often contain structures to improve access to the water as well as to minimize erosion. A common practice is the construction of piers and barriers at the water-land interface. In intensively developed reservoirs, these structures can become the most abundant littoral features (Barwick 2004). Often, the presence of these structures is associated with the removal of coarse woody debris and aquatic macrophytes, leading to reduced habitat complexity; however, some of these structures have been modified to increase, or at least mitigate habitat loss (e.g., via addition of riprap). Barwick (2004) found greater species richness and centrarchid abundance in coarse woody debris and developed sites that contained complex structures (i.e., riprap, piers) compared to undeveloped sites that lacked habitat complexity. Trial et al. (2001) found riprap and undeveloped, naturally vegetated areas to have similarly high species richness and diversity, while bulkheads and undeveloped, non-vegetated areas had significantly lower species richness and diversity. Similarly, Jennings et al. (1999) found riprap sites to have greater species richness than bulkheads or shorelines with no erosion control structure, presumably due to the increased habitat complexity provided by rip rap. One method used to increase habitat complexity in developed areas is the use of artificial brush structures. Barwick et al. (2004) found higher fish abundances on developed shorelines that contained such structures compared to developed shorelines without the structures.

In addition to the fish community response to shoreline development, other response variables have been studied, including sediment composition, and macroinvertebrate communities. Francis et al. (2007) found that littoral sediments of undeveloped lakes in the U.S. Pacific northwest contained 34-77% organic matter, whereas developed lakes contained 1-3% organic matter. Brauns et al. (2007) demonstrated that littoral macroinvertebrate species richness in seven lakes was significantly lower on recreational beaches and bulkheads compared to natural shorelines, and found no significant difference between riprap and natural shorelines, concluding that this was due to the habitat complexity offered by riprap habitats. Schmude et al. (1998) found no significant differences for macroinvertebrate abundance or diversity among riprap, bulkhead, and undeveloped shorelines, but did see a trend of higher macroinvertebrate abundance and diversity at riprap sites, suggesting that shoreline development that alters habitat complexity may also alter macroinvertebrate diversity. Alternatively, some studies have shown little effect of shoreline development or the removal of coarse woody debris on macroinvertebrates (Smokorowski et al. 2006; Roth et al. 2007; Helmus and Sass 2008).

Given the increasing human development around our water bodies, there has been an increase in the number of studies designed to determine how our aquatic systems are affected. Although the number of studies has increased, the results remain conflicting, sometimes showing positive, negative, or even no responses to shoreline development. Given this, I conducted a replicated field study in Lake Martin, Alabama to examine the role of shoreline development on the aquatic community.

My goal for this study was to quantify the influence of shoreline development on the aquatic community. More specifically, I was interested in whether shoreline development influenced fishes, and whether that influence was manifested through effects on lower trophic

levels. In particular, I quantified water quality characteristics (i.e., water temperature, dissolved oxygen concentrations, chlorophyll-a concentrations, and turbidity), and abundance and species composition of zooplankton, aquatic macroinvertebrates, larval fishes, and fishes at four different shoreline habitat types (undeveloped shoreline, shoreline with bulkhead, shoreline with riprap, and shoreline with both bulkhead and riprap at the bottom of the bulkhead), as well as sportfish age, growth, and diet composition. Given that water should be well mixed throughout the study area, I did not expect to find differences in water quality parameters. However, I did expect to find higher densities of zooplankton, macroinvertebrate, and larval fish in more complex habitats (i.e., riprap). Similarly, I expected to find higher abundance of juvenile and adult fish in more complex habitats. I also expected that differences in macroinvertebrates across shoreline habitats would lead to differences in sportfish diets.

Methods

Study area and sampling locations

Lake Martin is a tributary storage impoundment on the Tallapoosa River in east-central Alabama (Figure 1). The reservoir was impounded in 1927 to provide hydroelectric generation and water supply (Alabama Power 2009). Lake Martin has a full pool elevation of 149 m above sea level with 16,188 ha of surface area at full pool and a 7,770 sq-km watershed (Alabama Power 2009). It is highly dendritic with a shoreline development index of 23, such that there is a relatively large amount of littoral zone.

This study was conducted in the Blue Creek arm of Lake Martin (Figure 1a). Four replicate sites within each of four shoreline development or habitat types (undeveloped shoreline, shoreline with bulkhead, shoreline with riprap, and shoreline with both bulkhead and riprap at the base of the bulkhead) were identified using visual inspection. Sampling began when the lake was at or near full pool during spring and was conducted once per week. During 2009, sites were sampled from 3 April through 7 July, and in 2010, sampling began on 13 April and ended 14 July.

Sample collections

Water samples were collected from the surface using 500 ml dark polyethylene bottles, placed directly on ice, and returned to the lab. Chlorophyll-a concentration was determined by filtering 500 mL of water onto glass fibre filters, extracting chlorophyll in 95% ethanol, and then measuring fluorescence using a fluorometer (Turner Designs Aquafluor). Turbidity was measured with a nephelometer (NTU; HF Scientific, Inc. Microw TPW). Dissolved oxygen (mg/L) and temperature (°C) were measured at the water surface at each site (Yellow Springs Instruments Model 550 A meter).

One zooplankton sample was collected every two weeks at each site using a 30 cm diameter, 90 cm long hand-towed plankton net (50 µm mesh). Samples were taken from a depth of 1 m to the surface, concentrated, and preserved in 95% ethanol. Samples were counted until at least 200 individuals of the most abundant taxa were counted or until the entire sample was counted (Dettmers and Stein 1992; Welker et al. 1994). Cladocerans were identified to genus and copepods were identified to family as calanoids, cyclopoids, or nauplii.

To quantify abundance and species composition of aquatic macroinvertebrates, 9-plate Hester-Dendy samplers (total sampling area is 0.10 m^2) were used (Hester and Dendy 1962; Rabeni 1996). Four Hester-Dendy samplers were attached to a brick and placed at each site in 1-2 m of water. One sampler was retrieved every two weeks. Three samplers at each site were used for analyses; the fourth served as a spare in the event that one was lost. Samplers were placed on ice and returned to the laboratory, where all macroinvertebrates were removed, identified to order, and counted.

Larval fish were sampled using two different methods: a push net (50 cm diameter, 150 cm long, 500 µm mesh) with an attached flow meter (General Oceanics, Inc., to allow calculation of sample speed and volume of water filtered), and quatrefoil light traps (similar to Floyd et al. [1984]) constructed of molded plexiglass and PVC with an attached LED light stick (Aquatec USA). One larval fish push was collected at each site during each week of sampling. Pushes were conducted for 4-5 minutes along the shoreline of the entire site at approximately 1 m/second. Light traps were set approximately once every two weeks at each site along the shoreline in 1-2 m of water. Traps were deployed at sunset and retrieved at sunrise for a total set time of approximately 12 hours per site. Light traps were not used during 2010 due to the low catch rates in 2009. Larval fish were preserved in 95% ethanol, identified to genus, and counted

to estimate number of larval fish/m^3 (for push net) and number of larval fish/trap night (for light traps).

To sample the fish community, I used two types of boat mounted, pulsed-DC electrofishing (Smith-Root Inc. DC Electrofisher 7.5 GPP), including boom electrofishing and prod pole electrofishing (to more effectively sample around shallow shoreline areas). Start and end points were identified at each site to define transects. Both types of gear were used at each transect. All fish $\geq$ 100 mm total length (TL), other than Alabama bass *Micropterus henshalli*, were measured (TL, nearest mm), weighed (wet weight, nearest g), and released. All fish $<$ 100 mm were placed on ice and returned to the laboratory, where they were identified to species, measured (TL, nearest mm) and weighed (wet weight, nearest g). All Alabama bass $\geq$ 100 mm TL were placed on ice and returned to the laboratory for age-and-growth assessment, as well as to quantify diet composition.

Fish abundance and species composition

Fish abundance was estimated as relative abundance via catch-per-unit-effort (CPUE). For larval fish pushes, CPUE is expressed as number of larval fish per m^3 of water filtered. Light trap CPUE is expressed as larval fish per trap night. For electrofishing, the two gears were combined and CPUE expressed as number of fish per hour.

To evaluate fish community composition, we quantified species richness and species diversity for each habitat type. Shannon's diversity index was used to characterize species diversity (Shannon and Weaver 1949), calculated as follows:

$$H' = - \sum_{i=1}^{s} (p_i)(\log_e p_i)$$

where s = number of species, and p_i = proportion of the total sample represented by the ith species.

Sport fish age and growth

Alabama bass collected for age-and-growth analysis were measured (TL, nearest mm) and weighed (wet weight, nearest g). Sagittal otoliths were removed and stored dry plastic vials. Two independent readers determined age by use of a dissecting microscope to count otolith annuli on whole saggital otoliths. Using an eyepiece micrometer, radii to each annulus and the total distance posterior-most edge were measured (nearest 0.001 mm). For all fish age-5 and older, otoliths were set in two-part epoxy and sectioned using a low speed diamond wheel saw (South Bay Technology Model 650). Sections were mounted on a slide using thermoplastic cement and polished for clarity. An image analysis system was used to measure the distance from the focus to the outer edge of each annulus and to the edge for the sectioned otoliths. Back-calculated length at the last annulus was calculated for individual fish via the direct proportion method (DeVries and Frie 1996).

Sport fish diet composition

Stomachs from Alabama bass $\geq$ 100 mm TL were removed in the laboratory and diet items were identified to the lowest practical taxonomic level. Biomass of prey consumed was determined from length-weight regressions found in this study and published literature values (i.e., Benke et al. 1999, Norris 2007). Diet proportions were determined by averaging estimates calculated for individual fish.

Statistical Analyses

Mixed-model repeated-measures analysis of variance (PROC MIXED, SAS Institute 2008) was used to analyze temperature, dissolved oxygen concentration, chlorophyll-a concentration, turbidity, zooplankton density, larval fish density, fish abundance, species richness, and species diversity. Macroinvertebrate densities were analyzed within each of the

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three sampling periods with analysis of variance (PROC GLM, SAS Institute 2008). Alabama bass used for age-and-growth and diet analysis for a particular habitat type were pooled across replicate sites within shoreline habitat types to determine the mean length-at-age and mean diet biomass for that habitat type. The mean lengths at a particular age and mean individual diet biomass were then analyzed with analysis of variance (PROC GLM, SAS Institute 2008). For the mixed-model repeated-measures analysis of variance, error structures were compared using SAS, and the error structure with the lowest AIC was selected. Statistical significance for all tests was set at $P \leq 0.05$. The time $\times$ habitat type interaction was not significant for any of the response variables, so only the main effects of habitat type and time were considered. When significance was found, least-square means multiple comparison tests were used. A power analysis was used to determine the number of sites needed in each shoreline habitat type to find significance for macroinvertebrates (PROC GLMPOWER, SAS Institute 2008). For this procedure, power was set at 0.8.

Results

Temperature and Dissolved Oxygen

During 2009, average temperature ranged from 16.3°C to 32.1°C, and did not differ among shoreline habitat types (ANOVA: $F_{3,16} = 0.70$; $P = 0.57$). Average temperature differed through time (ANOVA: $F_{9,125} = 1,655.74$; $P < 0.0001$), being lowest in early April and peaking in July (Figure 2a). Similarly, dissolved oxygen concentration did not differ among shoreline habitat types during 2009 (ANOVA: $F_{3,16} = 0.36$; $P = 0.78$), but differed through time (ANOVA: $F_{9,125} = 146.21$; $P < 0.0001$), with dissolved oxygen concentration being highest during April, and declining thereafter (Figure 3a). Similarly, average temperature during 2010 did not differ among shoreline habitat types (ANOVA: $F_{3,12} = 1.36$; $P = 0.30$), with values ranging from 21.9°C to 33.5°C. Again, average temperature differed through time (ANOVA: $F_{12,162} = 484.90$; $P < 0.0001$), being lowest in April, and peaking in late June (Figure 2b). Dissolved oxygen concentration did not differ among shoreline habitat types during 2010 (ANOVA: $F_{3,12} = 2.65$; $P = 0.10$). As was the case in 2009, dissolved oxygen concentration differed through time (ANOVA: $F_{12,162} = 137.71$; $P < 0.0001$), with dissolved oxygen concentration being highest during April, and declining through our sampling (Figure 3b).

Chlorophyll-a

During 2009, average chlorophyll-a concentration ranged from 1.48 to 3.77 µg/l, and did not differ among shoreline habitat types (ANOVA: $F_{3,12} = 2.29$; $P = 0.12$). There was a significant time effect (ANOVA: $F_{9,106} = 10.98$; $P < 0.0001$), with values being lowest during April through mid-May, peaking in late May, and then declining again in late June (Figure 4a). Similarly, average chlorophyll-a concentration during 2010 ranged from 2.10 to 3.84 µg/l, and did not differ among shoreline habitat types (ANOVA: $F_{3,12} = 0.26$; $P = 0.85$). There was a

significant time effect (ANOVA: $F_{6,90} = 37.12$; $P < 0.0001$), with values being lowest in April through early May, peaking in mid to late-May, declining in June, and then increasing again in July (Figure 4b).

<center>*Turbidity*</center>

During 2009, average turbidity level ranged from 1.27 to 4.43 NTUs, and did not differ among shoreline habitat types (ANOVA: $F_{3,16} = 0.74$; $P = 0.54$). As with chlorophyll-a, there was a significant time effect (ANOVA: $F_{9,109} = 10.48$; $P < 0.0001$), with the highest turbidity level in April, followed by a decline in May and June, with the exception of a peak in late May after a storm event (Figure 5a). Average turbidity level during 2010 ranged from 1.54 to 8.48 NTUs, and did not differ among habitat types (ANOVA: $F_{3,12} = 3.04$; $P = 0.07$). Again, similar to both the 2010 chlorophyll-a data and the 2009 turbidity data, there was a significant time effect (ANOVA: $F_{6,90} = 151.53$; $P < 0.0001$), with the highest turbidity levels in April, followed by a steady decline throughout the remainder of the study (Figure 5b).

<center>*Zooplankton*</center>

During 2009, zooplankton density did not differ among shoreline habitat types (ANOVA: $F_{3,16} = 0.45$; $P = 0.72$), although it did differ across sampling dates (ANOVA: $F_{7,51} = 9.01$; $P < 0.0001$). The highest zooplankton density occurred in all habitat types in April and then declined throughout the remainder of our 2009 sampling period (Figure 6a). Copepods (calanoids and cyclopoids) were the most abundant taxonomic groups throughout our sampling. In addition, *Bosmina* and *Daphnia* were present in relatively high abundances early in our sampling, but declined through time. Other forms that were present at lower densities in 2009 included *Holopedium*, *Ilyocryptus*, and *Diaphanosoma*. Zooplankton density in 2010 did not differ among shoreline habitat types (ANOVA: $F_{3,12} = 0.39$; $P = 0.77$), although it did differ through

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time (ANOVA: $F_{6,90} = 6.09$; $P < 0.0001$). The highest zooplankton density occurred April through May, declined in June, and then increased somewhat again in July (Figure 6b). *Bosmina* was the most abundant taxonomic group throughout our 2010 sampling. In addition, copepods (calanoids and cyclopoids) and *Daphnia* were present in relatively high abundance early in our sampling, but declined through time. Other forms that were present at lower densities included *Holopedium*, *Ilyocryptus*, and *Diaphanosoma*.

<center>*Macroinvertebrates*</center>

The density of macroinvertebrates on the Hester-Dendy samplers did not differ among shoreline habitat types during any of the three sampling periods in 2009 (Figure 7a; ANOVA; first period: $F_3 = 0.98$; $P = 0.44$; second period $F_3 = 0.43$; $P = 0.73$; third period: $F_3 = 0.35$; $P = 0.79$). Three taxonomic groups (chironomids, ostracods, physid snails) were most abundant in all but the undeveloped shoreline habitat type, where lymnaed snails, physid snails, and caddisfly larvae were most abundant (Table 1). Similarly, macroinvertebrate density on the Hester-Dendy samplers also did not differ among shoreline habitat types during any of the three sampling periods in 2010 (Figure 7b; ANOVA; first period: $F_3 = 0.46$; $P = 0.72$; second period: $F_3 = 2.33$; $P = 0.13$; third period: $F_3 = 2.77$; $P = 0.09$). Although not significant, macroinvertebrate density approached significance for the second and third periods. A power analysis determined 32 sites (8 per shoreline habitat type) would have been needed to find significance for macroinvertebrate density. Three taxonomic groups (chironomids, ostracods, and mayfly larvae) were most abundant in all but the riprap habitat, where chironomids, water mites, and physid snails were most abundant (Table 1).

Larval Fish

For larval fish pushes in 2009, larval fish density did not differ among shoreline habitat types (ANOVA: $F_{3,33} = 0.65$; $P = 0.59$), although it did differ across time ($F_{12,33} = 5.47$; $P < 0.0001$). Larval fish density peaked in April and again in late May in all habitat types (Figure 8a). Most larvae in April were *Pomoxis* larvae, while *Lepomis* was the most abundant group in May. In addition, larval shad *Dorosoma* spp., and larval Cyprinids were collected, but at a lower density. In 2010 larval fish pushes, larval fish density again did not differ among shoreline habitat types (ANOVA: $F_{3,12} = 0.86$; $P = 0.49$), but did differ across time ($F_{12,180} = 6.63$; $P < 0.0001$). As in 2009, larval fish density was highest in April and again in mid to late May in all habitat types (Figure 8b). Again, most larvae in April were *Pomoxis*, while *Lepomis* was the most abundant taxonomic group in May. In addition, larval shad *Dorosoma*, Cyprinids, and yellow perch were collected, but at lower densities.

For 2009 light trap collections, there were no differences for mean number of larvae collected among shoreline habitat types (ANOVA: $F_{3,15} = 0.72$; $P = 0.56$), and mean number of larvae did not differ through time (ANOVA: $F_{4,50} = 1.85$; $P = 0.13$) (Figure 9). Taxa of larval fish collected in light traps versus the larval fish push net were similar, with far fewer individuals collected in the light traps.

Fish Community

Juvenile and adult fish relative abundance differed among habitat types in 2009 (ANOVA: $F_{2,16} = 4.52$; $P = 0.02$), being highest at riprap, bulkhead with riprap, and undeveloped sites (Figure 10a). Bulkhead sites had lower fish relative abundance than at riprap and bulkhead with riprap ($t_{16} = -3.25$; $P = 0.0050$; $t_{16} = -2.99$; $P = 0.0087$, respectively), but did not differ from that at undeveloped sites ($t_{16} = 1.46$; $P = 0.16$). Fish relative abundances differed through time

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(ANOVA: $F_{11,134} = 2.61$; $P = 0.0049$), being highest in May through early June, and again in early July (Figure 8a). Similarly, fish relative abundance in 2010 differed among habitat types (ANOVA: $F_{3,12} = 19.57$; $P < 0.0001$), with relative abundance highest at riprap sites and bulkhead with riprap sites (Figure 10b). Bulkhead sites had significantly lower relative abundance than bulkhead with riprap, riprap, and undeveloped sites ($t_{12} = -5.62$; $P = 0.0001$; $t_{12} = -7.30$; $P < 0.0001$; $t_{12} = -4.38$; $P = 0.0009$, respectively). Fish relative abundance differed through time (ANOVA: $F_{12,172} = 6.63$; $P < 0.0001$), being highest in April through early May, and again in early June (Figure 8b).

Species richness did not differ significantly among habitat types in 2009 (Figure 11a; ANOVA: $F_{3,15} = 2.42$; $P = 0.11$), nor across time (ANOVA: $F_{11,100} = 1.64$; $P = 0.10$). Twenty-three species were collected during 2009 (Table 2). Bluegill was the most common species at all four shoreline habitat types, followed by Alabama bass and two *Lepomis* species at the three developed shoreline habitat types (bulkhead, bulkhead with riprap, and riprap). At sites with undeveloped shoreline, two shiner species followed bluegill in abundance (Table 3). Species richness in 2010 differed among habitat types (Figure 11b; ANOVA: $F_{3,12} = 22.25$; $P < 0.0001$), with undeveloped sites having higher species richness than bulkhead, bulkhead with riprap, and riprap sites ($t_{12} = 6.45$; $P < 0.0001$; $t_{12} = 2.54$; $P = 0.03$; $t_{12} = 7.02$; $P < 0.0001$, respectively). Bulkhead and riprap sites had similar species richness ($t_{12} = 0.61$; $P = 0.55$). Twenty-two species were collected during 2010 (Table 4). As was the case in 2009, bluegill was the most common species at all four shoreline habitat types, followed by Alabama bass and two *Lepomis* species at three shoreline habitat types (bulkhead, bulkhead and riprap, and riprap). At undeveloped shoreline sites, two shiner species followed bluegill in abundance, similar to 2009 (Table 3).

Species diversity differed among habitat types in 2009 (ANOVA: $F_{3,15} = 4.77$; $P = 0.016$). Undeveloped sites had higher species diversity than all other habitat types; the other shoreline development types (bulkhead, bulkhead with riprap, and riprap) did not differ from one another (Figure 12a). This trend was consistent across time (ANOVA: $F_{11,133} = 1.07$; $P = 0.39$). Species diversity differed among habitat types in 2010 (ANOVA: $F_{3,12} = 17.79$; $P < 0.0001$), with no differences across time (ANOVA: $F_{12,172} = 1.70$; $P = 0.07$). As in 2009, undeveloped sites had higher species diversity than bulkhead, bulkhead with riprap, and riprap sites (Figure 12b; $t_{12} = 3.23$; $P = 0.0072$; $t_{12} = 2.62$; $P = 0.02$; $t_{12} = 7.21$; $P < 0.0001$, respectively).

Age and Growth

Mean length at age for Alabama bass did not differ among habitat types for ages 1-3 in 2009 (Figure 13a; ANOVA: $F_{3,14} = 0.05$; $P = 0.98$; $F_{3,10} = 0.63$; $P = 0.61$; $F_{3,5} = 0.81$; $P = 0.54$, respectively). Few Alabama bass age-4 and older were collected (Figure 14a). Similarly, during 2010 mean length at age for Alabama bass did not differ significantly among habitat types for ages 1-3 (Figure 13b; ANOVA: $F_{3,12} = 0.56$; $P = 0.56$; $F_{3,12} = 0.38$; $P = 0.77$; $F_{3,4} = 0.31$; $P = 0.82$, respectively). As in 2009, few Alabama bass age-4 and older were collected (Figure 14b).

Diets

Alabama bass diets were diverse (Figure 15). In 2009, *Micropterus* spp. contributed the highest proportion of diets for age-1 fish at bulkhead (46.7%) and bulkhead with riprap (40.0%) sites, and was the second highest at riprap sites (27.9%). Cyprinids were the most prevalent prey item for age-1 fish at undeveloped sites (30.6%). At age-2, *Lepomis* spp. made up the largest proportion of the diets at bulkhead (40.8%) and riprap sites (62.5%). Sample sizes for age-2 fish at bulkhead with riprap, and undeveloped sites were low, as was the case with age-3 and age-4 fish at all shoreline habitat types. There were no differences in mean biomass in diets of

Alabama bass among shoreline habitat types during 2009 (Figure 17a; ANOVA: $F_3 = 1.66$; P = 0.22). Alabama bass diets were again diverse in 2010 (Figure 16). Crappies, which were not found in the 2009 diets, made up the highest proportion of age-1 fish diets at bulkhead sites (27.8%). *Micropterus* spp. again made up the highest proportion of age-1 fish at bulkhead with riprap sites, followed by yellow perch (24.7%). Age-1 Alabama bass at riprap sites consumed mostly zooplankton (29.9%) and crappie (26.1%). Yellow perch made up the highest proportion of age-1 Alabama bass diets at undeveloped sites (26.1%). Diets of age-2 fish at bulkhead sites were fairly evenly distributed between yellow perch (36.7%), *Lepomis* spp. (30.4%), and macroinvertebrates (26.4%). Macroinvertebrates accounted for 42.7% of the diets of age-2 fish at bulkhead and riprap sites. Age-2 Alabama bass diets at riprap and undeveloped sites consisted primarily of *Lepomis* spp. (47.1% and 56.4%, respectively). Average biomass of diets did not differ among shoreline habitat types during 2010 (Figure 17b; ANOVA: $F_3 = 2.95$; P = 0.08).

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Discussion

Human development along shorelines can have strong effects on the aquatic biological communities in water bodies (Schindler et al. 2000; Trial et al. 2001; Scheuerell and Schindler 2004). Shoreline armoring is often associated with the removal of coarse woody debris and other complex habitats that could be important to the aquatic community. This loss of complex habitat in littoral areas could impact the aquatic community if shoreline development is not closely monitored. My goals in this study were to determine if there were any differences in a variety of abiotic and biotic factors associated with four different shoreline habitat types. Based on two years of sampling, my data suggest that there are no differences in abiotic variables or in lower trophic levels, but that fishes are affected.

I measured several abiotic factors at each site, and found no differences in turbidity, water temperature, or dissolved oxygen concentration among shoreline habitat types. This lack of difference in abiotic factors among shoreline habitat types was not surprising, based both on recent literature reviews (e.g., Smokorowski et al. 2006) and that I expected water to be well mixed spatially throughout the study area. All of these factors varied across time with these temporal trends consistent among the shoreline habitat types. These temporal trends were, therefore, likely due to regional influences (i.e., rainfall, runoff input, and other climatic factors) rather than due to local shoreline development.

Similarly, no statistically significant differences were found among shoreline habitat types in lower trophic levels (i.e., chlorophyll-a concentration, zooplankton density, and macroinvertebrate density). The lack of difference among shoreline habitat types in chlorophyll-a concentration was not surprising, for the same reasons as described above for the abiotic factors. However, I did expect that I might see differences among shoreline habitat types in both

zooplankton and macroinvertebrate density. I expected that differences in habitat complexity, with higher complexity in both riprap sites and in the undeveloped sites relative to bulkhead sites, could yield higher zooplankton density (Goforth and Carrnan 2005) and macroinvertebrate density (Schmude et al. 1998; Brauns et al. 2007). However, this was clearly not the case in my two years of data, suggesting that this habitat complexity is either not as critical to these lower trophic levels as I might have expected or that more complex foodweb effects might be important. The increased abundance and therefore consumptive demand of juvenile fishes in the riprap sites may have prevented macroinvertebrates, zooplankton, and larval fish from reaching high densities in this complex habitat, much as Gilinsky (1984) found the presence of bluegills significantly reduced macroinvertebrate density in complex habitats. These top-down effects (sensu Carpenter et al. 1985) may have been important in maintaining similar prey abundances across shoreline habitat types. Similar to my results, Carey et al. (2009) found no changes in macroinvertebrates with increased habitat complexity. As with the abiotic factors, I again found significant time effects for chlorophyll-a concentration and zooplankton density. These patterns were consistent among all shoreline habitat types, indicating that the temporal variation was due to broader system-wide effects and not local shoreline habitat differences.

I also found that larval fish density (determined by either the active push net sampling or by the passive light traps) did not differ among shoreline habitat types. This result was somewhat surprising, as I expected that habitat complexity differences might translate into fish abundance differences at all life stages. For example, early life stages of several fish species exhibit size-related migrations between the littoral and limnetic zone (Faber 1967; Werner 1967, 1969; Storck et al. 1978; Whiteside et al. 1985; Werner and Hall 1988). Those species that return to the littoral zone are thought to do so to minimize their risk of predation to larger

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piscivores, given that the habitat structure in the littoral zone or the shallow water at the shoreline edge provides protection from predators (Savino and Stein 1982; Werner and Hall 1983; DeVries 1990). Because shoreline that is modified with a bulkhead provides little structure (in the form of interstitial space) or shallow habitat for these age-0 fishes to avoid predation, I expected that larval fish abundance would be reduced in bulkhead habitats versus undeveloped shorelines or those with riprap. However, this was not the case, likely again due to the mixing of plankton in the open water across multiple shoreline habitat types. I did find temporal differences in larval fish push samples, but these again were consistent among shoreline habitat types, being driven by spawning of crappie in April and *Lepomis* in May of both years.

Shoreline habitat type did affect the relative abundance and community composition of juvenile and adult fishes. Both types of riprap shoreline types had higher overall relative abundance than did bulkhead sites for both years. The increased habitat complexity provided by riprap creates interstitial spaces that many fishes use as cover (Tabor and Wurtsbaugh 1991; Beauchamp et al. 1994). Similar to my results, Beauchamp et al. (1994) found higher densities of fish in rocky habitats that increased interstitial spaces, and Barwick (2004) found higher abundances of centrarchids in complex habitats relative to habitats that lacked complexity. Although I found no significant differences in species richness among shoreline habitat types in 2009, I did find higher species richness at undeveloped sites in 2010 when compared to the other three shoreline habitat types. Other studies have found higher species richness at riprap sites when compared to bulkhead structures (Jennings et al. 1999; Trial et al. 2001). In addition, species diversity was significantly higher at undeveloped sites for both years, while all other shoreline habitat types were similar. Trial et al. (2001) found species diversity at riprap and undeveloped, vegetated areas to be higher than bulkhead and undeveloped areas that lacked

habitat complexity in the form of vegetation. The high abundance of bluegill at bulkhead, bulkhead with riprap, and riprap sites served to reduce species diversity, while undeveloped sites had a more even distribution of species. Relative to predation effects, these results may point to an indirect, behavioral effect of piscivores, in that they restrict them to areas of structure where predation risk is minimized (Savino and Stein 1982), which in this case are the interstitial spaces of riprap. If small prey fishes are using these areas as refuge from predation, this could explain why I did not see differences at lower trophic levels.

The effects of predators and external competitors on the community using a patch of refuge can be strongly influenced by the size and edge length of that refuge. If the effects of the predators and external competitors are mostly limited to edges, then more continuous patches with less edge will be more effective refuges than the same area broken into smaller patches with greater edge length (Murcia 1995). In the case of Lake Martin shoreline habitats, if undeveloped shoreline is divided by developed shorelines into smaller and smaller patches, its effectiveness in maintaining greater species diversity could be compromised

There were no differences in mean length-at-age for Alabama bass in either year. This is not surprising, considering Alabama bass only spend short periods of time in the littoral zone in reservoirs similar to Lake Martin that experience winter drawdowns, and typically inhabit deeper water (Hunter and Maceina 2008). When in the littoral zone of the reservoir, they likely move among shoreline habitats such that no habitat-specific differences could develop. Although DiCenzo et al. (1995) found faster growth rates for Alabama bass in areas with higher chlorophyll-a concentration relative to low chlorophyll-a concentration, my sites showed no differences in chlorophyll-a concentration, further supporting this lack of habitat effect on growth.

Alabama bass consumed a wide variety of prey types including a relatively high proportion of yellow perch. This was unexpected given the relatively low occurrence of yellow perch in my collections. This further suggests that Alabama bass are moving among shorelines and possibly to offshore habitats (Hunter and Maceina 2008). Alabama bass also consumed a wide variety of prey types that were present in high abundances at the four shoreline habitat types in our study. Although there were some differences in diets among fishes from different shoreline habitat types (e.g., consistently higher levels of minnows in the diets of Alabama bass at undeveloped sites) total biomass of prey consumed per bass did not differ among shoreline habitat types. I expected Alabama bass at both riprap habitats (greater habitat complexity) to have lower biomass of prey in diets compared to bulkhead sites (lower habitat complexity) because increased habitat complexity has been shown reduce predator success (Savino and Stein 1982). However this was not the case, even further evidence that Alabama bass are moving among shoreline habitat types, and feeding in more profitable habitats.

Results from this study should aid managers in implementing shoreline development regulations on southeastern reservoirs. While I found no differences across shoreline habitat types in abiotic conditions or lower trophic levels (e.g., chlorophyll-a, zooplankton), I did find fishes to be affected by shoreline development. Fish abundances were lowest at bulkhead sites relative to any of the riprap sites. When considering species richness and species diversity, both tended to be highest at undeveloped sites versus any of the developed sites. It is clear that shoreline development of any sort affects the resulting fish community (in terms of species composition), but abundance can be enhanced by providing some degree of structure with interstitial spaces (e.g., riprap) as a component of any shoreline development efforts. Clearly human development of lake and reservoir shorelines is going to continue and this development

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will affect resident fishes. With my results, better informed decisions can be made by resource

managers about further development. My recommendations would be to maintain some

undeveloped shoreline whenever possible, but also, when shoreline development does occur,

substantive amounts of riprap should be included to provide interstitial spaces and shallow water

that many fishes require. Further, riprap shorelines should be extended in areas of existing

development so habitat fragmentation is minimized. When habitats are fragmented, which is the

case with typical shoreline development across individual property boundaries, predation can be

high at the edges of habitats (Murcia 1995). Long stretches of riprap would reduce such edge

effects, and allow some degree of predation refuge for juvenile and small bodied fish.

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Table 1. A list of the three most abundant macroinvertebrate groups by habitat type during 2009 and 2010.

Bulkhead	Bulkhead & Riprap	Riprap	Undeveloped
2009			
Chironomid larvae	Chironomid larvae	Chironomid larvae	Lymnea snail
Ostracod	Ostracod	Ostracod	Physa snail
Physa snail	Physa snail	Physa snail	Caddisfly larvae
2010			
Chironomid larvae	Chironomid larvae	Chironomid larvae	Chironomid larvae
Ostracod	Ostracod	Ostracod	Ostracod
Mayfly larvae	Mayfly larvae	Physa snail	Mayfly larvae

Table 2. Species collected during 2009. An "X" indicates the species was present at that particular shoreline habitat type. Habitat types are coded as follows: BH=Bulkhead, BR=Bulkhead & Riprap, RR=Riprap, UN=Undeveloped.

Common name	Scientific name	BH	BR	RR	UN
Blacktail shiner	*Cyprinella venusta*	X	X	X	X
Common carp	*Cyprinus carpio*	-	-	-	X
Gizzard shad	*Dorosoma cepedianum*	X	X	X	X
Threadfin shad	*Dorosoma petense*	X	X	X	X
Tallapoosa darter	*Etheostoma tallapoosae*	-	X	-	-
Channel catfish	*Ictalurus punctatus*	X	X	X	X
Redbreast sunfish	*Lepomis auritus*	X	X	X	X
Green sunfish	*L. cyanellus*	X	X	X	X
Warmouth	*L. gulosus*	-	X	X	X
Bluegill	*L. macrochirus*	X	X	X	X
Longear sunfish	*L. megalotis*	-	-	X	-
Redear sunfish	*L. microlophus*	X	X	X	X
Alabama bass	*Micropterus henshalli*	X	X	X	X
Largemouth bass	*M. salmoides*	X	X	X	X
Spotted sucker	*Minytrema melanops*	X	X	X	X
White bass	*Morone chrysops*	X	-	-	X
Blacktail redhorse	*Moxostoma poecilurum*	X	X	X	X
Weed shiner	*Notropis texanus*	X	X	X	X
Yellow perch	*Perca flavescens*	X	X	X	X
Mobile logperch	*Percina kathae*	-	-	X	X
Bullhead minnow	*Pimephales vigilax*	X	X	X	X
Black crappie	*Pomoxis nigromaculatus*	X	-	X	X
Flathead catfish	*Pylodictis olivaris*	X	X	X	X
	Total species:	18	18	20	21

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Table 3. A list of the three most abundant fish species for each shoreline habitat type during 2009 and 2010.

Bulkhead	Bulkhead & Riprap	Riprap	Undeveloped
Bluegill	Bluegill	Bluegill	Bluegill
Lepomis macrochirus	*Lepomis macrochirus*	*Lepomis macrochirus*	*Lepomis macrochirus*
Alabama bass	Green sunfish	Green sunfish	Blacktail shiner
Micropterus henshalli	*Lepomis cyanellus*	*Lepomis cyanellus*	*Cyprinella venusta*
Green sunfish	Alabama bass	Redbreast sunfish	Weed shiner
Lepomis cyanellus	*Micropterus henshalli*	*Lepomis auritus*	*Notropis texanus*

Table 4. Species collected during 2010. An "X" indicates the species was present at that particular shoreline habitat type. Habitat types are coded as follows: BH=Bulkhead, BR=Bulkhead & Riprap, RR=Riprap, UN=Undeveloped.

Common name	Scientific name	BH	BR	RR	UN
Blacktail shiner	*Cyprinella venusta*	X	X	X	X
Gizzard shad	*Dorosoma cepedianum*	X	X	X	X
Threadfin shad	*Dorosoma petense*	X	X	X	X
Tallapoosa darter	*Etheostoma tallapoosae*	-	X	-	-
Channel catfish	*Ictalurus punctatus*	X	X	X	X
Redbreast sunfish	*Lepomis auritus*	X	X	X	X
Green sunfish	*L. cyanellus*	X	X	X	X
Warmouth	*L. gulosus*	X	X	X	X
Bluegill	*L. macrochirus*	X	X	X	X
Redear sunfish	*L. microlophus*	X	X	X	X
Pretty shiner	*Lythrurus bellus*	X	X	-	X
Alabama bass	*Micropterus henshalli*	X	X	X	X
Largemouth bass	*M. salmoides*	X	X	X	X
Spotted sucker	*Minytrema melanops*	X	X	X	X
White bass	*Morone chrysops*	X	-	X	X
Blacktail redhorse	*Moxostoma poecilurum*	X	X	X	X
Golden shiner	*Notemigonus crysoleucas*	-	X	X	-
Weed shiner	*Notropis texanus*	X	X	X	X
Yellow perch	*Perca flavescens*	X	-	X	X
Bullhead minnow	*Pimephales vigilax*	-	X	-	X
Black crappie	*Pomoxis nigromaculatus*	X	-	X	X
Flathead catfish	*Pylodictis olivaris*	X	X	X	X
	Total Species:	19	19	19	20
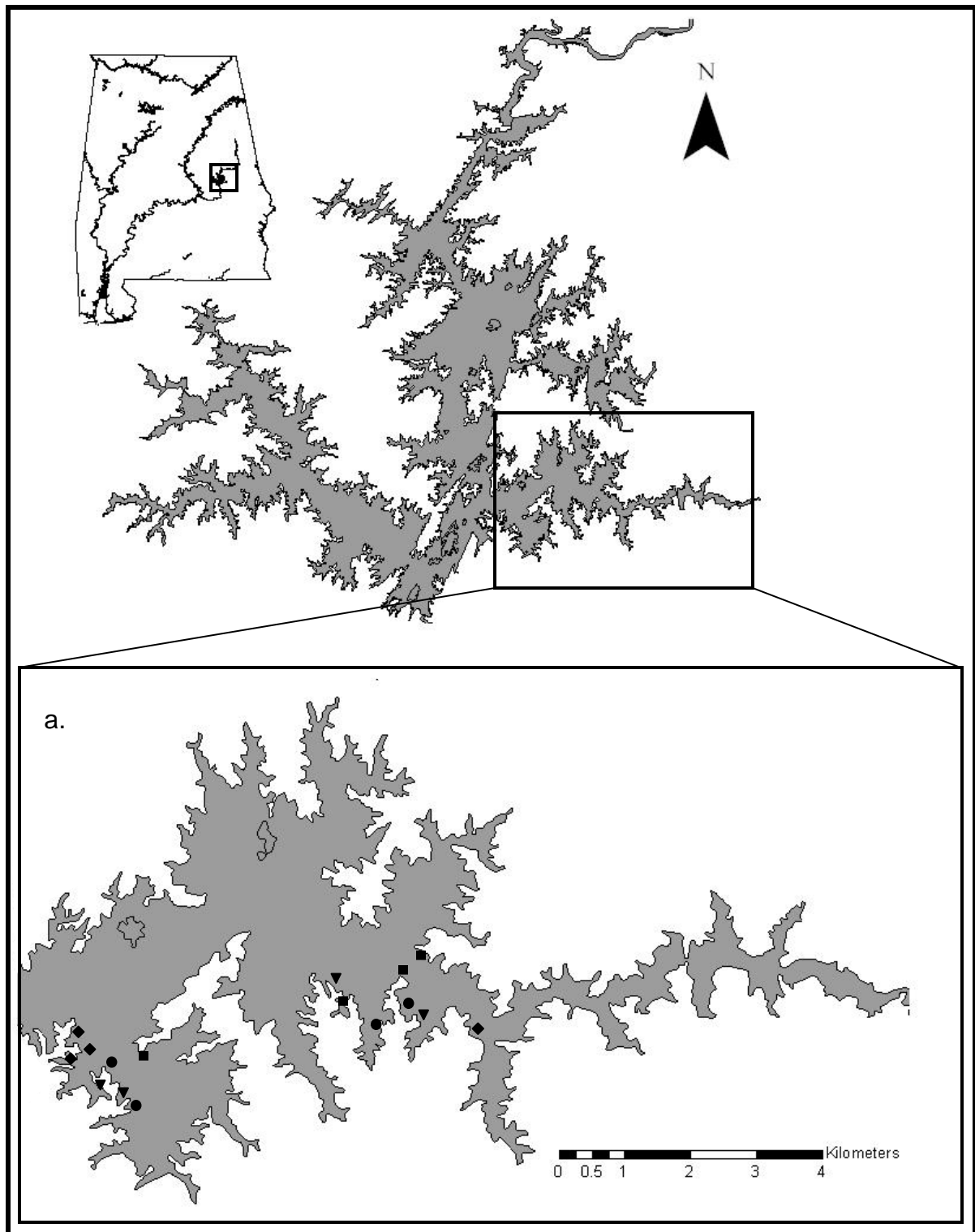
Figure 1. Map of Lake Martin, Alabama with inset of study area, Blue Creek (a). Shaded area

represents water. Symbols represent sample sites and are the same as in subsequent figures.
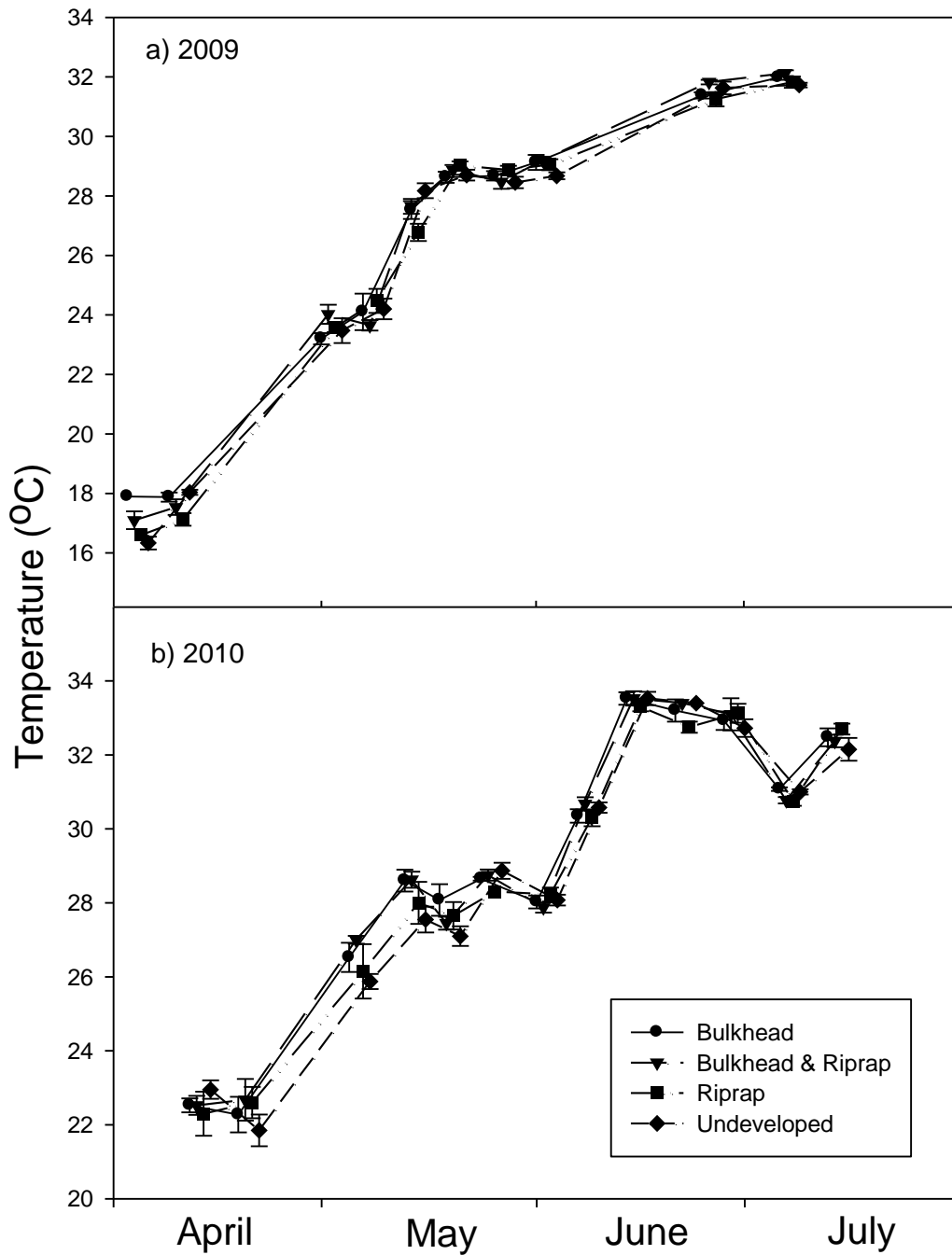
Figure 2. Average (± 1 SE) temperature (°C) plotted through time during a) 2009 and b) 2010 at each of four shoreline habitat types.
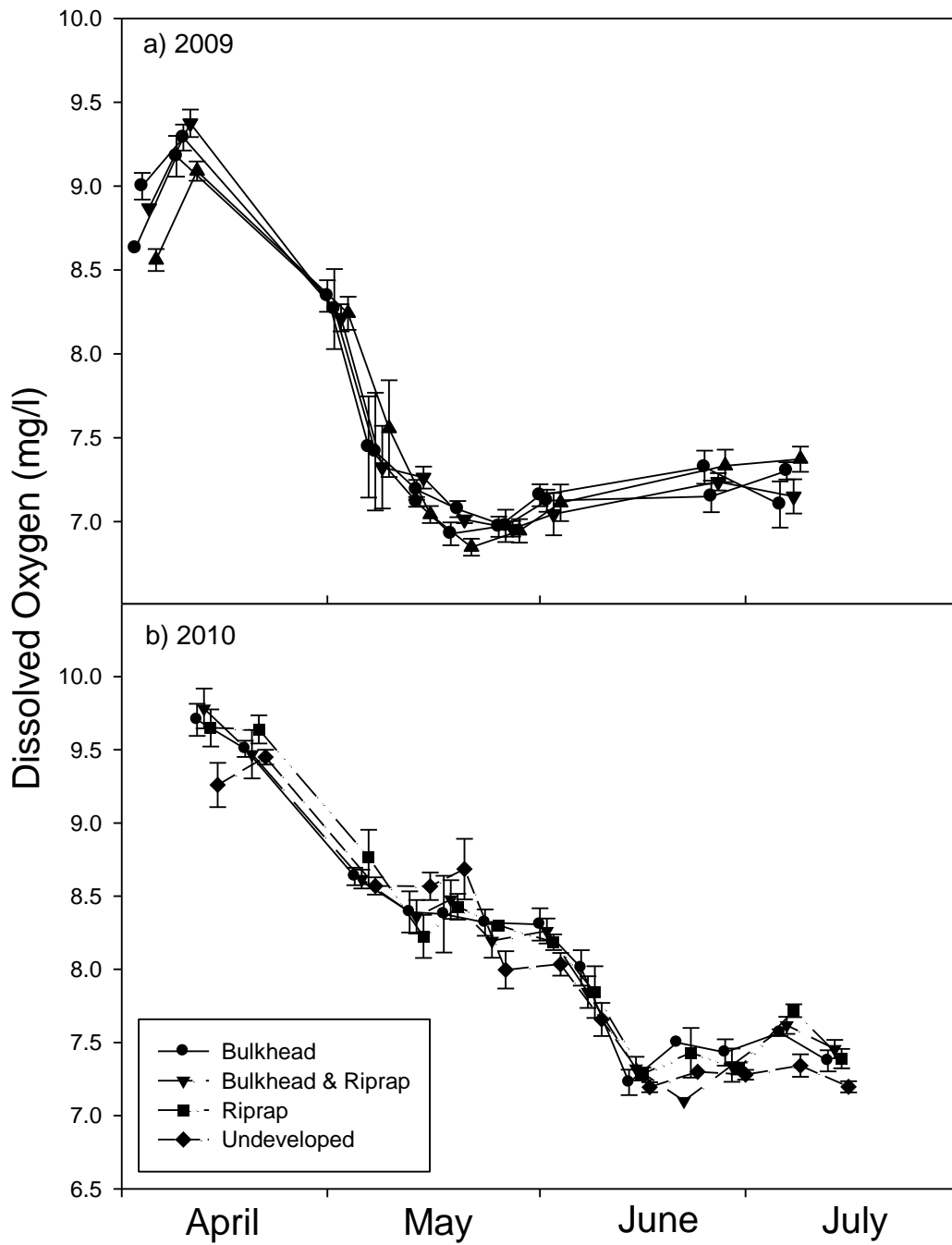
Figure 3. Average (± 1 SE) dissolved oxygen (mg/l) plotted through time during a) 2009 and b) 2010 at each of four shoreline habitat types.
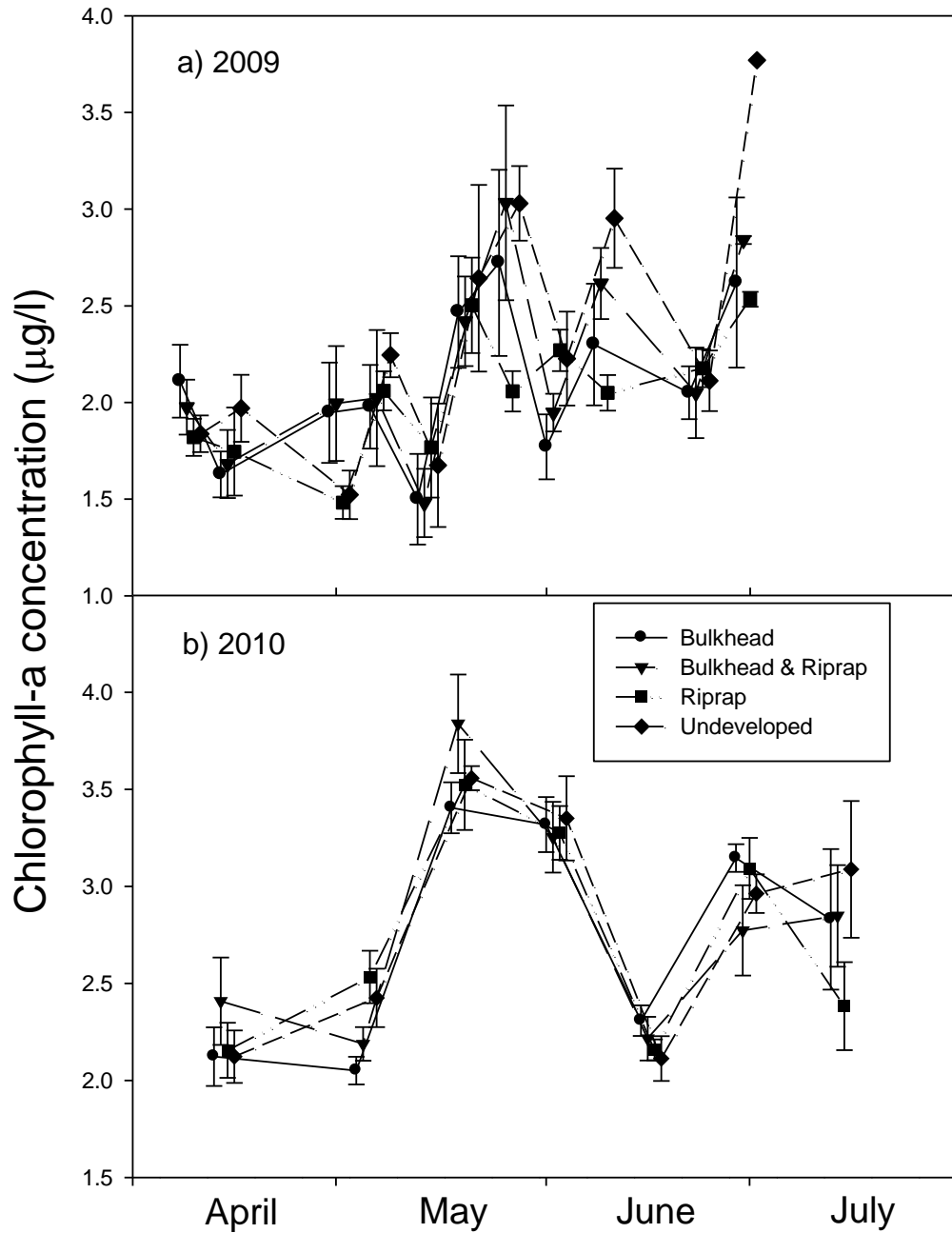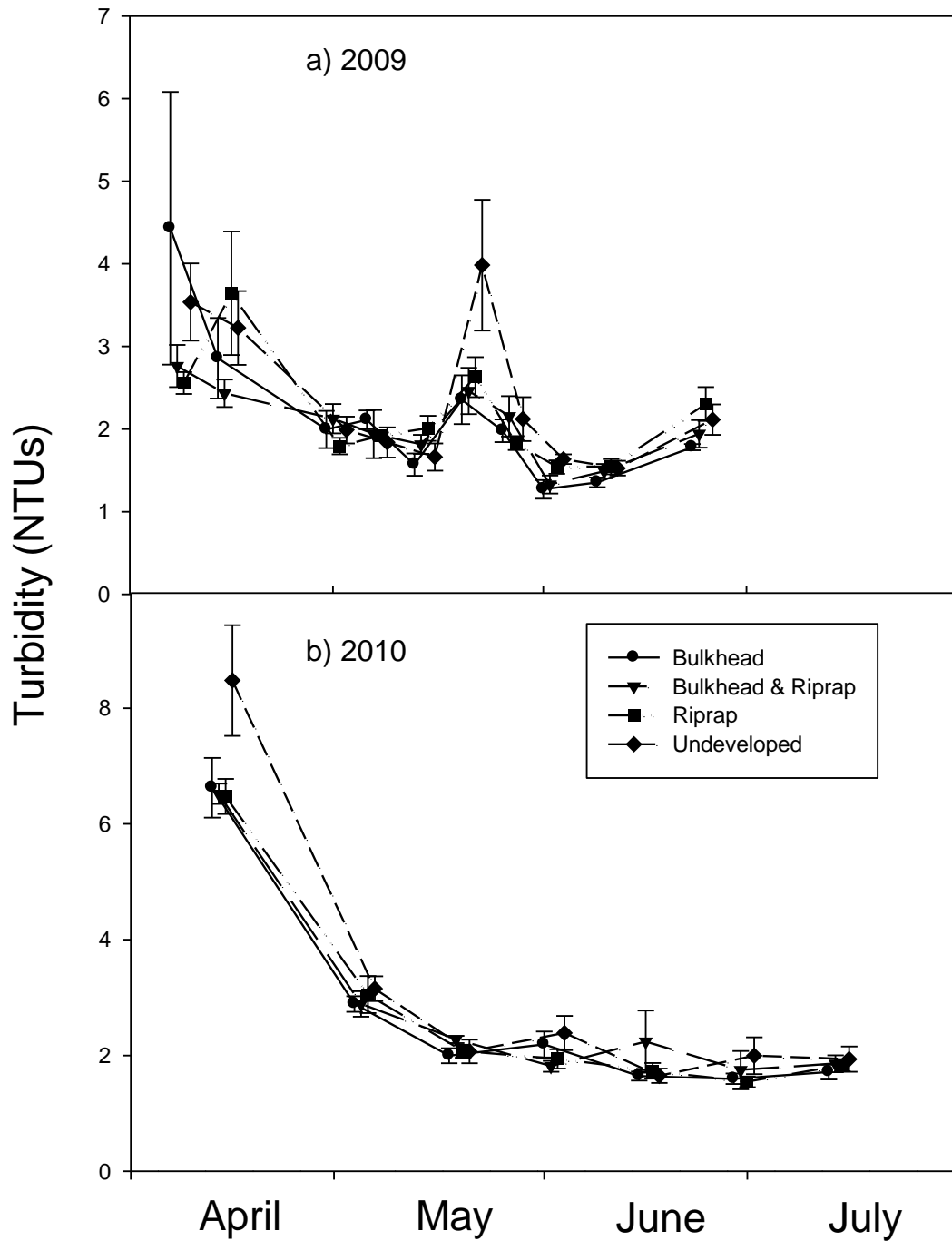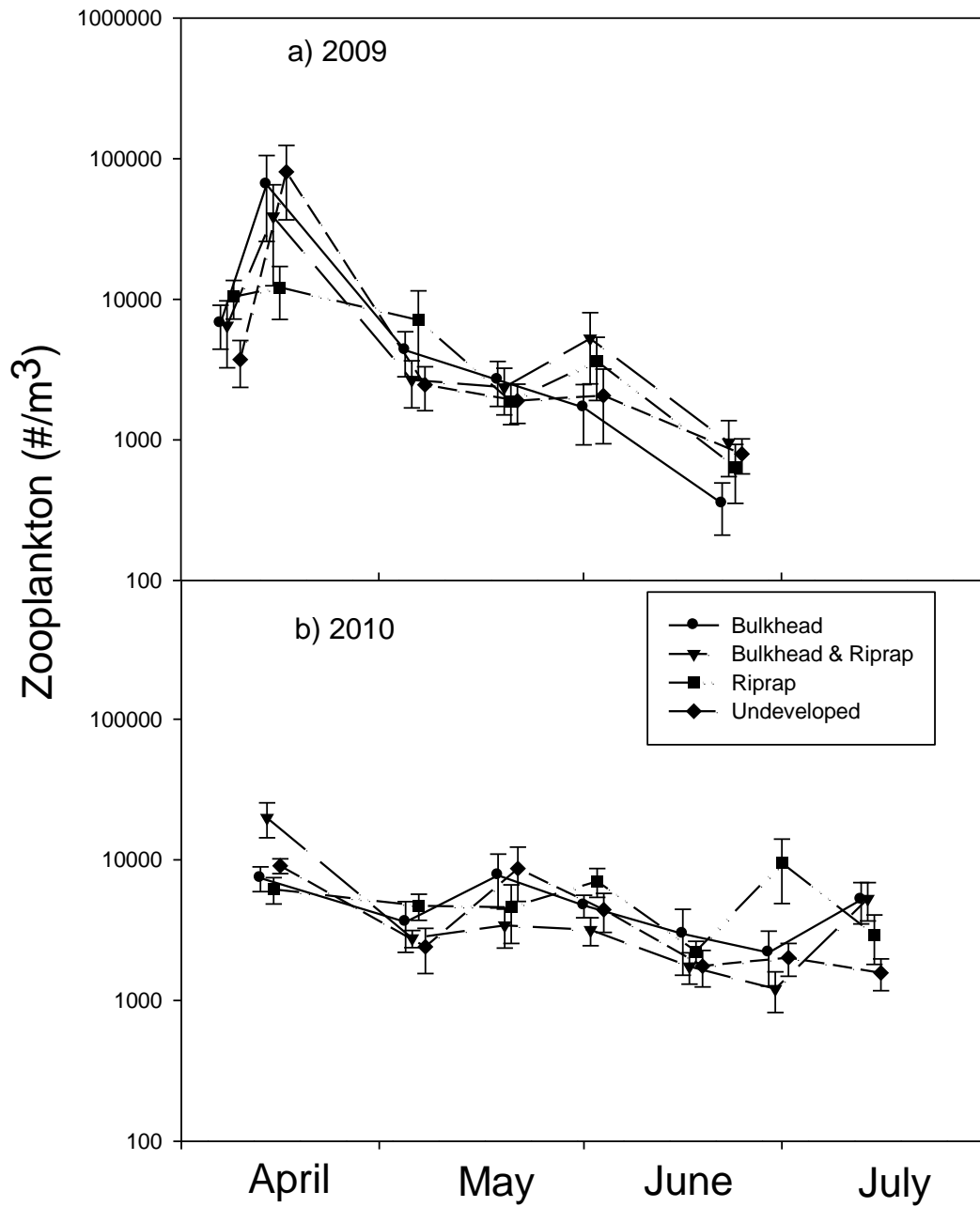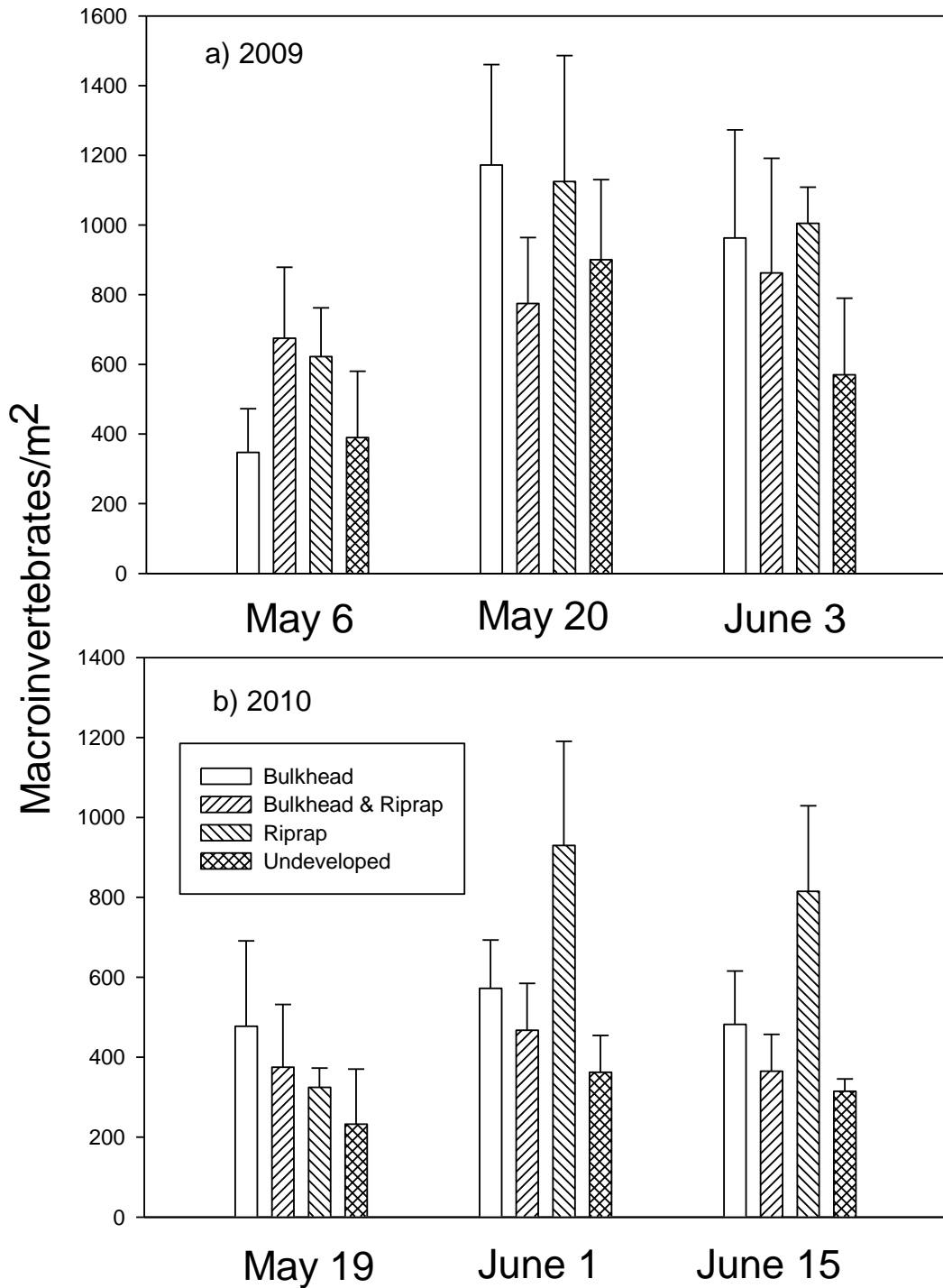
Figure 4. Average (± 1 SE) chlorophyll-a concentration (µg/l) plotted through time during a) 2009 and b) 2010 at each of four shoreline habitat types.
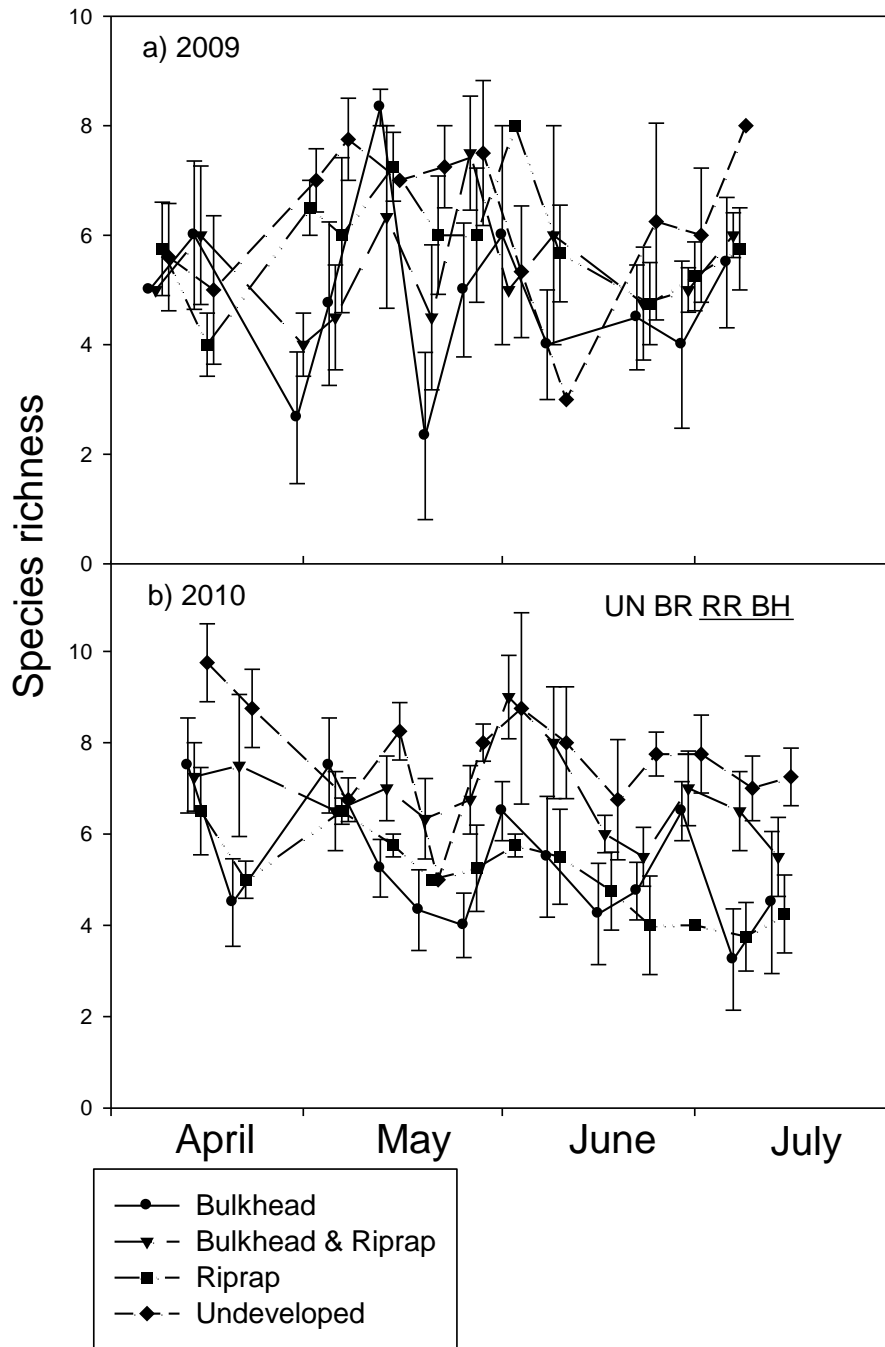
Figure 5. Average (± 1 SE) turbidity (NTUs) plotted through time during a) 2009 and b) 2010 at each of four shoreline habitat types.

Figure 6. Average ($\pm$ 1 SE) zooplankton density (#/m^3) plotted through time during a) 2009 and b) 2010 at each of the four shoreline habitat types.

Figure 7. Average (± 1 SE) macroinvertebrate density (#/m^2) in Hester-Dendy samplers plotted

for each of 3 dates during a) 2009 and b) 2010 at each of four shoreline habitat types.
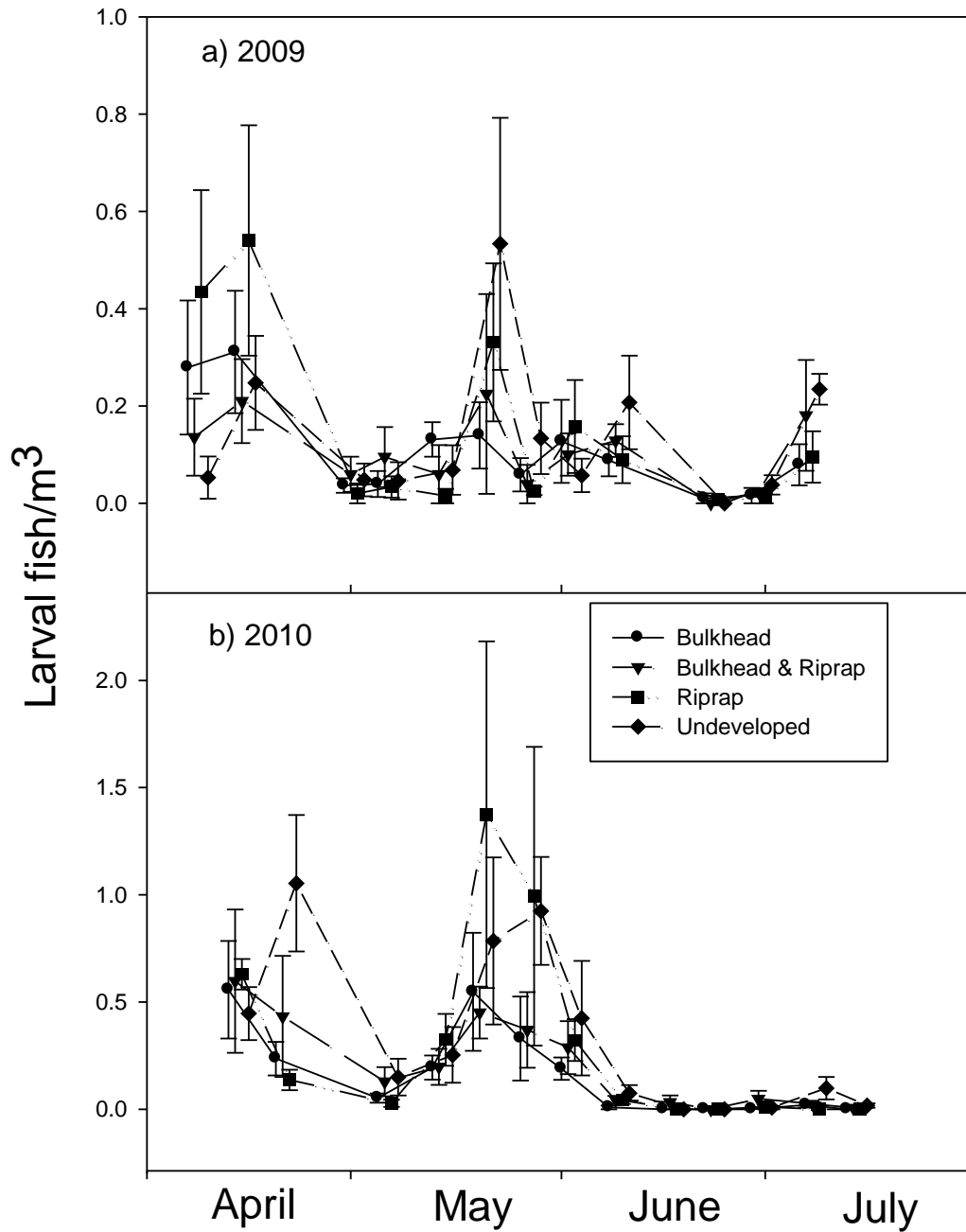
Figure 8. Average ($\pm$ 1 SE) larval fish density (individuals/m^3) from push net samples plotted through time during a) 2009 and b) 2010 at each of four shoreline habitat types.
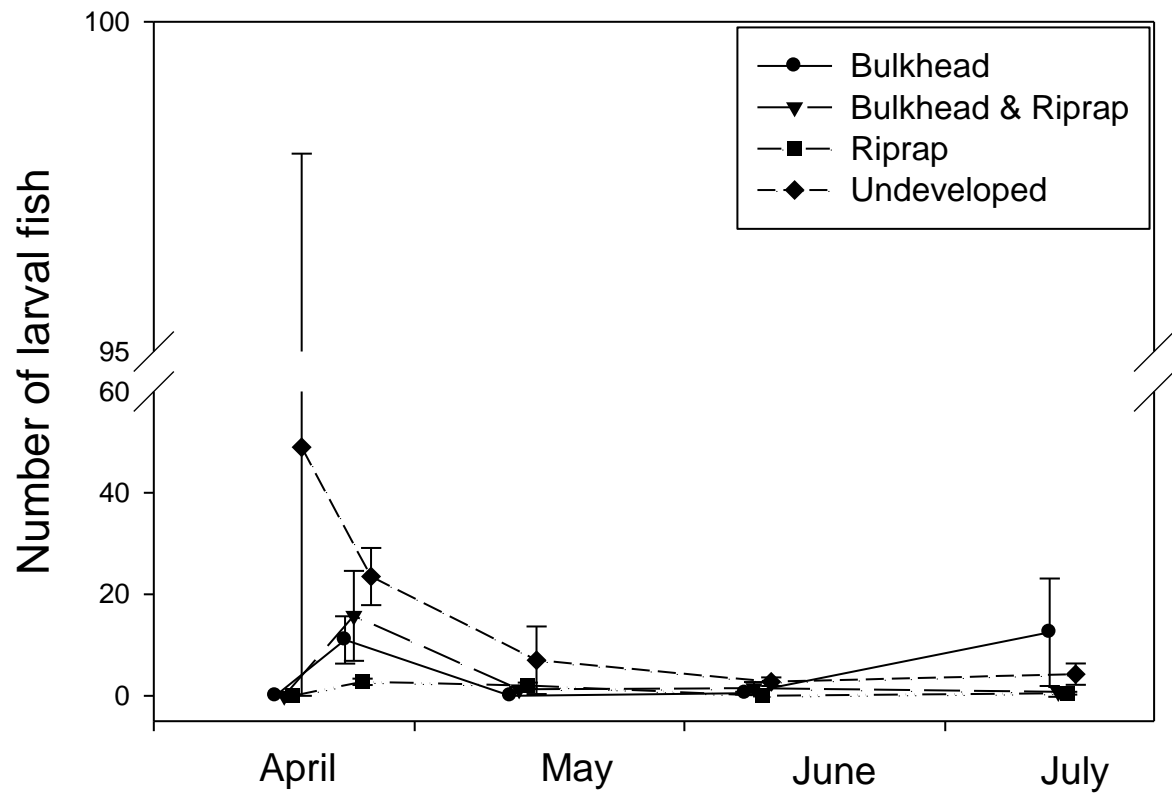
Figure 9. Average (± 1 SE) number of larval fish collected in light traps plotted through time during 2009 at each of four shoreline habitat types.
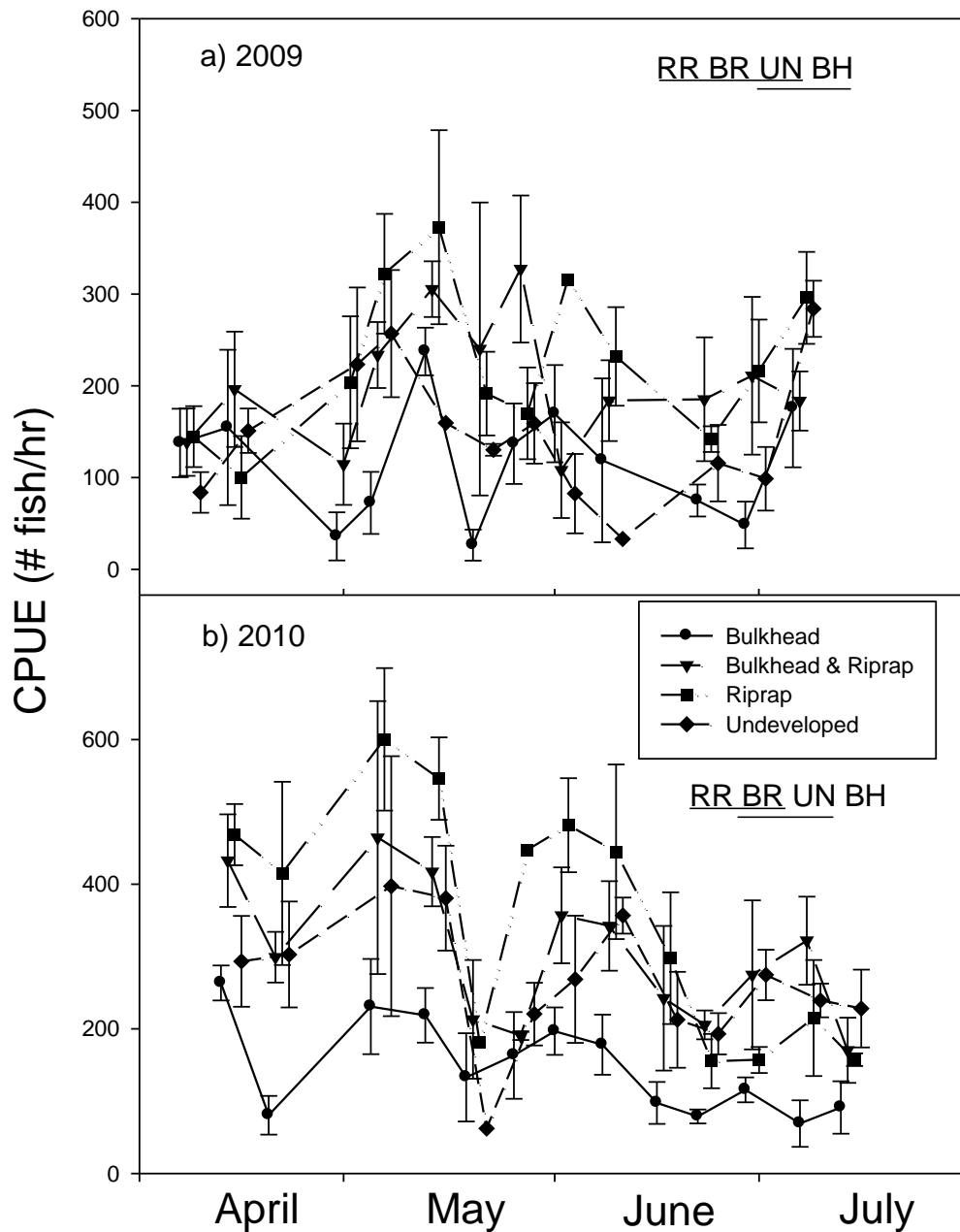
Figure 10. Fish abundances (expressed as catch-per-unit-effort = #fish/hr) plotted through time during a) 2009 and b) 2010 at each of four habitat types. Shoreline treatment abbreviations that share an underline did not differ significantly. BH=bulkhead, BR=bulkhead with riprap, RR=riprap, UN=undeveloped.
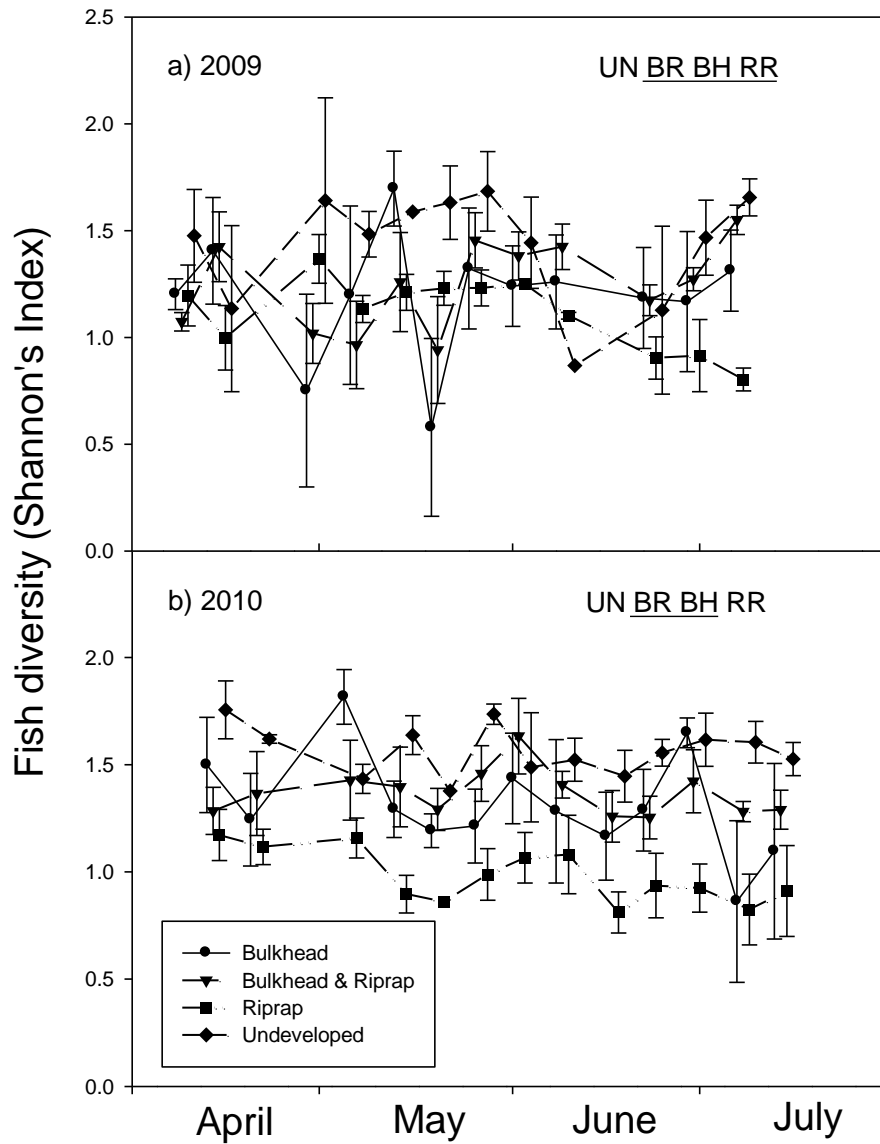
Figure 11. Average (± 1 SE) species richness plotted through time during a) 2009 and b) 2010 at each of four shoreline habitat types. Shoreline treatment abbreviations that share an underline did not differ significantly. BH=bulkhead, BR=bulkhead with riprap, RR=riprap, UN=undeveloped.

Figure 12. Average (± 1 SE) fish species diversity (Shannon's Index) plotted through time during a) 2009 and b) 2010 at each of four shoreline habitat types. Shoreline treatment abbreviations that share an underline did not differ significantly. BH=bulkhead, BR=bulkhead with riprap, RR=riprap, UN=undeveloped.
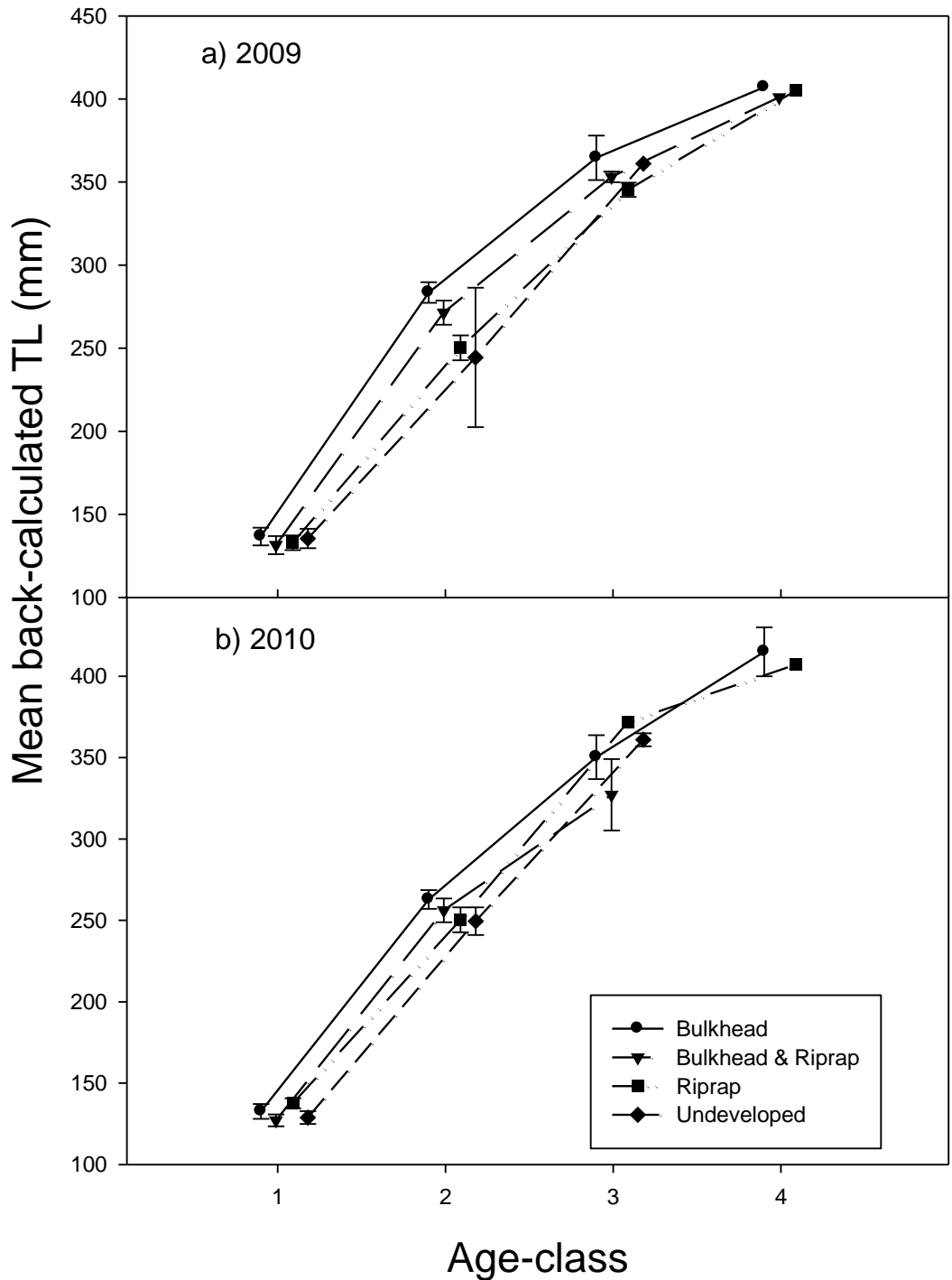
Figure 13. Average (± 1 SE) length-at-age for Alabama bass during a) 2009 and b) 2010 for each of four shoreline habitat types.
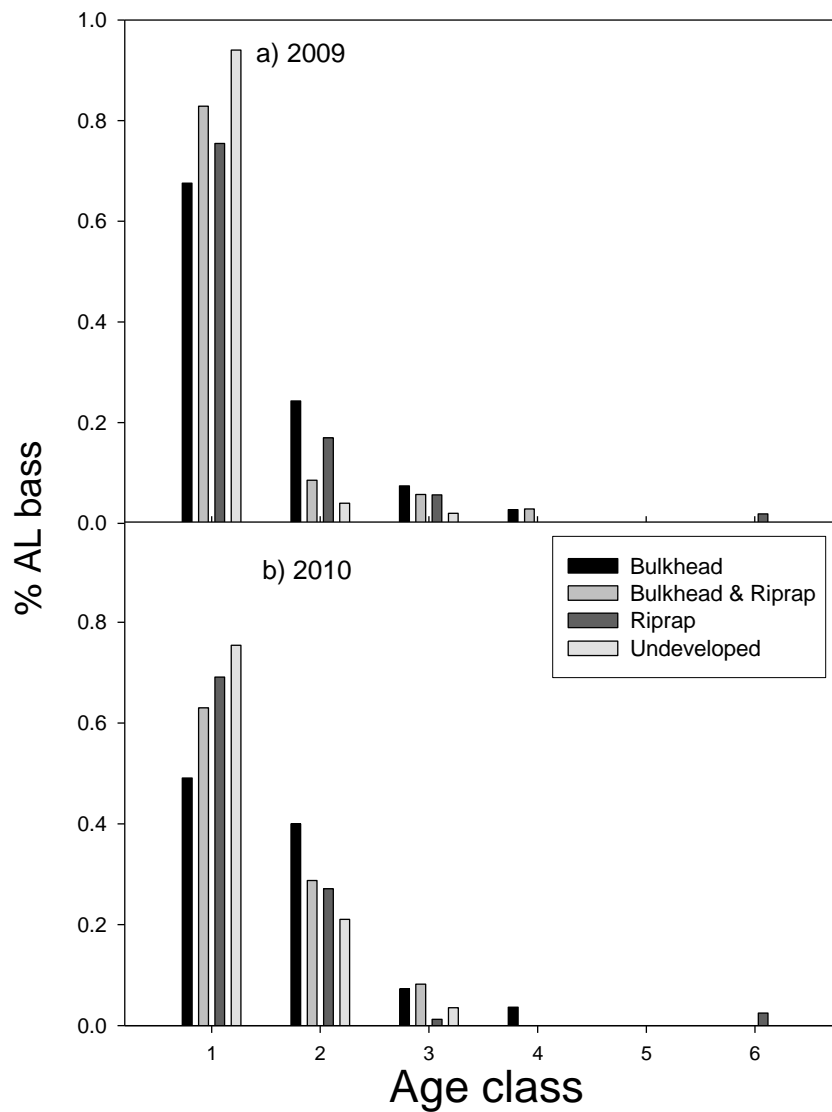
Figure 14. Age-frequency distribution of Alabama bass collected during a) 2009 and b) 2010 for each of four shoreline habitat types.
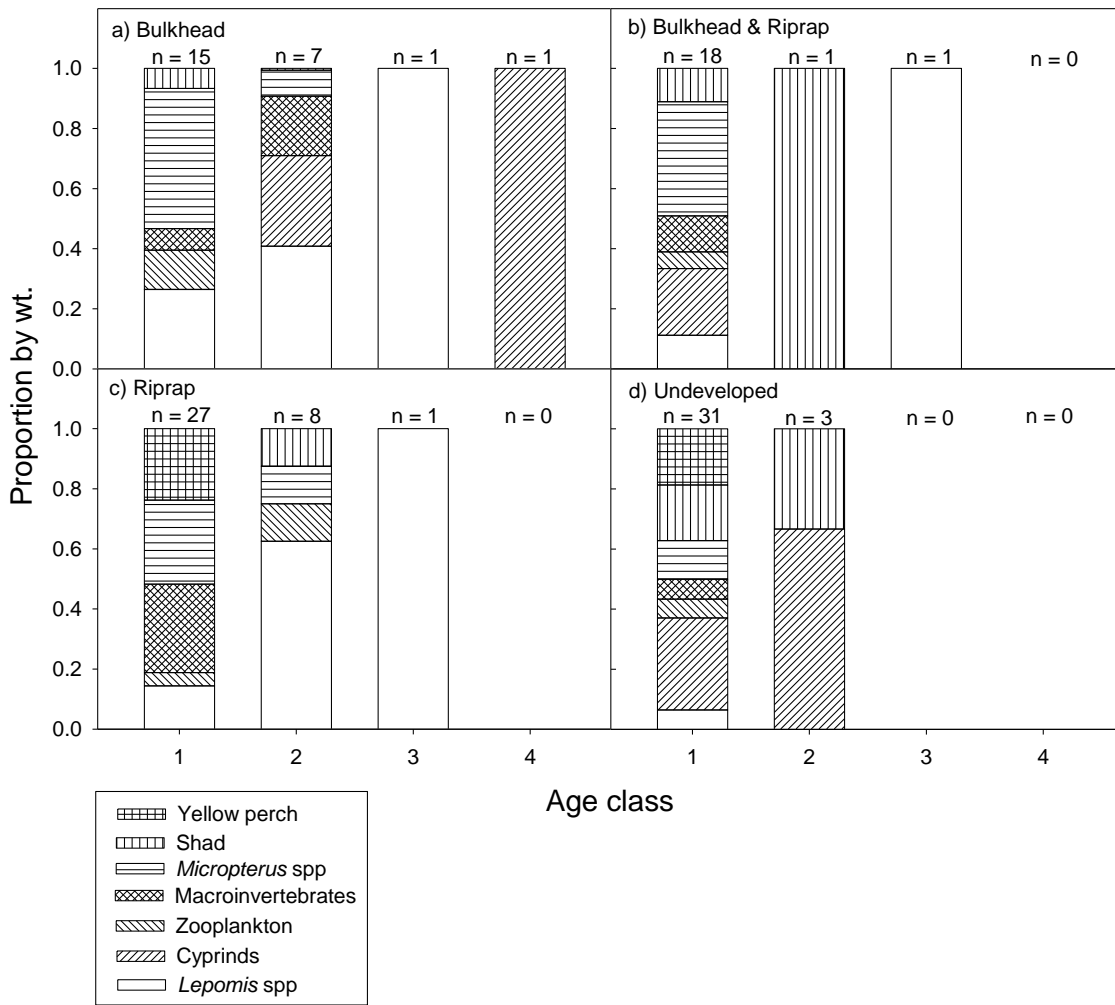
Figure 15. Proportion by weight of Alabama bass diets collected during 2009 at each of four shoreline habitat types (i.e., a) Bulkhead, b) Bulkhead and Riprap, c) Riprap, and d) Undeveloped), plotted by age-class. The number on top of each bar indicates the number of fish that had food items in the stomach.
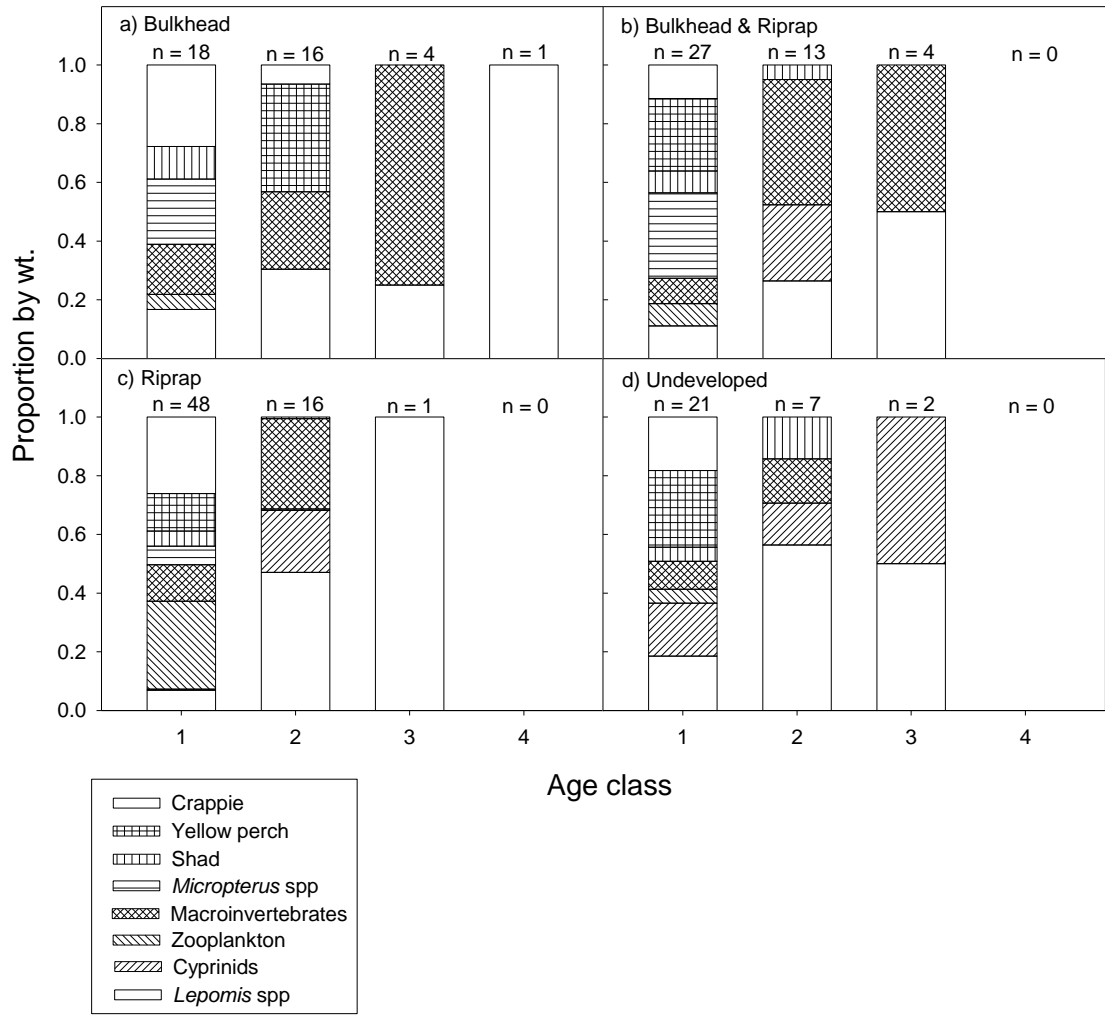
Figure 16. Proportion by weight of Alabama bass diets collected during 2010 at each of four

shoreline habitat types (i.e., a) Bulkhead, b) Bulkhead and Riprap, c) Riprap, and d)

Undeveloped), plotted by age-class. The number on top of each bar indicates the number of fish
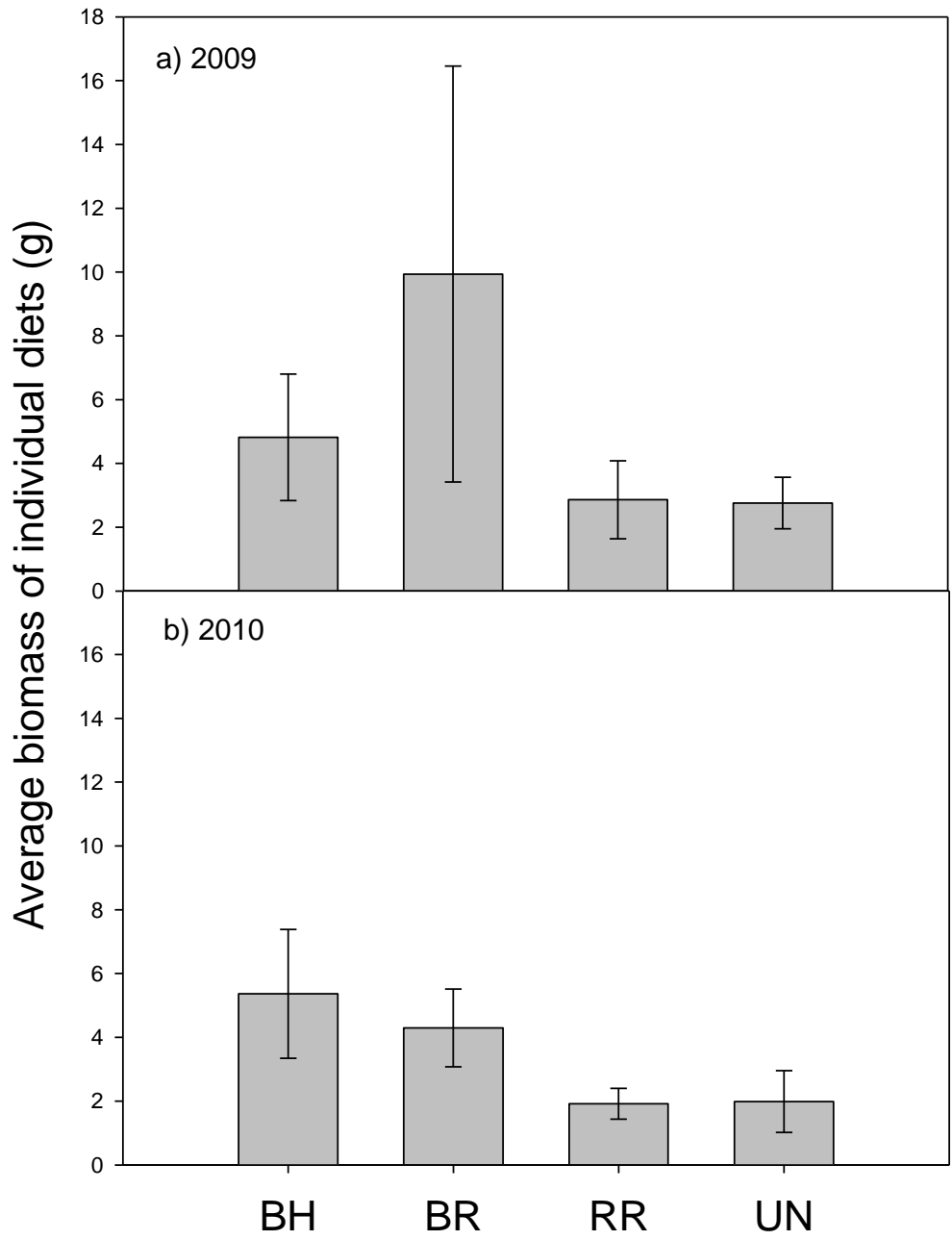
that had food items in the stomach.

Figure 17. Average (± 1 SE) biomass of individual Alabama bass diets plotted by shoreline habitat type for a) 2009 and b) 2010.