A Study of Native and Introduced Clupeids in Mobile River Basin Reservoirs

By
Ryan Jacob Bart

A thesis submitted to the graduate faculty of
Auburn University
in partial fulfillment of the
requirements for a Degree of
Master of Science
Auburn, Alabama

Auburn, Alabama
August 4, 2018

Keywords: Blueback Herring, hydroacoustics, bioenergetics, caloric density

Dennis R. DeVries, Advisor, Professor, Fisheries, Aquaculture, and Aquatic Sciences
Russell A. Wright, Advisor, Associate Professor, Fisheries, Aquaculture, and Aquatic Sciences
Conor McGowan, Research Associate Professor, Forestry and Wildlife Sciences

Abstract

In 2010, Blueback Herring *Alosa aestivalis*, a nonnative species were first found in Lewis Smith Lake, Alabama. Since this initial discovery, Blueback Herring have been found in several other waters in Alabama including Lake Martin, Yates Lake, and the Lewis Smith Lake dam tailrace. The introduction and population increase of Blueback Herring creates potential for competition with native fish like Threadfin Shad *Dorosoma petenense*. To quantify the abundance and habitat use of pelagic fishes, hydroacoustic surveys were conducted during July 2016, February 2017, and September 2017 in Lewis Smith Lake, Lake Martin, Bankhead Lake, and Yates Lake. Dissolved oxygen and temperature profiles were recorded down to 40 meters (or the lake bottom) in each lake to determine the available pelagic fish habitat. Summer dissolved oxygen was highest in the epilimnion, depleted at the thermocline, and increased again in the hypolimnion in Lewis Smith Lake and Lake Martin. The layer of cool oxygenated water beneath the thermocline was found to provide suitable habitat for large piscivores during the summer months, and hydroacoustics results suggest that this habitat is used. To assess the comparative forage values of Blueback Herring and Threadfin Shad, their energy densities were quantified using bomb calorimetry of fish from Lewis Smith Lake from April 2016 through December 2017. Caloric densities of Blueback Herring were higher relative to Threadfin Shad for all seasons. These values were incorporated into bioenergetics models to estimate the potential impact on piscivore growth. A positive growth potential was found if piscivores consumed higher proportions of Blueback Herring, however this does not warrant the introduction of this species in other systems.

Acknowledgments

I would like to thank my advisors, Dennis DeVries and Russell Wright for providing the opportunity to work under their direction and guidance during the research process. I would also like to thank Conor McGowan for serving on my committee and his input on my thesis. A special thanks to Tammy DeVries for her assistance processing calorimetry samples, lab workup of fish, and diet analysis. I thank Ben Staton for his assistance and patience when teaching me more efficient methods of analyzing my data. I would especially like to thank Patrick Anderson for the multiple field seasons at Smith Lake and am glad we can now laugh at all the logistical challenges we had while on the water. I would also like to thank all the grad students and technicians who have helped me in the field and in the lab including Lindsay White, Adrian Stanfill, Davis Walley, Byron Thomas, Garret Kratina, Mae Aida, Hennery Hershey, Leah Parker, Tyler Coleman, Mary Szoka, and Dustin McKee. I would also like to give a special thanks to the Alabama Department of Conservation and Natural Resources for funding this project.

Table of Contents

List of Tables

List of Figures

I. Density and Distribution of Pelagic Fish in Alabama Reservoirs

Introduction

There have been many introductions of non-native species to aquatic ecosystems, and interactions between native and non-native species have often complicated how these ecosystems and the fisheries therein are managed. Disrupting links in the food web can cause abiotic and biotic changes to aquatic ecosystems (MacIsaac 1996), sometimes leading to population declines of important native species (Hansen et al. 1995). The introduction of non-native species can also have effects on primary producers (Tumolo and Flinn 2017). In some cases, introduction of an invasive species can not only cause ecological disturbances, but also cultural and economic damage by diminishing subsistence and recreational fisheries (Pimentel et al. 2005). Regulating the transport of species outside their native range has been a primary strategy in keeping non-native species from becoming invasive, but as non-natives spread, managing their presence has become increasingly important.

Non-native species sometimes are spread intentionally often without full consideration of their potential impacts. For example, opossum shrimp *Mysis relicta*, native to Eastern North America, has been introduced in many lakes and reservoirs outside its native range as additional prey for gamefish. One such case was found to be successful in increasing both growth rate and size of Kokanee Salmon *Oncorhynchus nerka* in Kootenay Lake, British Columbia (Northcote 1972). However, when Mysis was introduced into Flathead Lake, Montana, Spencer et al. (1991) found that the Kokanee Salmon population collapsed, resulting in an ecological change that ultimately affected the annual aggregation of Bald Eagle *Haliaeetus leucocephalus*, Black Bear

Ursus americanus, and Brown Bear *Ursus arctos*, that gather to feed on migrating salmonids. These examples emphasize the complexity and underlying effect that can be involved and the need to consider possible interactions in aquatic ecosystems when introducing non-native species to meet management goals. Rainbow Smelt *Osmerus mordax* was first introduced to Crystal Lake, Michigan in 1912 and quickly expanded its range throughout the US Great Lakes (Evans and Loftus 1987). During the 1970s when predator densities were low in the Great Lakes, Rainbow Smelt populations grew rapidly. Predator populations increased as a result of the abundant forage source and eventually reduced the numbers of Rainbow Smelt; but, unfortunately Rainbow Smelt also had a negative effect by consuming larval coregonids (Gorman 2007). These examples show that there can be temporary benefits to introducing additional forage to a fishery, but over the long term, these introductions may permanently alter balanced ecosystems that were previously sustainable, often mediated through unexpected pathways.

Blueback Herring *Alosa aestivalis* is an anadromous species native to the Atlantic coast ranging from the St. John's River, Florida to Cape Breton, Nova Scotia (Loesch and Lund Jr, 1977). A classic paper by Brooks and Dodson (1965) found that Blueback Herring fed selectively on larger zooplankton which caused a shift to smaller zooplankton dominating the community when Blueback Herring invaded coastal lakes. This size-selective zooplanktivory was evident in Lake Theo, Texas where Blueback Herring selectively fed on large-bodied cladocerans, leading to a shift from a cladoceran dominated system to a copepod dominated system (Guest and Drenner 1991). These changes in zooplankton communities lead to increased competition among planktivorous fishes.

Despite their potential for negative impacts, Blueback Herring has become a popular bait choice for anglers in the Southeastern United States due to its effectiveness at catching sportfish. It has also been stocked into several reservoirs as additional forage. When Blueback Herring were introduced into Jocassee Reservoir, South Carolina, they consumed larger and more zooplankton prey than did native Threadfin Shad *Dorosoma petenense* (Davis and Foltz 1991) allowing them to outcompete Threadfin Shad (the native forage fish species in many Southeastern US reservoirs). Blueback Herring are more tolerant of low temperatures than are Threadfin Shad (Prince and Barwick 1981) which allows them to flourish in cold, deep reservoirs and inhabit areas unavailable to Threadfin Shad. Deep reservoirs in the Southeastern US have been extensively stocked with Striped Bass for recreational fishing (Sutton and Ney 2001; Churchill et al. 2002; Raborn et al. 2011). These deep-water ecosystems provide optimal habitat for pelagic piscivores and contain large amounts of forage such as Threadfin Shad and Blueback Herring to support them. In cases throughout the Southeastern US, Striped Bass have been found to get “squeezed” between the temperature above their thermal tolorance and available oxygen during warm months (Coutant 1985; Matthews et al. 1985; Moss 1985; Sammons and Glover 2013). When this occurs, they are vulnerable to anoxia and hypolimnetic dam discharges.

Sampling deep pelagic habitat is an obstacle when trying to assess the population with gears such as gill nets, midwater trawls, and electrofishing. By using hydroacoustics, most of the vertical profile can be sampled, allowing deep and shallow areas of the reservoir to be assessed. Originally developed for detection of submarines during World War I, hydroacoustics has been used to estimate fish stock densities since the early 1930s (Brandt 1996). A Hydroacoustic transducer sends an acoustic signal through the water column and records the depth, location, and strength of the echo that returns. Sampling is typically done using a transect design that follows the channel of the reservoir, multiple parallel transects over an area of interest, or a zigzag pattern associated with the shoreline (Jolly and Hampton 1990). This method is non-lethal, and allows for sampling of a large volume of water easily relative to the amount of time it takes to sample the fish community.

As with any sampling method, there are limitations to the use of hydroacoustics in the assessment of fish populations. Hydroacoustics sampling typically cannot identify fish to species directly from the signal. Verification collection with traditional gears such as gill nets (Dennerline et al. 2012) or midwater trawls (Burczynski et al. 1987) is required to apportion signal returns to species of fishes. Hydroacoustics equipment is also expensive, and requires specialized computer software and expertise to process the data. Even with these limitations, hydroacoustics has been widely used and to survey aquatic populations and help managers make better informed decisions when regulating fisheries (e.g., Pedersen and Boettner 1992; Stockwell et al. 2009; Ransom and Steig 1994).

The objectives of this chapter were to (1) estimate relative densities of pelagic fish in multiple reservoirs in the Mobile Basin, (2) quantify whether Blueback Herring and Threadfin Shad differ in their vertical distribution within the pelagic zone, (3) determine the longitudinal distribution upstream versus downstream, and (4) determine habitat use by large piscivores in the pelagic zone.

Methods

Site Descriptions

Lewis Smith Lake is a reservoir located in northwest Alabama impounding the Sipsey River in the upper reaches of the Mobile River in Winston, Cullman, and Walker counties. The reservoir consists of three main arms (Figure 1) and is considered to be mesotrophic (Bayne et al. 1998). The shoreline is moderately developed with many seasonal homes. The reservoir has a mean depth of 20 m and a retention time of 435 days (Bayne et al. 1998). Steep shorelines provide little littoral area for a reservoir with over 800 km of shoreline and 8,583 ha of surface area (Shepherd and Maceina 2009). The west and least productive arm is the Sipsey Creek arm, and the east or Ryan Creek arm is most productive, based on chlorophyll-a concentration. The middle arm is the Rock Creek arm which has a productivity intermediate between the other two arms. Blueback Herring has been established in Lewis Smith Lake since 2010 and is currently found throughout each arm (Grove 2016). I sampled the same seven sampling sites as Grove (2016), which included two sites in each arm and one in the forebay (Figure 1).

Lake Martin is a dendritic reservoir located in Tallapoosa, Coosa, and Elmore counties in east central Alabama and is part of the Tallapoosa River watershed. Lake Martin has a surface area of 16,188 ha, average depth of 13 m, and a maximum depth of 45 m (Bayne et al. 1989). In addition to the main channel, Lake Martin has two primary arms, the Kowaliga arm, and the Blue Creek arm (Figure 2). Blueback Herring have been found in Lake Martin but the date of their introduction or the number released is unknown.

Yates Lake is the reservoir immediately downstream from Lake Martin and is located in Elmore and Tallapoosa counties, Alabama. Yates Dam was constructed in 1928 and impounded an area of 801 ha (Bayne et al. 1989). Yates is a run-of-the-river reservoir (Figure 3) with an average depth of 5 m and a maximum depth of 18 m. The reservoir is polymictic due to upstream water releases from Martin Dam, in combination with being relatively shallow. A population of Blueback Herring has been identified in Yates Lake, although the size and extent of this population is unknown.

Bankhead Lake has a surface area of 3,723 ha and is downstream of Lewis Smith Lake (Bayne 1989). Bankhead Lake is located in Tuscaloosa, Walker, and Jefferson counties and is a run-of-the-river reservoir (Figure 4) with several shoals located outside the main channel. Blueback Herring have not been found in Bankhead Lake.

Hydroacoustics Surveys

Hydroacoustics equipment (BioSonics Inc., DT-X portable echo sounder, 420 kHz split beam frequency) was chosen to detect all sizes of fish including young-of-year clupeids (Rudstam et al. 2012). In cases when *Chaoborus* and other large-bodied zooplankton densities are high, 420 kHz frequency hydroacoustics can overestimate fish densities by reflecting off masses of invertebrates and producing a signal similar to fish (Degan and Wilson 1995). Although using high hydroacoustic frequencies are known to inflate density estimates due to *Chaoborus*, Degan and Wilson (1995) did not find significant differences in fish densities when using several different frequencies. The acoustic beam was directed downward perpendicular to the water surface. Transects were conducted in the middle of the channel in each arm starting at the upper reach and going downstream (Figure 1,2,3,4). Data were collected at night when pelagic fish tend to be more dispersed in the water column. Each transect was approximately 30 km in each arm and boat speed 7 km/hr.

Hydroacoustic data were collected during summer 2016, winter 2017, and summer 2017, and were also compared to survey data from summer 2014 from Grove (2016). Survey segments between surveys were compared using ANOVA and Tukeys HSD to determine significant differences for Lewis Smith Lake, Lake Martin, and Yates Lake. Bankhead Lake transect segments were compared using a t-test. Summer surveys were conducted when the reservoirs were stratified. In previous research on Lewis Smith Lake, Grove (2016), found there was an adequate amount of dissolved oxygen beneath the thermocline during summer. Hydroacoustics allowed me to estimate the abundance and sizes of fish using the pelagic habitat, including this deeper oxygenated stratum. Winter hydroacoustic surveys followed the same procedures as the summer transects. Dissolved oxygen and temperature profiles were taken at 2-m increments to the maximum length of the cable (40 m) or until the lake bottom was reached at the upstream, middle, and downstream segments of each arm in each reservoir to identify suitable habitat that is available to Blueback Herring, Threadfin Shad, and other pelagic fish species. Dissolved oxygen profiles were compared with hydroacoustics data to determine if fish were present at all depths (above and below the thermocline) where dissolved oxygen was deemed sufficient.

Love’s (1971) dorsal aspect equation was used to convert target strength of individual fish to estimated lengths:

$$TS = 19.1 * \log_{10} (L_m) + 0.9 * \mathrm{Log}_{10} (\lambda) - 23.9$$

where TS is the target strength of the returning echo from the fish, λ is the wave length of the acoustic signal sent out, and L_m is the length of the target (in m). Counts of single returns and estimated size distributions were analyzed using R software (R Core Team 2015). Similar to Vondracek and Degan (1995), estimates of densities were blocked in averages every 250 m to reduce correlation and the coefficient of variation.

Species Verification

Gill nets have been used successfully to supplement hydroacoustics data for species verification of targets (Balwin and McLellan 2008; Dennerline et al. 2012). Gill nets were set after hydroacoustics sampling at each of the standard sampling sites on Lewis Smith Lake (Figure 1). Gill nets were set at each site with two 30 m long with 2.5-cm stretch mesh gill nets, and two 38 m long with 5-18 cm stretch mesh gill nets suspended in each stratum for a total of four gill nets set at each of the seven standard sampling sites. All gillnetted fish were placed on ice in the field, brought back to the lab, identified to species, measured, and weighed.

Shocking in pelagic water is not a common sampling method for pelagic fish species due to its depth limitation. However, I found this method to be effective for sampling pelagic fish due to the diel movements of clupeids and vulnerability of smaller fish at night. Electrofishing was conducted by using a (Smith Root 7.5 GPP) electrofisher at night. Pelagic fish sampling was conducted on Lewis Smith Lake for the winter 2017 and summer 2017 surveys, Lake Martin for the summer 2017 survey, and Bankhead for the winter 2017 and summer 2017 surveys.

To understand annual fluctuations of clupeid species proportions in Lewis Smith Lake, electrofishing data from the standard sampling sites were used. Standard electrofishing consisted of 2, 10-minute transects at each site, and was conducted every month from March 2016 through November 2017. Proportions of clupeid species were quantified and categorized for each season. Proportions of gillnetted fish were used, but low catch rates of clupeids in the small mesh did not inform well of what species were present.

Results

Total Fish Densities

ANOVA and Tukey HSD post hoc results indicated there were significant differences between surveys within transect locations for Lewis Smith Lake (Figure 5, Table 1). The total fish density estimates for winter 2017 were extremely low compared to the summer surveys. The summer 2017 density estimate was lower than the summer 2016 density estimate (Figure 5), possibly due to differences in dissolved oxygen between surveys (Figure 6). The highest densities of fish for the summer and winter surveys were found in the epilimnion (Figure 7), an area with the highest dissolved oxygen concentration and primary productivity. Single target lengths showed that most targets were small fish (<10cm) across all of the surveys (Figure 8).

In Lake Martin, transect segments differed between surveys as indicated by the ANOVA and post hoc results (Figure 9, Table 1). The average density estimates of fishes for the entire lake followed the same pattern as Lewis Smith Lake by being lower in summer 2017 than in either summer 2014 or summer 2016 and being lowest of all during the winter 2017 survey (Figure 9). Dissolved oxygen for the summer Lake Martin surveys was highest at the surface and decreased as depth increased, although the summer 2016 survey showed an increase in oxygen just below the thermocline (Figure 10). Fish densities were highest for Lake Martin in the epilimnion and decreased with depth (Figure 11). Most single targets that were detected were small fish (<10 cm) for all surveys (Figure 12).

There were significant differences between surveys within transect segments for Yates Lake (Figure 13, Table 1). The average fish density for Yates Lake were higher during the summer 2016 survey than the summer 2017 survey (Figure 13). Whole lake winter 2017 fish density estimates were low relative to both summer surveys (Figure 13). The summer 2016 surveys indicate higher densities in the lower segment of the reservoir compared to the upper segment (Figure 13). Winter 2017 Yates lake fish densities had similar upstream and downstream densities (albeit low densities), and summer 2017 surveys yielded higher densities in the upstream segment and lower densities in the downstream segment of the reservoir. Dissolved oxygen concentrations during the surveys showed a steady decrease with depth (Figure 14). Winter 2017 dissolved oxygen profile showed a higher concentration of dissolved oxygen compared to the summer 2017 profile but still similarly declined with depth. Fish density by depth data for Yates Lake summer surveys had highest densities in the epilimnion, but were near the bottom of the reservoir during the winter 2017 survey (Figure 15). Most targets detected from Yates lake were small fish (<10 cm) (Figure 16).

In Bankhead Lake t-test comparisons indicated there were significant differences between surveys for all survey segments. Winter 2017 whole lake total fish densities in Bankhead Lake were much lower than summer 2017 whole lake fish densities (Figure 17). Winter total fish densities were higher in the downstream segment and lower in the upstream segment of the reservoir (Figure 17). In contrast, summer 2017 fish densities were higher in the upstream segment and lower in the downstream segment of the reservoir (Figure 17). Dissolved oxygen profiles for winter and summer 2017 surveys of Bankhead Lake showed a decline with depth (Figure 18). Dissolved oxygen was higher during the winter survey compared to the summer survey but decreased rapidly near the bottom of the reservoir. Bankhead Lake fish densities across depths showed the highest densities in the epilimnion for both the winter 2017 and summer 2017 surveys (Figure 19). Most fish that were detected from each survey were small (<10 cm) (Figure 20).

Species Verification

Pelagic electrofishing on Lewis Smith Lake associated with the winter 2017 survey yielded 506 Blueback Herring and 131 Threadfin Shad during 3.5 hours of electrofishing (Figure 21). There were no clupeids caught in gillnets during the winter 2017 species verification sampling. Pelagic electrofishing for the summer 2017 survey yielded 108 Blueback Herring and 120 Threadfin Shad for 1.16 hours of electrofishing (Figure 22). A total of 19 Blueback Herring and no Threadfin Shad were caught in the species verification gillnets during the summer 2017 survey (Figure 23).

Lake Martin pelagic electrofishing for the summer 2017 survey yielded 750 Threadfin Shad and 0 Blueback Herring during 1.66 hours of electrofishing (Figure 24). Although no Blueback Herring were sampled during the pelagic electrofishing, they are known to be present in Lake Martin. No gillnets were set in Lake Martin for species verification.

Pelagic electrofishing on Bankhead Lake for the winter 2017 survey yielded 159 Threadfin Shad, and 6 Gizzard Shad *Dorosoma cepedianum* (Figure 25). Summer 2017 pelagic electrofishing on Bankhead lake yielded 52 Threadfin Shad and 9 Gizzard Shad during 1.5 hours of electrofishing (Figure 26). No Blueback Herring were collected in either sampling effort. No gillnets were set in Bankhead lake due to the limited success collecting clupeids in gillnets in the other reservoirs.

Catch Rates

Electrofishing catch rates in Lewis Smith Lake for clupeids were highest during February 2017 for Blueback Herring and May and August 2017 for Threadfin Shad (Figure 27). Gillnetting catch rates in Lewis Smith Lake were low throughout the year for Threadfin Shad, but peaked in May 2017 for Blueback Herring (Figure 27). Threadfin Shad contributed a higher proportion of the clupeid catch during the standard electrofishing surveys compared to Blueback Herring in all but winter 2017 (Figure 28).

Habitat Use of Piscivores

Hydroacoustic results show large targets (≥35cm) were located in habitat immediately above and below the thermocline during the summer surveys in Lewis Smith Lake (Figure 29). The 2017 winter survey of Lewis Smith Lake showed most large targets were detected at depths >20 m, although numbers were extremely low (Figure 29). In Lake Martin, the summer 2014 and summer 2016 survey large targets were found at depths above and below the thermocline, and were distributed throughout the water column during the winter 2017 survey (Figure 30). Large targets from the summer 2017 survey in Lake Martin were only located in the epilimnion but extended deeper in the profile compared to previous summer surveys (Figure 30). Large targets during the Yates Lake surveys were located throughout the water column with no visible pattern to their distribution (Figure 31). Bankhead Lake large targets were detected primarily at a depth of 20 m for the winter 2017 survey and 10 m for the summer 2017 survey (Figure 32).

Discussion

Most fish detected with hydroacoustics were smaller than 10 cm which were considered prey for piscivores. Based on verification sampling, both Blueback Herring and Threadfin Shad were most abundant in Lewis Smith Lake, while Threadfin Shad were most abundant in the other three reservoirs. Winter hydroacoustics surveys yielded extremely low-density estimates for all surveys which could be due the limited ability to detect clupeids high in the water column. Large piscivores used the oxygenated layer immediately above and below the thermocline during the summer surveys, while the winter surveys indicated that they used a broader range of depths.

Below, I consider factors that might affect pelagic fish distribution in Southeastern US reservoirs as well as how Blueback Herring might influence these distributions.

Total Fish Densities

Dissolved oxygen concentration and temperature can influence the density and distribution of pelagic fish species. Blueback Herring have been found to be restricted by temperature and oxygen in reservoirs with cold hypolimnetic water (Nestler et al. 2002). Because Blueback Herring are known to use colder water temperatures than Threadfin Shad (Prince and Barwick 1981), I expected to see higher densities beneath the thermocline than what was observed in Lewis Smith Lake and Lake Martin. Most small fish were located in the epilimnion during all summer and winter surveys. Other studies have found when conducting hydroacoustic surveys, most fish were located just above the thermocline to take advantage of optimal temperature and dissolved oxygen (Mathews et al. 1985; Taylor et al. 2005). The location of prey fish could also depend on time of day. Appenzeller and Leggett (1995) conducted hydroacoustic surveys over multiple 24-hour periods to monitor diel movements of prey fish in Lake Memphremagog, Quebec, and found Rainbow Smelt *Osmerus mordax* to migrate from deeper water to shallower water during the night and back to deep water during the day. Given that my surveys were conducted at night, perhaps more pelagic prey fish would be located beneath the thermocline during the day.

I generally found fish densities to be highest upstream (e.g., Summer 2014 and 2016 Martin, Summer 2016 Yates, and Summer 2017 Bankhead surveys). Taylor et al. (2005) also found fish densities to be highest in upstream portions of Badin Lake, North Carolina, but concluded this finding might be due to habitat features such as temperature and prey. I observed high fish densities in several upstream survey segments, this pattern is likely due to temperature, season, or system-specific variability. In Lewis Smith Lake, Lake Martin, and Bankhead Lake, the upstream portions of each arm tend to be shallower allowing the water to warm faster in the spring. However, these characteristics can depend on the how each lake varies in size and shape.

Winter surveys yielded extremely low-density estimates across all reservoirs. Because this occurred in all of my study reservoirs, there is likely a common element that influenced such low estimates. Appenzeller and Leggett (1995) found when surface temperatures were <18°C, Rainbow Smelt occupied the water closest to the surface making them undetectable to the acoustic beam. To combat this obstacle, Taylor et al. (2005) mounted an additional sideward oriented transducer to detect fish nearer the surface. In my study, the transducer was oriented down which might explain why my winter density estimates were small relative to my summer surveys. I also conducted my surveys in the middle channel of the reservoir which would not be able to detect fish near the shore.

Yates Lake fish density distributions throughout the reservoir were difficult to predict, potentially due to the Lake Martin Dam releases. Given that Yates Lake is such a shallow system and Lake Martin Dam discharges high volumes of cold water into the lake, fish are consistently exposed to high variation in flow a temperature. This would also explain why for each survey, fish were either distributed near the upstream reach or downstream reach. Edwards (1978) found cold water releases can impact fish distribution and cause reduced nutrient availability. By consistently manipulating the aquatic environment, fish distributions are equally impacted.

Species Verification

Identifying the species detected with hydroacoustics is an integral part of applying the results to biological questions about an ecosystem. Capture methods such as midwater trawls, and gillnets have been a reliable indicator of species present in during hydroacoustics surveys

(Balwin and McLellan 2008; Emmrich et al. 2010). Although these capture methods were used, I found pelagic electrofishing to be the most effective at catching clupeids during the hydroacoustic survey periods, but this method was limited to sampling fish close to the surface. Although some studies have assigned hydroacoustic targets based on thermal stratification (e.g., Grove 2016; Nestler et al. 2002), in my study Blueback Herring were caught with electrofishing during the summer standard sampling, indicating they can tolerate higher temperatures than previously reported. Although I did not separate species based on thermal stratification, I did collect specimens to estimate what the hydroacoustic targets were for each survey.

Gill net species verification yielded only Blueback Herring for the summer 2017 verification. I expected gill nets to be more effective at catching clupeids based on the success in other studies success catching Threadfin Shad (Allen et al. 2000; Van Den Avyle et al. 1995). Although gillnets have enjoyed some success catching clupeids, they are size selective (Rudstam et al. 1984). Baldwin and McLellan (2008) illustrated gillnets were effective at verifying acoustic targets, although their study focused on Kokanee Salmon *Oncorhynchus nerka* which are much larger than inland clupeids. In combination with pelagic electrofishing data, gillnets were effective at sampling deeper depths and larger clupeids. Regular sampling of clupeids via standard electrofishing and gillnetting transects over the period of the study provided an index of species proportions in Lewis Smith Lake. This evidence gave additional support for verification sampling during the hydroacoustics surveys. From verification sampling, most small fish in Lewis Smith Lake are Threadfin Shad, Blueback Herring abundance is substantial abundant given their relatively recent introduction to Lewis Smith Lake.

Habitat Use of Piscivores

Large targets used strata immediately above and below the thermocline. These targets were likely Striped Bass due to their known pelagic activity and regular stocking in Alabama reservoirs (Shephard and Maceina 2009). Striped Bass occupy deeper limnetic areas as lakes start to stratify and water temperatures warm (e.g., Farquhar and Gutreuter 1989; Matthews et al. 1989; Schaffler et al. 2002). Below the thermocline, dissolved oxygen concentrations were as low as 2 mg/L for the Lewis Smith Lake and Lake Martin summer surveys. Thompson et al. (2010) found Striped Bass occupied temperatures 20-23°C as long as the dissolved oxygen concentration were at least 2 mg/L. Thompson et al. (2010) also found when dissolved oxygen reached hypoxic conditions (<2 mg/L) beneath the thermocline, Striped Bass relocated to the top of the thermocline which would explain why there were high concentrations of large targets above and below the thermocline during the summer surveys in Lewis Smith Lake and Lake Martin.

During the summer survey in Bankhead Lake, there was no increase in oxygen concentration beneath the thermocline; however, large targets still occupied the deeper water with low concentrations of dissolved oxygen just above the thermocline. This indicates Striped Bass were taking advantage of cold water habitat, similar to Lewis Smith Lake, during the summer in Bankhead Lake. There was no habitat use patterns observed of large target distribution by depth for Yates Lake, but this may be due to the lower detection and shorter survey transect lengths. Lake Martin Dam releases may also be influencing where large fish are distributed in Yates Lake given that the aquatic environment is constantly changing (Edwards 1978).

When studying Watts Reservoir in Tennessee, Cheek et al. (1985) found that during winter and early spring, Striped Bass were spatially distributed in the tributary arms in addition to the main body of the reservoir, and during summer fish were limited to the arms where dissolved oxygen concentrations were greater than 4 mg/L and temperatures were cooler (18-20°C) compared to the main channel (24°C). These observations support my hydroacoustic results and provide explanations to why large targets were located in deep water with low oxygen concentrations and cool water temperatures.

Results from this chapter suggest that Blueback Herring and Threadfin Shad occupy similar habitats in large reservoirs. Although Blueback Herring have been found to be thermally restricted to cold water (Prince and Barwick 1981), they were periodically caught during summer using pelagic electrofishing. Pelagic fish densities were highest during summer and lowest during winter, but this may be due to where they are located within the profile and the ability to detect them (Appenzeller and Leggett 1995). Although most prey fish were located in the epilimnion, most piscivores were located immediately above and below the thermocline. To get a better understanding of the impacts of Blueback Herring introductions, further research should be done on the thermal and dissolved oxygen tolerances of Blueback Herring.

II. Assessing the Caloric Density of Blueback Herring and Threadfin Shad in Lewis Smith Lake: Estimating the Impact on Piscivore Growth Rates

Introduction

Non-native species have been found to impact aquatic ecosystems (e.g., Knapp and Matthews 2000; Latini and Petrere 2004; Lowe et al. 2008). The effects of non-native introductions can vary from changing nutrient dynamics of aquatic habitats (Capps and Flecker 2013) to direct competition with native fish (Bergstrom and Mensinger 2009). Successful aquatic introductions are sometimes intentional (Jones et al. 1994) or unintentional (Magoulick and Lewis 2002), and are more successful at invading based on their ability to adapt to a new environment. Although introductions of non-native fish have often been intended to benefit aquatic ecosystems, negative effects often occur that are not predicted and may not manifest themselves until after the non-native has been established.

Non-native species can change predator-prey interactions within aquatic ecosystems. Magoulick and Lewis (2002) found non-native zebra mussels *Dreissena polymorpha* in Lake Dardanelle, Arkansas contributed a large proportion of prey to both Redear Sunfish *Lepomis microlophus* and Blue Catfish *Ictalurus furcatus* diets. Although prey with higher caloric densities were available, Redear Sunfish and Blue Catfish continued to consume a large proportion of the introduced zebra mussels, negatively affecting their growth. In the case of an intentional introduction, Jones et al. (1994) found Rainbow Smelt *Osmerus mordax* introduction to Horsetooth Reservoir, Colorado led to a 50% increase in Walleye *Stizostedion vitreus* growth.

Despite this initial increase in Walleye growth, a follow-up study by Johnson and Goettl (1999) found that Rainbow Smelt introduction resulted in reduced zooplankton densities, from 40-80 organisms/L to less than 1 organism/L. Walleye recruitment failed post introduction of Rainbow Smelt which was explained by predation on larval Walleye by Rainbow Smelt, and/or by the altered zooplankton community (Johnson and Goettl 1999). The introduction of non-native species can yield positive results, but all potential impacts should be considered before adding supplemental forage to an aquatic ecosystem.

The primary native forage in Lewis Smith Lake, Alabama is Threadfin Shad *Dorosoma petenense*, but the 2010 introduction of Blueback Herring *Alosa aestivalis* added a new planktivore and potential forage species to the system. Grove (2016) found little change in growth rates of Largemouth Bass *Micropterus salmoides* and Alabama bass *Micropterus henshalli* after Blueback Herring introduction; however, he did find a significant increase in relative weight of both black bass species. If the caloric density of Blueback Herring is higher than that of native forage, consumption of them could affect piscivore relative weight. The potential increase in energy return may also influence the behavior of the piscivores if they alter their habitat choice to overlap with Blueback Herring.

Caloric density varies across species (e.g., Bryan et al. 1996; Marchand and Boisclair 1998; Eggleton and Schramm 2002) and seasonally. For example, Walleye caloric density in West Blue Lake, Manitoba was highest in the fall and lowest in the spring (Kelso 1973). A prey species, Rainbow Smelt in Lake Michigan, had the highest caloric densities before winter and prior to spawning (Foltz and Norden 1977). Understanding species differences and seasonal variation in caloric density of forage fish, particularly for an introduced, non-native species, will help determine the overall effects an introduced species might have on the recipient system, including its piscivores.

Bioenergetics models are a tool that can be used to assess growth potential and consumption rates of fishes. The basic bioenergetic relationship is, that all energy consumed by a fish is either expended as respiration, waste, or converted to biomass of reproductive output (Brandt and Hartman 1993). Models can be used to simulate the effect of changing food type or amount on growth, or conversely to estimate the amount of food consumed needed for observed growth. Essington et al. (2001) used the von Bertalanffy size at age function to estimate growth in combination with bioenergetics modeling to estimate fish consumption. The relationship between consumption and growth can be affected by temperature, reproductive timing, activity cost, prey availability, metabolism, and predator and prey caloric densities (Hewett and Kraft 1993). If consumption is determined, prey proportions, caloric densities, and other parameters can be altered to estimate their effect on fish growth (e.g., Rice and Cochran 1984).

In this study, I quantified caloric densities of introduced Blueback Herring and native Threadfin Shad to determine their relative energetic value to gamefish species. I then incorporated these prey energetic values in bioenergetics simulations to determine potential growth differences for piscivores that consumed differing relative amounts of Blueback Herring and Threadfin Shad at ambient temperatures of each forage species.

Methods

Calorimetry

Because each arm of Lewis Smith Lake has different levels of productivity that could affect resident organism energy content, I chose to compare caloric densities of Blueback

Herring and Threadfin Shad in the two arms with the highest and lowest productivity. As such, I collected Blueback Herring and Threadfin Shad by electrofishing and gillnetting from the Sipsey (low productivity) and Ryan Creek (higher productivity) arms of Lewis Smith Lake (Figure 1) over 4 seasons for 2016 and 2017, defined as April-June = spring, July-September = summer, October-December = fall, and January-March = winter. To determine differences among seasons, I used a two-way analysis of variance with an interaction term (ANOVA). Differences in caloric density versus length were also compared using linear regression. Caloric density by length regressions were conducted by species and season.

All collected fish were euthanized with MS-222, placed on ice, and brought to the lab for processing. I measured total length (mm) and wet weight (g) for each specimen. Individual fish were oven dried at 70°C until they reached a constant weight, after which specimens were ground with a mortar and pestle into a homogenized powder (Glover et al. 2010). This powder was again oven dried to constant weight and the sample pressed into pellets approximately 0.10g to 0.20g in mass. Individual pellets were analyzed using a Parr 1425 semimicro bomb calorimeter. If two samples did not yield estimates of caloric density within 2% of one another, a third sample was run. The caloric density of all pellets from an individual fish were averaged to estimate the final caloric density. The dry weight caloric density estimates were then converted to wet weight caloric densities as:

$$\text{cal/g}_w = (\text{wt}_d / \text{wt}_w) * \text{cal/g}_d$$

where cal/g_d is the dry weight caloric density, wt_d is the dry weight of the individual, wt_w is the wet weight of the individual, and cal/g_w is the wet weight caloric density. Wet weight caloric densities were compared across seasons, between arms of the reservoir, and species. Caloric density estimates were then incorporated into the prey energy density in the bioenergetics model to determine their potential effect on the growth rate in mass of individual piscivores.

Piscivore Collection

Electrofishing and gillnetting were conducted monthly at the seven standard sampling sites on Lewis Smith Lake (red dots on Figure 1) during March 2016 through December 2017. During each sampling event, fish were collected, euthanized with MS-222, and brought back to the lab on ice. In the lab length, weight, sex, otoliths, and stomach contents were collected. All diet items were identified to species and measured to the nearest mm if the total length was obtainable or µm if otoliths were found. Otoliths found in the stomachs of piscivores were identified to the lowest taxonomic level using an otolith species key (unpublished data). All length estimates were converted to the estimated biomass using species specific biomass-length relationships. Diet proportions were calculated for individual fish and then averaged across individuals by season.

Piscivore Growth

Piscivore growth was determined by aging otoliths from piscivores collected during the standard sampling. Estimated length at age was determined by using the von Bertalanffy (1938) length at age function:

$$Lt = L\infty \left(1 - e^{-k\,(t - t_o)}\right)$$

where Lt represents length at time t, L∞ is the maximum theoretical length, k represents the growth rate, t is the time at the age of interest, and t_0 is the time at which length is 0. Length weight relationships were modeled using:

$$W=aL^b$$

where W represents the predicted weight at length, a is the intercept, L is the length, and b is the slope. These data were then incorporated into the bioenergetics simulations as the baseline growth with the current diet proportions.

Bioenergetic Simulations

To understand the growth potential in piscivores, bioenergetics simulations were conducted to model the differences in growth of piscivores that consumed different proportions of Threadfin Shad and Blueback Herring. To run the simulations, Fish Bioenergetics Model 4 was used which runs off the graphical user interface (Shiny) in program R (Deslauriers et al. 2017). The model is simply an energy balance equation:

$$C = G + (M + SDA) + F + U$$

where C is the total consumed energy, G represents growth, M represents respiration, SDA is specific dynamic action, F is waste lost due to excretion, and U is waste due to ingestion (Hartman and Hayward 2007). Averaged seasonal caloric density estimates from the energetic analysis of Threadfin Shad and Blueback Herring were used to conduct the bioenergetic simulations. Differences in growth were estimated for Striped Bass *Morone saxatilis*, Alabama bass *Micropterus henshalli*, and Largemouth Bass *Micropterus salmoides* due to their recreational importance and documented consumption of pelagic forage fish.

Bioenergetic models to estimate growth have been developed and tested for Largemouth Bass (Rice and Cochran 1984), and Striped Bass (Eldridge et al. 1982). Alabama Bass are a separate but similar species to Spotted Bass (Baker et al. 2008). Physiological parameters used in bioenergetic modeling have not been developed for Alabama Bass or Spotted Bass, however for the purpose of comparison I used the Largemouth Bass model to test for the effect of changing diet.

Temperatures in the model were determined from the standard sampling temperature profiles and location of large targets from hydroacoustic surveys (Chapter 1) (Figure 33). Two habitat scenario temperatures were used for the Striped Bass simulations; one of which represented water above the thermocline and one that represented water below the thermocline. A warmer habitat temperature was used for Alabama Bass and Largemouth Bass given that they occupy the same littoral habitat within the reservoir. The warmer habitat temperature was determined from the averaged Lewis Smith Lake temperature profiles from a depth of 2 m across sites for 2016 and 2017.

To determine potential growth differences due to shifts in diet to include more or less Blueback Herring for piscivores, bioenergetics model was used to estimate the annual total consumption and p-value (proportion of maximum consumption) by an individual piscivore. The models were fitted to one year of growth, based on the weight at age for each species, and habitat temperatures. One-year simulations were conducted for a fish growing from average size at age 4 to average size at 5 for Alabama Bass and Largemouth Bass, and from average size at age 6 to average size at 7 for Striped Bass. These ages were chosen to simulate adult life stages that rely on piscivorous feeding. The total mass of prey consumed by a predator or the p-value derived from the simulation with the observed diet proportions were kept constant in subsequent simulations. To simulate the effect of Blueback Herring and Threadfin Shad on piscivore growth, the proportion of Blueback Herring in the diet was varied from 0 to the total clupeid proportion (Threadfin Shad and Blueback Herring in the diet combined) observed from the diets collected. All other diet item proportions were kept the same as that observed in piscivore stomachs. Piscivore growth potential was then determined by looking at differences in growth at the end of the one-year simulations using the consumption and p-value from the observed diet simulation but altering the proportions of Blueback Herring and Threadfin Shad.

Results

Caloric Density Differences

Caloric density did not significantly differ between arms in Lewis Smith Lake for Blueback Herring ($F_{1,205} = 0.120$, $P = 0.720$) or Threadfin Shad ($F_{1,223} = 1.083$, $P = 0.299$). However, the arm season interaction was significant for Blueback Herring ($F_{3,205} = 7.758$, $P < 0.001$) and Threadfin Shad ($F_{3,205} = 2.758$, $P = 0.043$) (Figure 34). Although the interactions were significant, the general pattern in caloric density across seasons were similar between arms for each species (Figure 34).

Fish length had a significant effect on Blueback Herring caloric density during summer and fall (Figure 35, Table 2). Length also had a significant effect on caloric density for Threadfin Shad during the summer, fall, and winter (Figure 35, Table 2). Linear regressions for all fish analyzed, showed length had a significant positive effect on both Blueback Herring and Threadfin Shad caloric densities (Figure 36, Table 2). Threadfin Shad caloric densities increased faster than Blueback Herring, but the maximum lengths differed indicating that Blueback Herring had a higher caloric density when lengths of the two are equal.

Piscivore Diets

Five diet items were identified in Alabama Bass diets. Threadfin Shad contributed a larger proportion than did Blueback Herring across all seasons, with additional contributions from Brook Silverside *Labidesthes sicculus*, crayfish, and sunfishes (*Lepomis* spp.) (Figure 37). Largemouth Bass had the same 5 prey types in their diets as Alabama Bass, but clupeids contributed a much smaller proportion (Figure 38). Blueback Herring were only found in Largemouth Bass diets during winter and fall, and Threadfin Shad were only found during winter, spring, and fall (Figure 38). The proportion of Blueback Herring in Striped Bass diets were highest during the summer and lowest during the winter (Figure 39). Mean lengths of Blueback Herring consumed by piscivores were longer than Threadfin Shad across seasons (Figure 40). However, Threadfin Shad are found more often in piscivore diets compared to Blueback Herring with the exception of Striped Bass in summer (Figures 37,38,39).

Observed Piscivore Growth

Alabama Bass and Largemouth Bass von Bertalanffy growth curves were similar to one another in Lewis Smith Lake (Figure 41). Despite this similarity, von Bertalanffy model parameters predicted Largemouth Bass to have a higher L_∞ (Table 3). Striped Bass von Bertalanffy growth indicated they grew more than twice the length of black basses (Figure 41, Table 3). von Bertalanffy estimates that incorporated length-weight regressions resulted in weight-at-age predictions (Figure 42, Table 4) that were used in bioenergetics simulations.

Bioenergetics Simulations

Bioenergetics simulations evaluated the effects of temperature, diet proportions, and prey caloric density on Lewis Smith Lake piscivores. My data for Threadfin Shad and Blueback Herring caloric densities across seasons were included in the model, and other prey item caloric density values were obtained from published values (Table 5). Bioenergetic simulations for Alabama Bass demonstrated a positive growth potential as Blueback Herring contributed a larger proportion of their diets (Figure 43, Table 6). Largemouth Bass bioenergetics simulations demonstrated a much smaller growth potential compared to Alabama Bass when Blueback Herring contributed a larger proportion of their diet (Figure 44, Table 6). Striped Bass bioenergetics simulations show there was a large growth potential as Blueback Herring contributed a larger proportion of their diet in both warm and cold-water condition (Figure 45, Table 7). The warm water simulations showed that Striped Bass would consume more prey and grow to a larger size when in that habitat. All simulations showed that when the model was fit to the observed p-value rather than consumption, there was a larger piscivore growth potential.

Discussion

In this chapter, I quantified caloric densities of Threadfin Shad and Blueback Herring, and incorporated the estimated values in bioenergetics simulations to determine potential growth differences for piscivores. Blueback Herring caloric density was highest in spring and summer and lowest in the fall and winter, but was greater than that of Threadfin Shad across all seasons. Threadfin Shad caloric densities were highest in spring, summer, and fall, and lowest in winter. Regression analysis indicated that as length increased, so did caloric density for both species, however this fluctuates by season. Bioenergetics models indicated that a shift to feeding on a high proportion of Blueback Herring caused an increase in growth for piscivores. This relative increase was highest in Striped Bass and lowest in Largemouth Bass. Therefore, Blueback Herring do have the potential to exert positive influences on piscivore growth.

Caloric Density Differences

When Kelso (1973) recorded seasonal caloric density differences in Walleye, the variation was attributed to natural growth and metabolic processes. Foltz and Norden (1977) found consumption, fat stores, and reproduction determined seasonal caloric density in Western Lake Michigan Rainbow Smelt. Zooplankton abundance and planktivore density can also impact consumption by and energy content of fish (Madenjian et al. 2000). Although multiple factors can influence caloric density, I expected the caloric density of Blueback Herring and Threadfin Shad to be similar due to both being relatively the same size, same taxonomic family, and occupying similar habitat.

Piscivore Diets and Growth

Largemouth Bass diet proportions were similar to those in Grove (2016) with a relative low proportion of clupeids. This was expected due to their tendency to occupy nearshore habitats removed from pelagic clupeids. Striped Bass and Alabama Bass had higher proportions of clupeids in their diets likely because they were using pelagic habitat more than Largemouth Bass. The proportion of Blueback Herring in Striped Bass diets was lowest during winter and highest during summer, while Threadfin Shad contributed most during winter and least during the summer. Grove (2016) observed similar diet proportions across seasons for Striped Bass which may be a result of the habitats that pelagic fish are using. Given that Striped Bass, Threadfin Shad, and Blueback Herring are pelagic, they are more likely to encounter each other (Shaffler et al. 2003). Piscivores such as Striped Bass are known to target pelagic prey, while Largemouth Bass and Alabama Bass are typically located near shore.

On average, consumed Blueback Herring were larger than Threadfin Shad for all piscivores, although numerically more Threadfin Shad were consumed. Potentially, these observations were caused by either a higher abundance of Threadfin Shad relative to Blueback Herring, or piscivores selectively fed on Threadfin Shad. Rudershausen et al. (2005) found marine age-1 Striped Bass selectively fed on *Alosa* spp. but were unable to determine the mechanisms behind the selection. Although prey selection by piscivores has been observed, there is little physical difference between Threadfin Shad and Blueback Herring, so that is likely not a contributing factor. Hydroacoustics surveys indicated Threadfin Shad were more abundant in

Lewis Smith Lake supporting the observation of more Threadfin Shad in piscivore diets (Chapter 1). Although I observed size and quantity differences of clupeids in piscivore diets, this is potentially a result of where piscivores and prey are located by season.

Growth parameters of Largemouth Bass and Alabama Bass were similar. Generally, Alabama Bass do not get as large as Largemouth Bass, which can be seen from my von Bertalanffy growth parameters. Grove (2016) found the von Bertalanffy L_∞ growth parameter was 424.8 for Largemouth Bass, 506.9 for Alabama Bass, and 891.9 for Striped Bass. My data indicated an increased L_∞ for Largemouth Bass and Striped Bass, and a decreased L_∞ for Alabama Bass relative to Grove (2016). Although differences were observed in growth between Largemouth Bass and Alabama Bass, these differences may not be a direct impact from the introduction of Blueback Herring. Several factors can influence growth of piscivores. When studying an estuarian ecosystem, Glover et al. (2013) found Largemouth Bass growth rates depended on body size, distance from the marine source, and freshwater inflow. Changes in growth rates can also be influenced by changes in diets (Olsen 1996). Therefore, if habitat and quality of prey are optimized, the potential for piscivore growth increases.

Bioenergetics Simulations

Bioenergetics simulations indicated there is a positive effect on growth potential for all piscivores if they consumed larger proportions of Blueback Herring. This potential growth enhancement was influenced by observed caloric density differences between Blueback Herring and Threadfin Shad. Depending on the proportion of Threadfin Shad and Blueback Herring consumed, the extent of the potential growth was differentially affected. Because Striped Bass consumed the largest proportion of clupeids, they exhibited the largest growth potential. Largemouth Bass had the smallest growth potential due to the low proportion of clupeids in their diet likely because they occupy the shallow nearshore portions of the reservoir. Although models indicate there is a potential for increases in growth due to consumption of Blueback Herring, the introduction of a new prey does not guarantee a better recreational fishery. When significant growth increases were recorded in Walleye after the introduction of Rainbow Smelt to a Colorado reservoir, planktivore stocking appeared to represent a positive management strategy (Jones et al. 1994). However, Johnson and Goettl (1999) found the long-term effects of stocking Rainbow Smelt included reduced Walleye condition and ultimately a negative effect on the fishery. Although Blueback Herring introductions in Alabama have been recent, there is the potential for them to increase sizes of sportfish, with other ecological effects could follow.

Striped Bass bioenergetic simulations indicate ambient temperature can influence their growth and consumption. In deep reservoirs, Striped Bass have been found above and below the thermocline (Cheek et al. 1985). By using cold water habitat, Striped Bass can consume less prey to achieve the same growth as in warm water as indicated by bioenergetics simulations. Due to lower metabolic costs associated with lower temperatures (Clarke and Johnston 1999). Although using cold water habitat is metabolically beneficial, most prey fish were located in warmer water in the epilimnion. This suggests that Striped Bass may not be as thermally confined in Alabama Reservoirs relative to other water bodies (Cheek et al. 1985, Coutant 1985, Thompson et al. 2010).

Management Implications

The potential exists for some positive outcomes due to the introduction of Blueback Herring on the growth of piscivores particularly for Striped Bass in Lewis Smith Lake; however, the introduction of non-native species should not be considered for recreational fishing enhancement unless potential negative impacts of their interactions are assessed. Introduction events have had positive and negative changes to aquatic ecosystems (e.g., Bryan et al. 1996, Davis and Foltz 1991, Evans and Loftus 1987, Foltz and Norden 1977, Gorman 2007, Guest and Denner 1991, Johnson and Goettl 1999, Knapp and Matthews 2000, Lowe et al. 2008, Miller and Crowl 2006, Northcote 1972, and Prince and Barwick 1981). Unfortunately, the influences of these introductions are often difficult to predict and can only be confirmed after the usually irreversible introduction occurs. Before introductions of additional prey species to aquatic systems are used as a management strategy, careful consideration of potential negative and positive consequences should be simulated.

Literature Cited

Allen, M. S., M. V. Hoyer, and D. E. Canfield Jr. 2000. Factors related to Gizzard Shad and Threadfin Shad occurrence and abundance in Florida lakes. Journal of Fish Biology 57:291-302.

Appenzeller, A. R., and W. C. Leggett. 1995. An evaluation of light-mediated vertical migration of fish based on hydroacoustic analysis of the diel vertical movements of Rainbow Smelt (Osmerus mordax). Canadian Journal of Fisheries and Aquatic Science 52:504-511.

Baker, W. H., C. E. Johnston, G. W. Folkerts. 2008. The Alabama bass, Micropterus henshalli (Teleostei: Centrarchidae), from the Mobile River basin. Zootaxa 1861:57-67.

Baldwin, C. M. and J. G. McLellan. 2008. Use of gill nets for target verification of a hydroacoustic fisheries survey and comparison with kokanee spawner escapement estimates from a tributary trap. North American Journal of Fisheries Management 28:1744-1757.

Bayne, D. R., W. C. Seesock, E. Reutebuch, S. Holm. 1998. Lewis Smith Lake, phase 1 diagnostic/feasibility study, final report. Alabama Department of Environmental Management.

Bayne, D. R., W. C. Seesock, L. D. Benefield. 1989. Water Quality Assessment of Alabama Public Lakes. Alabama Department of Environmental Management.

Bergstrom, M. A., A. F. Messinger. 2009. Interspecific resource competition between the invasive Round Goby and three native species: Logperch, Slimy Sculpin, and Spoonhead Sculpin. Transactions of the American Fisheries Society 138:1009-1017.

Brandt, S.B., 1996. Acoustic Assessment of Fish Abundance and Distribution", Pages 385–432. Edited by: B. R. Murphy, D. W. Willis. In Fisheries techniques, American Fisheries Society, Bethesda, Maryland, 2nd edition.

Brandt, S. B. and K. J. Hartman 1993. Innovative approaches with bioenergetics models: future applications to fish ecology and management. Transactions of the American Fisheries Society 122:731-735.

Brooks, J. L. and S.I. Dodson. 1965. Predation, body size, and composition of plankton. Science 150:28-35.

Bryan, S. D., C. A. Soupir, W. G., W. G. Duffy, and C. E. Freiburger. 1996. Caloric density of three predatory fishes and their prey in Lake Oahe, South Dakota. Journal of Freshwater Ecology 11:153-161.

Burczynski, J. J., P. H. Michaletz, G. M. Marrone. 1987. Hydroacoustic assessment of the abundance and distribution of Rainbow Smelt in Lake Oahe. North American Journal of Fisheries Management 7:106-116.

Capps, K. A., and A. S. Flecker. 2013. Invasive aquarium fish transform ecosystem nutrient dynamics. Proceedings of the Royal Society 280:20131520.

Cheek, T. E., M. J. Van Den Avyle, and C. C. Coutant. 1985. Influences of water quality on distribution of Striped Bass in a Tennessee River impoundment. Transactions of the American Fisheries Society 114:67-76.

Churchill, T. N., P. W. Bettoli, D. C. Peterson, W. C. Reeves, and B, Hodge. 2002. Angler conflicts in fisheries management: A case study of the Striped Bass controversy at Norris Reservoir, Tennessee. Fisheries Management 27:10-19.

Clarke, A., and N. M. Johnston. 1999. Scaling of metabolic rate with body mass and temperature in teleost fish. Journal of Animal Ecology 68: 893-905.

Coutant, C. C. 1985. Striped Bass, temperature, and dissolved oxygen: a speculative hypothesis for environmental risk. Transactions of the American Fisheries Society 114:31-61.

Davis, B. M. and J. W. Foltz. 1991. Food of Blueback Herring and Threadfin Shad in Jocassee Reservoir, South Carolina. Transactions of the American Fisheries Society 120:605-613.

Degan, D. J., W. Wilson. 1995. Comparison of four hydroacoustic frequencies for sampling pelagic fish populations in Lake Texoma. North American Journal of Fisheries Management 15:924-932.

Dennerline, D. E., C. A. Jennings, and D. J. Degan. 2012. Relationships between hydroacoustic derived and gill net catch: implications for fish assessments. Fisheries Research 7889:123-124.

Deslauriers, D., S. R. Chipps, J. R. Breck, J. A. Rice, and C. P. Maden. 2017. Fish Bioenergetics 4.0: An R-Based Modeling Application. Fisheries 42: 586-596.

Edwards, R. J. 1978. The effect of hypolimnion reservoir releases on fish distribution and species diversity. Transactions of the American Fisheries Society 107:71-77.

Eggleton, M. A., and H. L. Schramm Jr. 2002. Caloric densities of selected fish prey organisms in the Lower Mississippi River. Journal of Freshwater Ecology 17:409-414.

Eldridge, M. B., J. J. Whipple, and M. J. Bowers. 1982. Bioenergetics and growth of striped bass, Morone saxatilis, embryos and larvae. Fisheries Bulletin 80: 461-474.

Emmrich, M., I. P. Helland, S. Busch, S. Schiller, T. Mehner. 2010. Hydroacoustic estimates of fish densities in comparison with stratified pelagic trawl sampling in two deep, coregonid-dominated lakes. Fisheries Research 105:178-186.

Essington, T. A., J. F. Kitchell, and C. J. Walters. 2001. The von Bertalanffy growth function, bioenergetics, and the consumption rates of fish. Canadian Journal of Fisheries and Aquatic Science 58: 2129-2138.

Evans, D. O., and D. H. Loftus. 1987. Colonization of inland lakes in the Great Lakes region by Rainbow Smelt, Osmerus mordax: their freshwater niche and effects on indigenous fishes. Canadian Journal of Fisheries and Aquatic Science 44: 249-266.

Farquhar, B. W., and S. Gutreuter. 1989. Distribution and migration of adult Striped Bass in Lake Whitney, Texas. Transactions of the American Fisheries Society 118:523-532.

Foltz, J. W. and C. R. Norden. 1977. Seasonal changes in food consumption and energy content of smelt (*Osmerus mordax*) in Lake Michigan. Transactions of the American Fisheries Society 106:230-234.

Glover, D. C., D. R. DeVries, and R. A. Wright. 2013. Growth of Largemouth Bass in a dynamic estuarine environment: an evaluation of the relative effects of salinity, diet, and temperature. Canadian Journal of Fisheries and Aquatic Science 70: 485-501.

Glover, D. C., D. R. DeVries, and R. A. Wright. 2010. Sample preparation techniques for determination of fish energy density via bomb calorimetry: an evaluation using Largemouth Bass. Transactions of the American Fisheries Society 139:671-675.

Gorman, O. T. 2007. Changes in a population of exotic Rainbow Smelt in Lake Superior: boom and bust, 1974-2005. Journal of Great Lakes Research 33: 75-90.

Grove, G. L. 2016. Evaluating the potential impacts of the introduced Blueback Herring, *Alosa aestivalis,* on the aquatic communities in Lewis Smith Lake, Alabama. Master's thesis. Auburn University, Auburn, Alabama.

Guest, W. C. and R. W. Drenner. 1991. Relationship between feeding of Blueback Herring and the zooplankton community of a Texas reservoir. Hydrobiologia 209: 1-6.

Hartman, K. J. and R. J. Hayward. 2007. Bioenergetics. Pages 515-560 in C .S. Guy, and M. L. Brown., editors. Analysis and Interpretation of Freshwater Fisheries Data. American Fisheries Society, Bethesda, Maryland.

Hartman, K. J., and S. B. Brandt. 1995. Estimating energy density of fish. Transactions of the American Fisheries Society 124:347-355.

Hansen, M. J., J. W. Peck, R. J. Schorfhaar, J. H. Selgeby, D. R. Schreiner, S. T. Schram, B. L. Swanson, W. R. MacCallum, M. K. Burnham-Curtis, G. L. Curtis, J. W. Heinrich, and R. J. Young. 1995. Lake trout (*Salvelinus namaycush*) populations in Lake Superior and their restoration 1959-1993. Journal of Great Lakes Research 21:152-175.

Hewett, S. W., and C. E. Kraft. 1993. The relationship between growth and consumption: comparisons across fish populations. Transactions of the American Fisheries Society 122:814-821.

Johnson, B. M. and J. P. Goettl Jr. 1999. Food web changes over fourteen years following introduction of Rainbow Smelt into Colorado reservoir. North American Journal of Fisheries Management 19:629-642.

Jolly, G. M. and I. Hampton. 1990. A stratified random transect design for acoustic surveys of fish stocks. Canadian Journal of Fisheries and Aquatic Science 47:1282-1291.

Jones, M. S., J. P. Goettl Jr., and S. A. Flickinger. 1994. Changes in Walleye food habits and growth following a Rainbow Smelt introduction. North American Journal of Fisheries Management 14:409-414.

Kelso, J. R. M. 1973. Seasonal energy changes in walleye and their diet in West Blue Lake, Manitoba. Transactions of the American Fisheries Society 102:363-368.

Knapp, R. A., and K. R. Matthews. 2000. Non-native fish introductions and the decline of the Mountain Yellow-Legged Frog from within protected areas. Conservation Biology 14:428-438.

Latini, A. O., and M. Petrere Jr. 2004. Reproduction of a native fish fauna by alien species: an example from Brazilian freshwater tropical lakes. Fisheries Management and Ecology 11:71-79.

Loesch, J. G., and W. A. Lund Jr. 1977. A contribution to the life history of the blueback herring, Alosa aestivalis. Transactions of the American Fisheries Society 106:6, 583-589.

Love, R. H. 1971. Dorsal-aspect target strength of an individual fish. Journal of the Acoustical Society of America 49: 816-823.

Lowe, S. R., D. J. Woodford, N. D. Impson, and J. A. Day. 2008. The impact of invasive fish and invasive riparian plants on the invertebrate fauna of the Rondegat River, Cape Floristic region, South Africa. African Journal of Aquatic Science 33:51-62.

Madenjian, C. P., R. F. Elliot, T. J. DeSorcie, R. M. Stedman, D. V. O'Connor, and D. V. Rottiers. 2000. Lipid concentrations in Lake Michigan fishes: seasonal, spatial, ontogenetic, and long-term trends. Journal of Great Lakes Research 26:427–444.

MacIsaac, H. J. 1996. Potential abiotic and biotic impacts of Zebra Mussels on the inland waters of North America. American Zoologist 36: 287-299.

Magoulick, D. D., and L. C. Lewis. 2002. Predation on exotic Zebra Mussels by native fishes: effects on predator and prey. Freshwater Biology 47:1908-1918.

Marchand, F. O., and D. Boisclair. 1998. Influence of fish density on the energy allocation pattern of juvenile Brook Trout (*Salvelinus fontinalis*). Canadian Journal of Fisheries and Aquatic Science 55:796-805.

Matthews, W. J., L. G. Hill, and S. M. Schellhaass. 1985. Depth distribution of Striped Bass and other fish in Lake Texoma (Oklahoma-Texas) during summer stratification. Transactions of the American Fisheries Society 114:84-91.

Matthews, W. J., L. G. Hill, D. R. Edds, and F. P. Gelwick. 1989. Influence of water quality and season on habitat use by Striped Bass in a large Southwestern reservoir. Transactions of the American Fisheries Society 118:243-250.

Moss, J. L. 1985. Summer selection of thermal refuges by Striped Bass in Alabama reservoirs and tailwaters. Transactions of the American Fisheries Society 114:77-83.

Nestler, J. M., R. A. Goodwin, T. M. Cole, D. Degan, and D. Dennerline. 2002. Simulating movement patterns of Blueback Herring in a stratified Southern impoundment. Transactions of the American Fisheries Society 131:55-69.

Northcote, T. G. 1972. Kootenay Lake: man's effects on the salmonid community. Journal Fisheries Research Board of Canada 29: 186-865.

Olsen, M. H. 1996. Ontogenetic niche shifts in Largemouth Bass: variability and consequences for first year growth. Ecological Society of America 77:179-190.

Pederson, M.G., and J.F. Boettner. 1992. Application of hydroacoustic technology to marine fishery management in Washington State. Fisheries Research 14:209-219.

Pimentel, D., R. Zuniga, D. Morrison. 2004. Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecological Economics 52: 273-288.

Popea, K. L., M. L. Brown, W. G. Duffy, P. H. Michaletz. 2001. A caloric-based evaluation of diet indices for Largemouth Bass. Environmental Biology of Fishes 61:329-339.

Prince, E. D., and D. H. Barwick. 1981. Landlocked Blueback Herring in two South Carolina reservoirs: reproduction and suitability and stocked prey. North American Journal of Fisheries Management 1:41-45.

Raborn, S. W., L. E. Miranda, and M. T. Driscoll. Effects of simulated removal of Striped Bass from a Southeastern reservoir. North American Journal of Fisheries Management 22:406-417.

Ransom, B. H., and T. W. Steig. 1994. Using hydroacoustics to monitor fish and hydropower dams. Lake and Reservoir Management 9:163-169.

RCoreTeam. 2018. R: A language and environment for statistical computing: R Foundation for Statistical Computing, Vienna, Austria. Available: www.R-project.org/.

Rice, J. A., and P. A. Cochran. 1984. Independent evaluation of a bioenergetics model for Largemouth Bass. Ecology 65:732-739.

Rice, J.A., J. E. Breck, S. M. Bartell, and J. F. Kitchell. 1983. Evaluating the constraints of temperature, activity, and consumption on growth of Largemouth Bass. Environmental Biology of Fishes 9:263-275.

Rudershausen, P. J., J. E. Tuomikoski, J. A. Buckel, and J. E. Hightower. 2005. Prey selectivity and diet of Striped Bass in Western Albemarle Sound, North Carolina. Transactions of the American Fisheries Society 134:1059-1074.

Rudstam, L. G., J. J. Magnuson, and W. M. Ton. 1984. Size selectivity of passive fishing gear: a correction for encounter probability applied to gill nets. Canadian Journal of Fisheries and Aquatic Science 41:1252-1255.

Rudstam, L.G., J.M. Jech, S.L. Parker-Stetter, J.K. Horne, P.J. Sullivan, and D.M. Mason. Fisheries acoustics. Pages 597-630 *in* A.V. Zale, D.L. Parrish, and T.M. Sutton, editors. Fisheries techniques, 3rd edition. American Fisheries Society, Bethesda, Maryland.

Sammons, S. M., and D. C. Glover. 2013. Summer habitat use of large adult Striped Bass and habitat availability in Lake Martin, Alabama. North American Journal of Fisheries Management 33:762-772.

Schaffler, J. J., J. J. Isely, and W. E. Hayes. 2002. Habitat use by Striped Bass in relation to seasonal changes in water quality in a southern reservoir. Transactions of the American Fisheries Society 131:817-827.

Shepherd, M. D., M. J. Maceina. 2009. Effects of Striped Bass stocking on Largemouth Bass and spotted bass in Lewis Smith Lake, Alabama. North American Journal of Fisheries Management 29:1232-1241.

Spencer, C. N., B. R. McClelland, J. A. Stanford. 1991. Shrimp stocking, salmon collapse, and eagle displacement. BioScience 41:14-21.

Stockwell, J. D., M. P. Ebener, J. A. Black, O. T. Gorman, T. R. Hrabik, R. E. Kinnunen, W. P. Mattes, J. K. Oyadomari, S. T. Schram, D. R. Schreiner, M. J. Seider, S. P. Sitar, and D. L. Yule. 2009. A synthesis of Cisco recovery in Lake Superior: implications for native fish rehabilitation in the Laurentian Great Lakes. North American Journal of Fisheries Management 29: 626-652.

Sutton, S.M., and J. J. Ney. 2001. Size-dependant mechanisms influencing first-year growth and winter survival of stocked Striped Bass in a Virginia mainstream reservoir. Transactions of the American Fisheries Society. 130:1-17.

Taylor, J. C., J. S. Thompson, P. S. Rand, and M. Fuentes. 2005. Sampling and statistical considerations for hydroacoustic surveys used in estimating abundance of forage fish in reservoirs. North American Journal of Fisheries Management 25:73-85.

Thomson, J. S., J.A. Rice, and D. S. Waters. 2010. Striped Bass habitat selection rules in reservoirs without suitable summer habitat offer insight into consequences for growth. Transactions of the American Fisheries Society. 139:1450-1464.

Tumolo, B. T., and M. B. Flinn. 2017. Top-down effects of and invasive omnivore: detection in long-term monitoring of large-river reservoir chlorophyll-a. Oecologia 185:293-303.

Van Den Avyle, M. J., J. Boxrucker, P. Michaletz, B. Vondracek, and G. R. Ploskey. 1995. Comparison of catch rate, length distribution, and precision of six gears to sample reservoir shad populations. North American Journal of Fisheries Management 15:940-955.

von Bertalanffy, L. 1938. A quantitative theory of organic growth. Human Biology 10:181-213.

Vondracek, B., and D. J. Degan. 1995. Among-and within-transect variability in estimates of shad abundance made with hydroacoustics. North American Journal of Fisheries Management. 15: 933-939.

Table 1. ANOVA statistics comparing differences between summer 2014, summer 2016, winter 2017, and summer 2017 surveys for each hydroacoustic segment.

Survey Differences by Segment			
Lewis Smith Lake			
Segment	**DF**	**F**	**P**
Rock	3;349	24.6	1.93 e-14
Sipsey	3;427	125.2	<2 e-16
Forebay	3;26	13.8	1.41 e-05
Ryan	3;381	49.95	<2 e-16
Whole Lake	3;1195	59.8	<2 e-16
Lake Martin			
Tallapoosa	3;64	158.6	<2 e-16
Upper Middle	3;138	12.0	5.17 e-17
Lower Middle	3;184	61.5	<2 e-16
Forebay	3;36	95.7	<2 e-16
Kowaliga	3;197	47.3	<2 e-16
Blue Creek	3;93	22.7	3.98 e-11
Whole Lake	3;732	22.4	7.46 e-14
Yates Lake			
Upper Middle	2;52	4.9	0.011
Lower Middle	2;68	31.6	1.93 e-10
Whole Lake	2;123	25.0	7.81 e-10

Table 2. Length effects on caloric density for Blueback Herring and Threadfin Shad by season. Asterisks represent statistical significance linear regression ($P < 0.05$) slope. All represents the effect of length on caloric density for all fish burned in calorimeter.

Effects of Length on Caloric Density Linear Regression Results

		Blueback Herring		**Threadfin Shad**	
	R^2	Intercept	Slope	Intercept	Slope
Spring	0.5287	1757.63	-3.0070	846.88	2.196
Summer	0.3512	1077.63	3.1901*	726.60	3.353*
Fall	0.2003	1385.49	-4.002*	573.58	6.168*
Winter	0.3382	976.21	0.4257	620.82	3.7065*
All	0.2051	1001.04	1.6887*	667.46	4.197*

Table 3. Von Bertalanffy length-at-age parameters for Alabama Bass, Largemouth Bass, and Striped Bass in Lewis Smith Lake, Alabama during 2016-2017.

Species	**Parameter**		
	k	**t_0**	**$L\infty$**
Alabama Bass	0.2343	-0.8262	553.31
Largemouth Bass	0.1562	-2.5699	593.61
Striped Bass	0.0981	-3.7091	1211.05

Table 4. Log transformed length weight regression parameters for Alabama Bass, Largemouth Bass, and Striped Bass

Species	**Parameter**	
	a	**b**
Alabama Bass	-12.1925	3.150
Largemouth Bass	-12.2872	3.173
Striped Bass	-11.2572	2.988

Table 5. Caloric density values (cal/g wet weight) for Lewis Smith Lake piscivores and prey types.

Prey Type	**Caloric Density (cal/g)**				**Reference**
	Winter	**Spring**	**Summer**	**Fall**	
Blueback Herring	1017	1370	1321	1064	This Study
Threadfin Shad	870	1029	980	1019	This Study
Brook Silverside	1050	1050	1050	1050	Popea et al. (2001)
Crayfish	750	750	750	750	Kelso (1973)
Lepomis spp.	1160	1160	1160	1160	Miranda and Muncy (1989)
Gizzard Shad	1220	1220	1220	1220	Miranda and Muncy (1989)
Largemouth Bass	4184	4184	4184	4184	Rice et al. (1983)
Striped Bass	6488	6488	6488	6488	Hartman and Brandt (1995)

Table 6. Alabama Bass and Largemouth Bass bioenergetics simulation results. “Observed Wt fit” represents the results obtained when the model was fit to the end weight of the piscivore. “Con fit” represents the results from when the model was fit to the consumption observed from the “Observed wt fit” simulation. “P-val fit” represents the results when the model was fit to the p-value from the “Observed wt fit” simulation. BBHR represents a Blueback Herring heavy diet and THSH represents a Threadfin shad heavy diet.

Prey Type	**Alabama Bass**					**Largemouth Bass**				
	Observed Wt fit	**BBHR Con fit**	**BBHR p-val fit**	**THSH Con fit**	**THSH p-val fit**	**Observed Wt fit**	**BBHR Con fit**	**BBHR p-val fit**	**THSH Con fit**	**THSH p-val fit**
Start Weight (g)	709	709	709	709	709	727	727	727	727	727
End Weight (g)	905	988	1086	882	862	925	943	957	922	920
P-val	0.362	0.346	0.362	0.366	0.362	0.379	0.376	0.379	0.379	0.379
Consumption (g)	2378	2378	2576	2378	2331	2553	2553	2586	2553	2548

Table 7. Striped Bass bioenergetics simulation results for warm and cold temperature scenarios. “Observed Wt fit” represents the results obtained when the model was fit to the end weight of the piscivore. “Con fit” represents the results from when the model was fit to the consumption observed from the “Observed wt fit” simulation. “P-val fit” represents the results when the model was fit to the p-value from the “Observed wt fit” simulation. BBHR represents a Blueback Herring heavy diet and THSH represents a Threadfin shad heavy diet.

Prey Type	**Cold Temperature Striped Bass**					**Warm Temperature Striped Bass**				
	Observed Wt fit	**BBHR Con fit**	**BBHR p-val fit**	**THSH Con fit**	**THSH p-val fit**	**Observed Wt fit**	**BBHR Con fit**	**BBHR p-val fit**	**THSH Con fit**	**THSH p-val fit**
Start Weight (g)	4895	4895	4895	4895	4895	4895	4895	4895	4895	4895
End Weight (g)	5836	6386	6788	5465	5243	5936	6440	7000	5432	5154
P-val	0.311	0.298	0.311	0.321	0.311	0.317	0.302	0.317	0.328	0.317
Consumption (g)	16526	16526	17688	16526	15749	20101	20101	21853	20101	19044

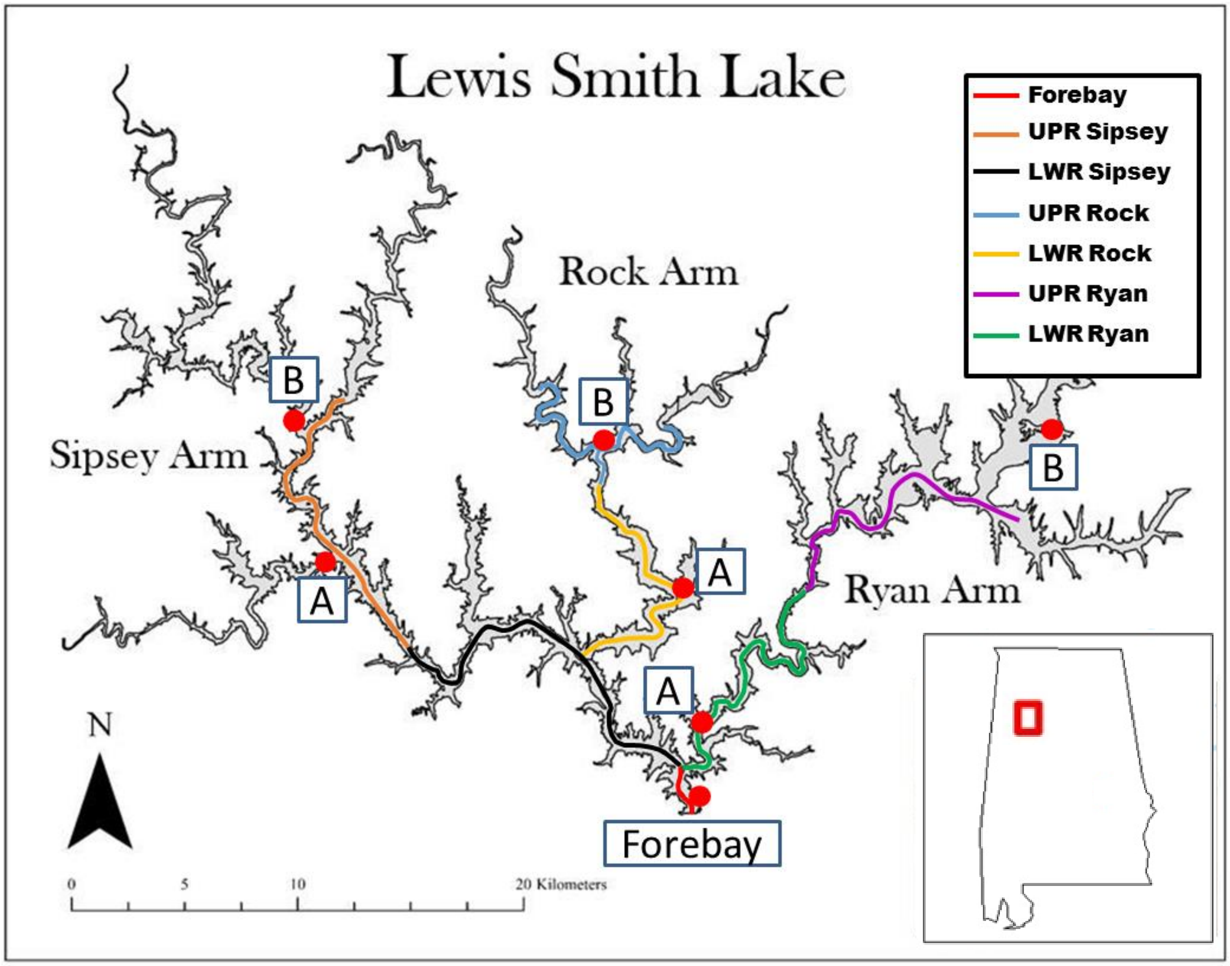


Figure 1. Location of Lewis Smith Lake, Alabama and Sipsey, Rock, and Ryan creek arms with hydroacoustic transect segments (indicated by colored lines) and standard sampling sites (indicated by red dots). Upstream sites are indicated by a letter ”B”, and downstream sites are represented by an “A” in each arm. A single site is located at the forebay. to represent all areas of the reservoir.

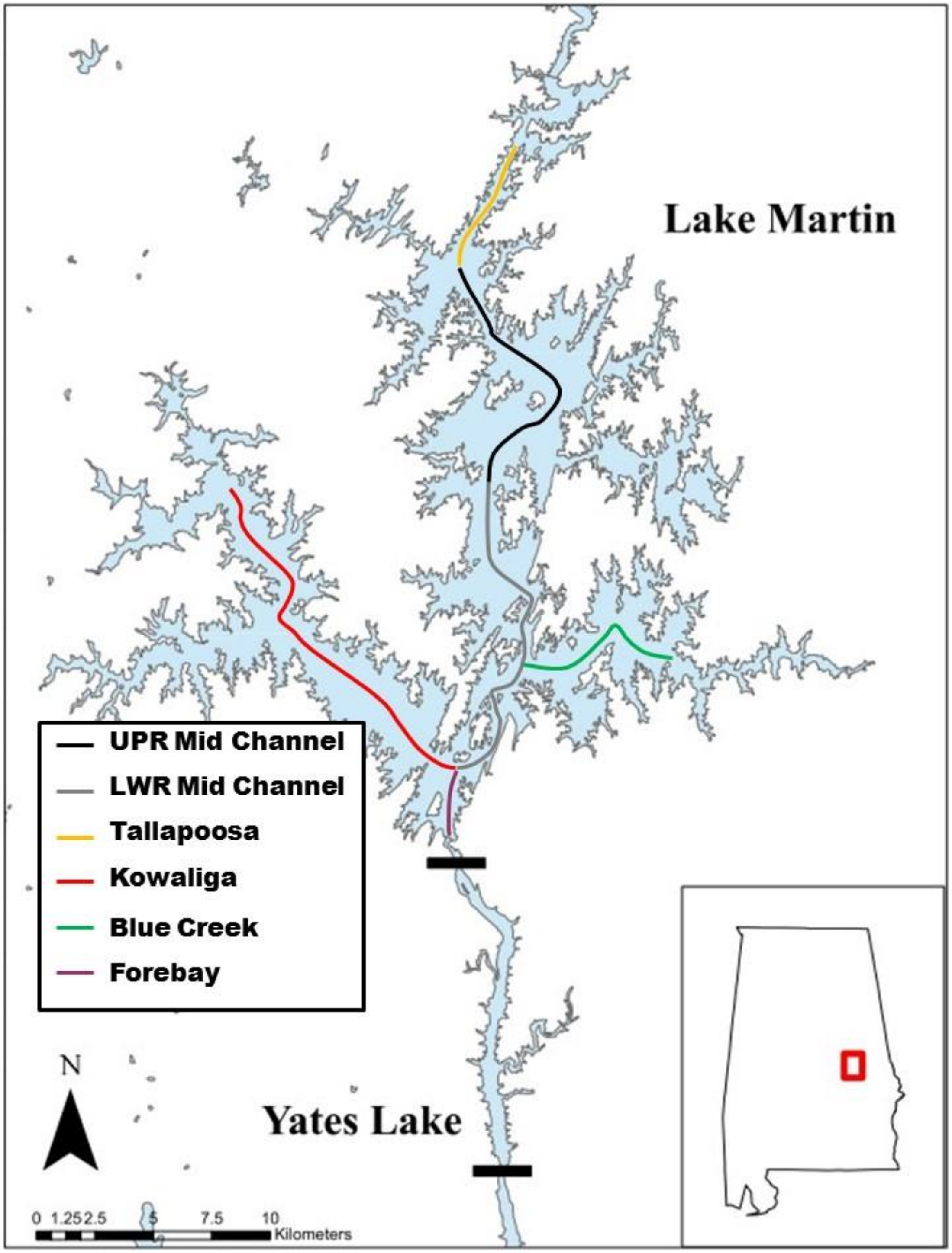


Figure 2. Location of Lake Martin, AL and Yates Lake, AL with Lake Martin hydroacoustic transect segments indicated by colored lines.

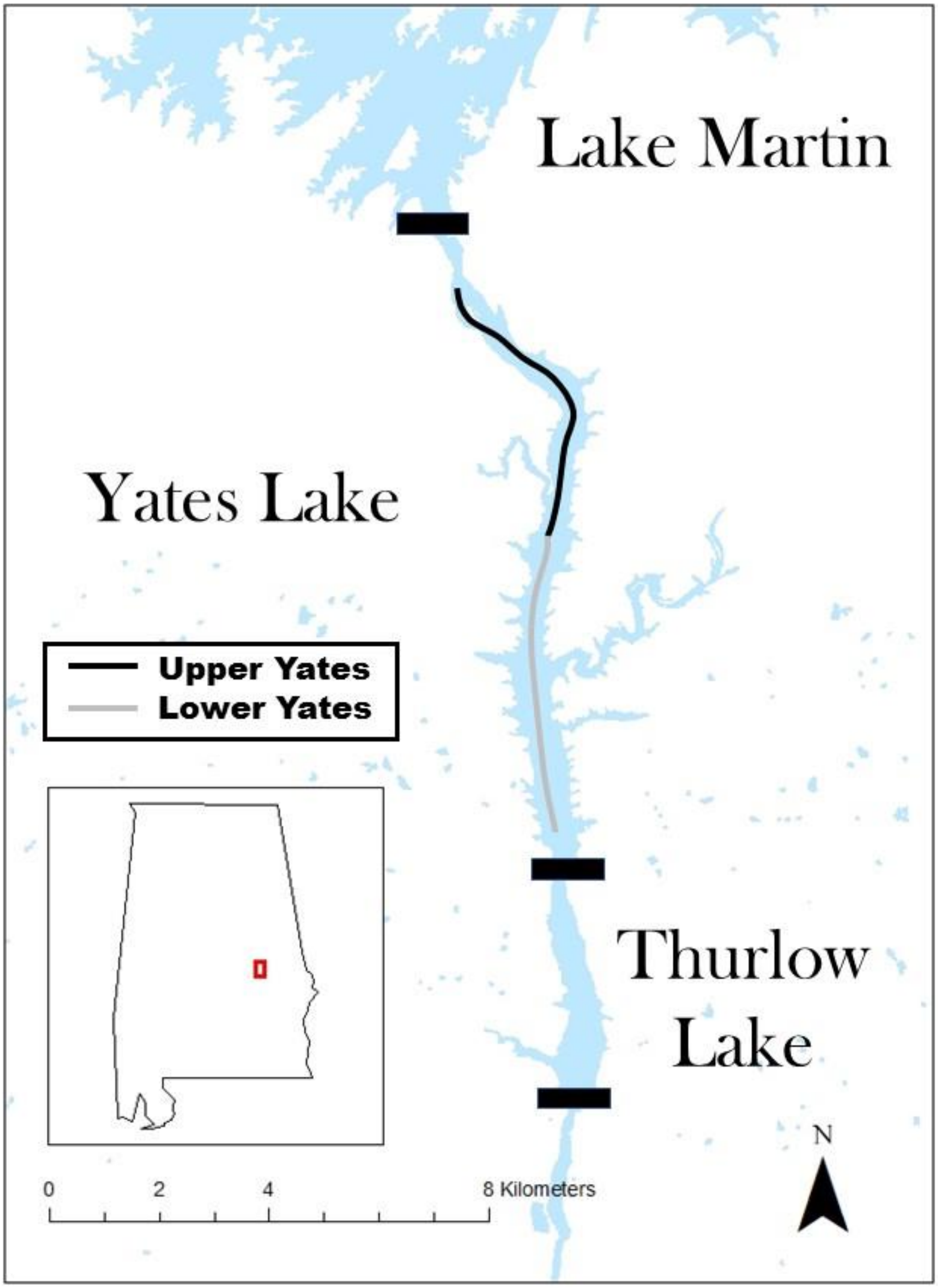


Figure 3. Location of Yates Lake with hydroacoustic transect segments indicated by colored lines.

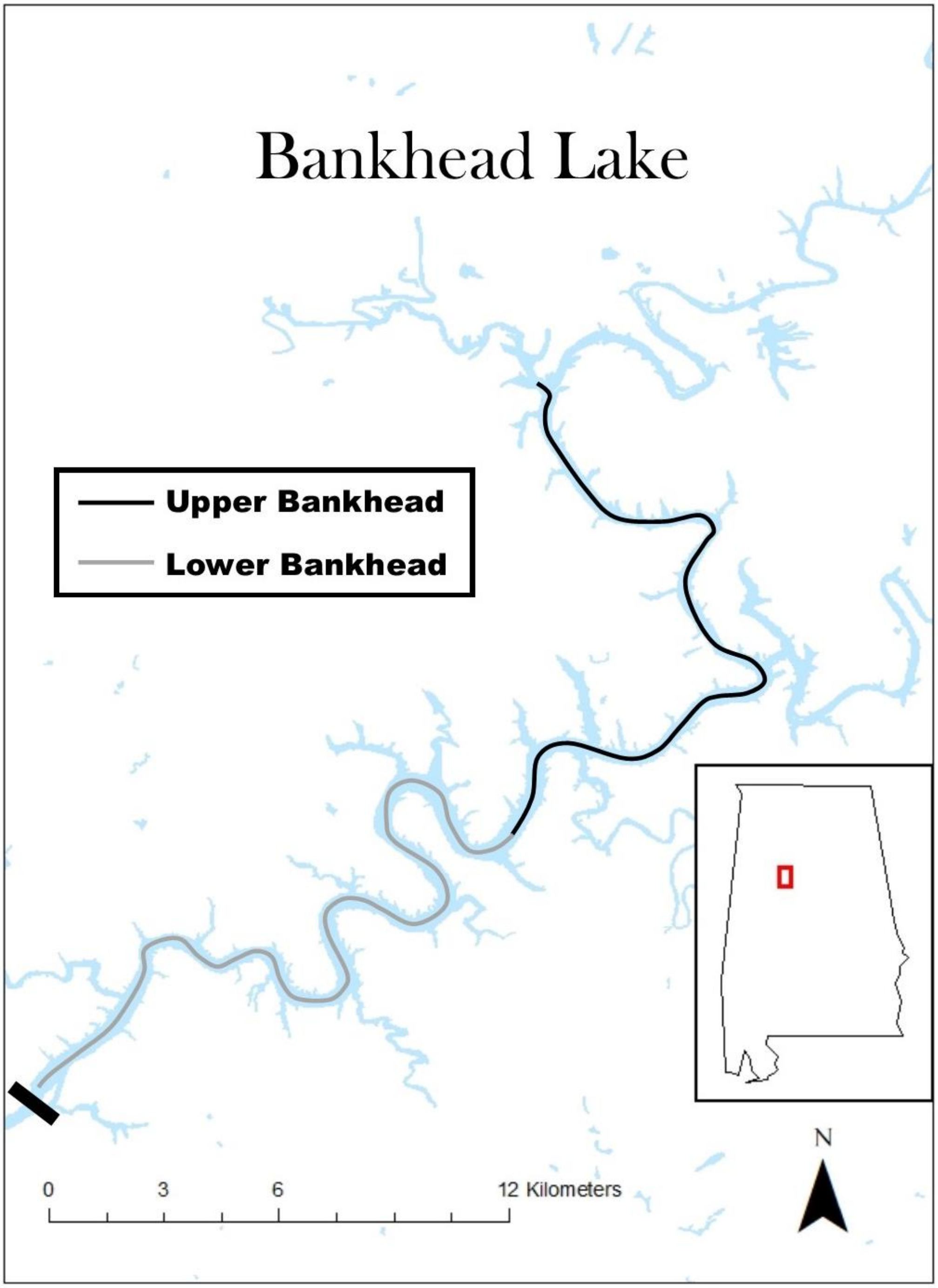


Figure 4. Location of Bankhead Lake, AL with hydroacoustic transect segments indicated by colored lines.

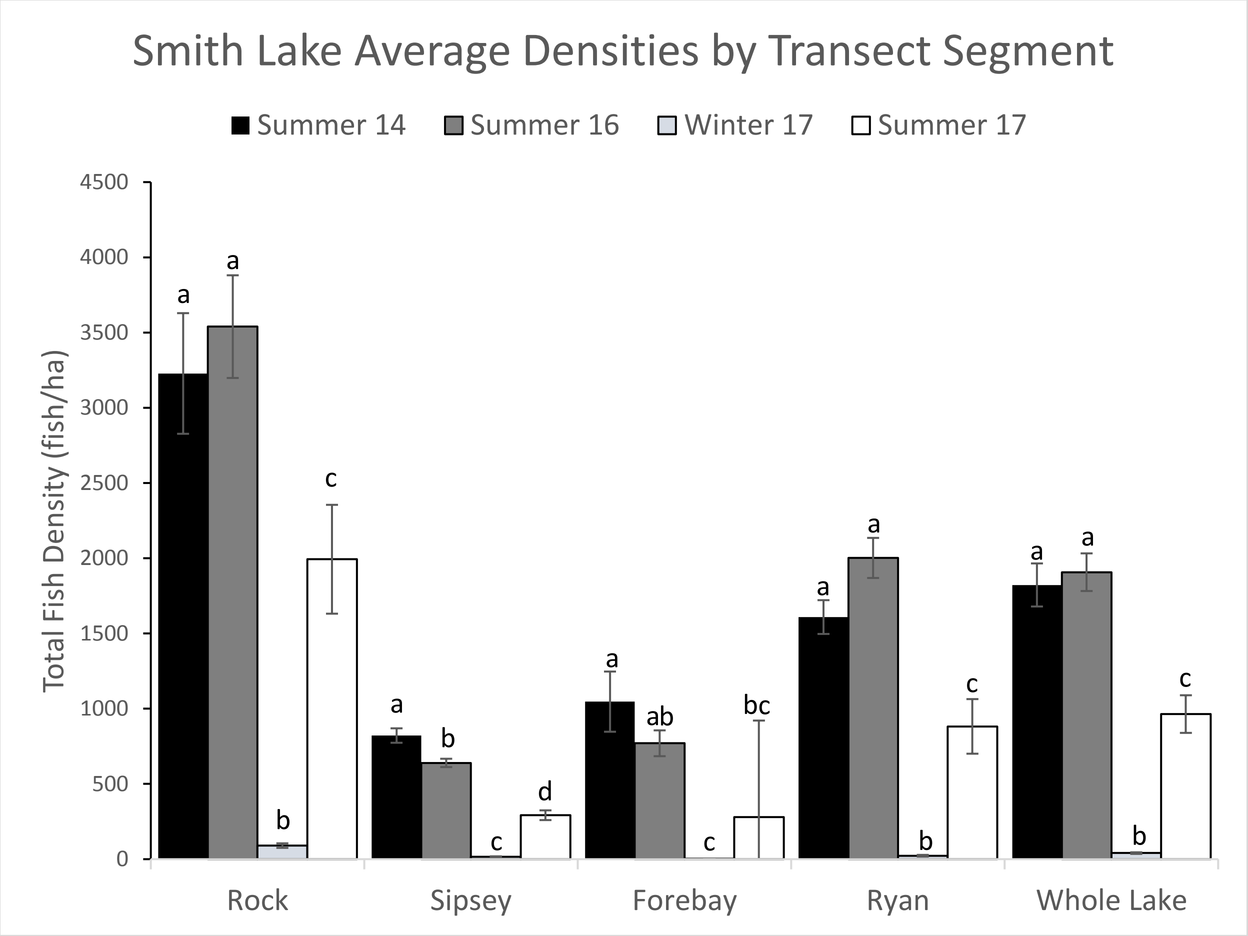


Figure 5. Lewis Smith Lake total fish densities (mean ± SE) in the Rock, Sipsey, Ryan, Forebay, and Whole Lake transect segments. Bars with different letters represent significant differences between segment surveys.

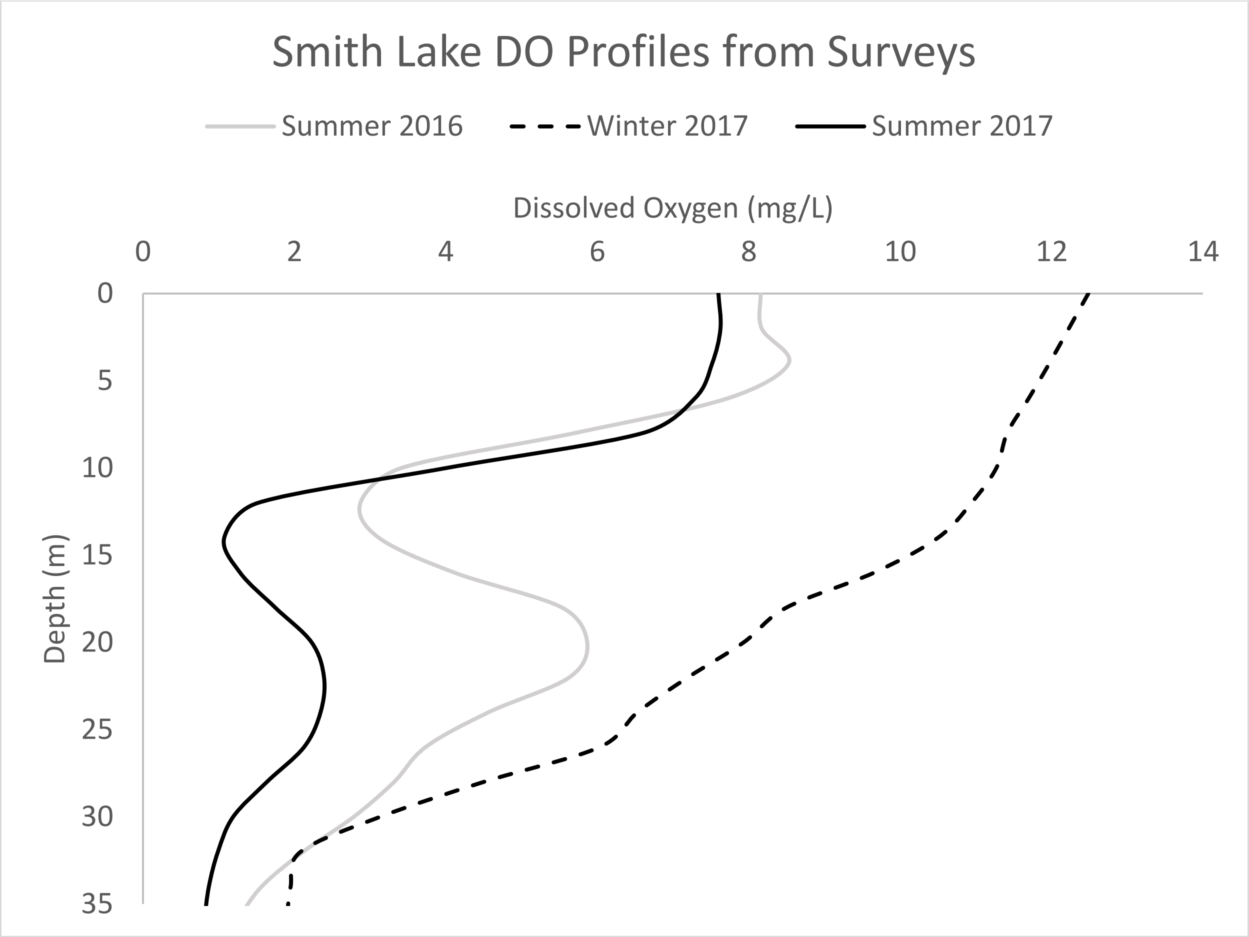


Figure 6. Lewis Smith Lake averaged dissolved oxygen profiles across sites for the summer 2016, winter 2017, and summer 2017 surveys.

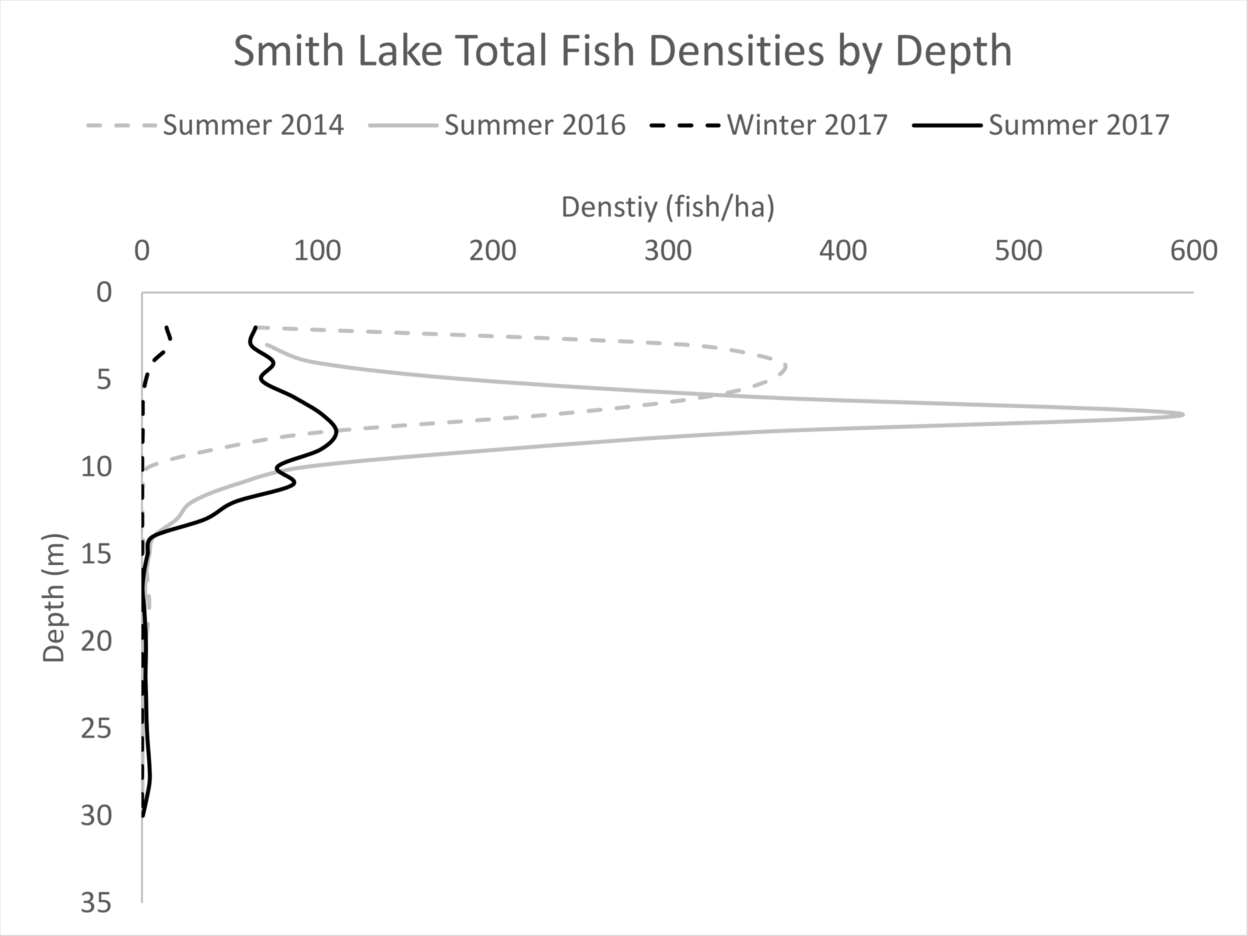


Figure 7. Lewis Smith Lake whole lake total fish densities across depths for the summer 2014, summer 2016, winter 2017, and summer 2017 surveys.

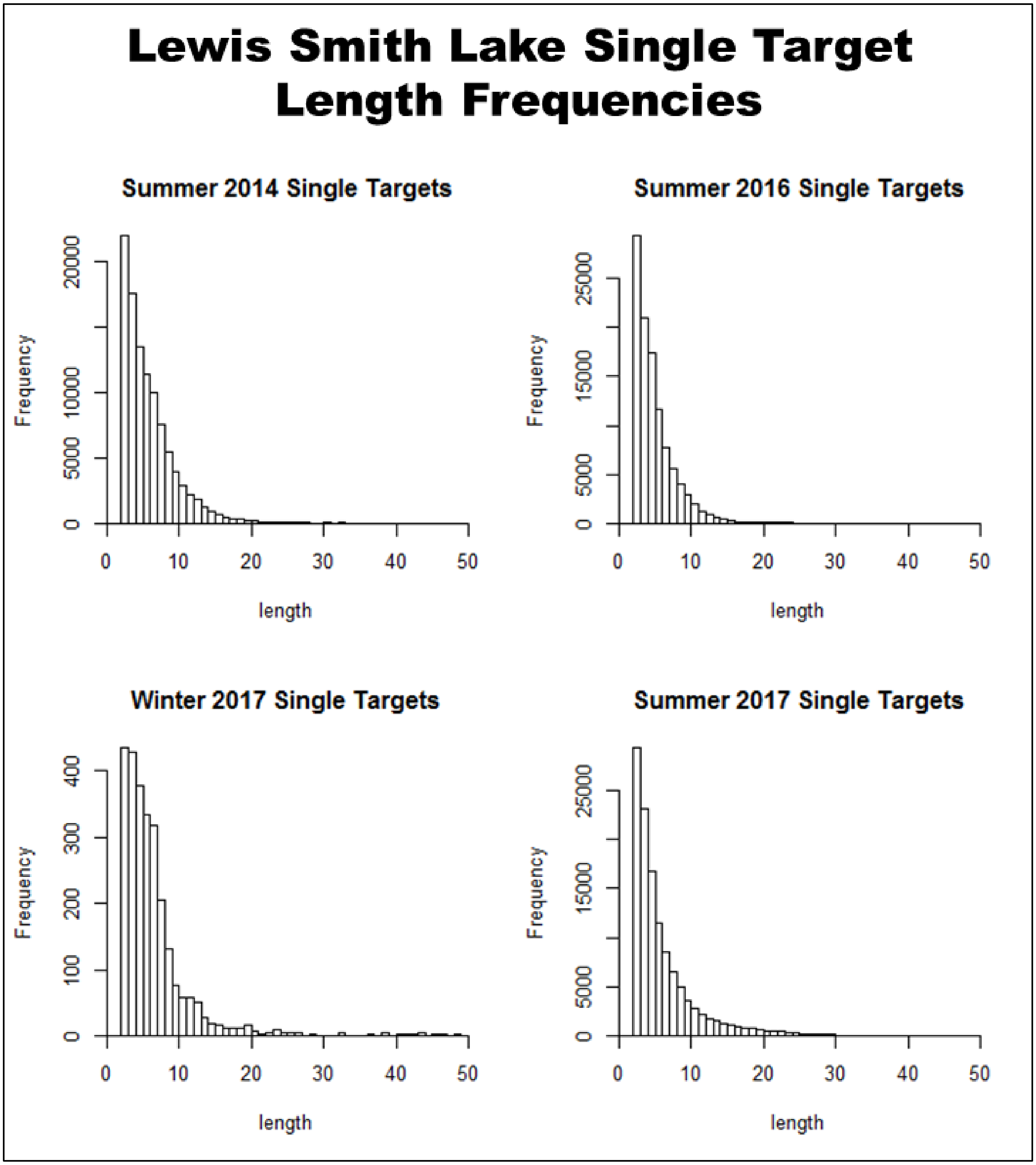


Figure 8. Lewis Smith Lake single target length-frequency histograms for summer 2014, summer 2016, winter 2017, and summer 2017 surveys.

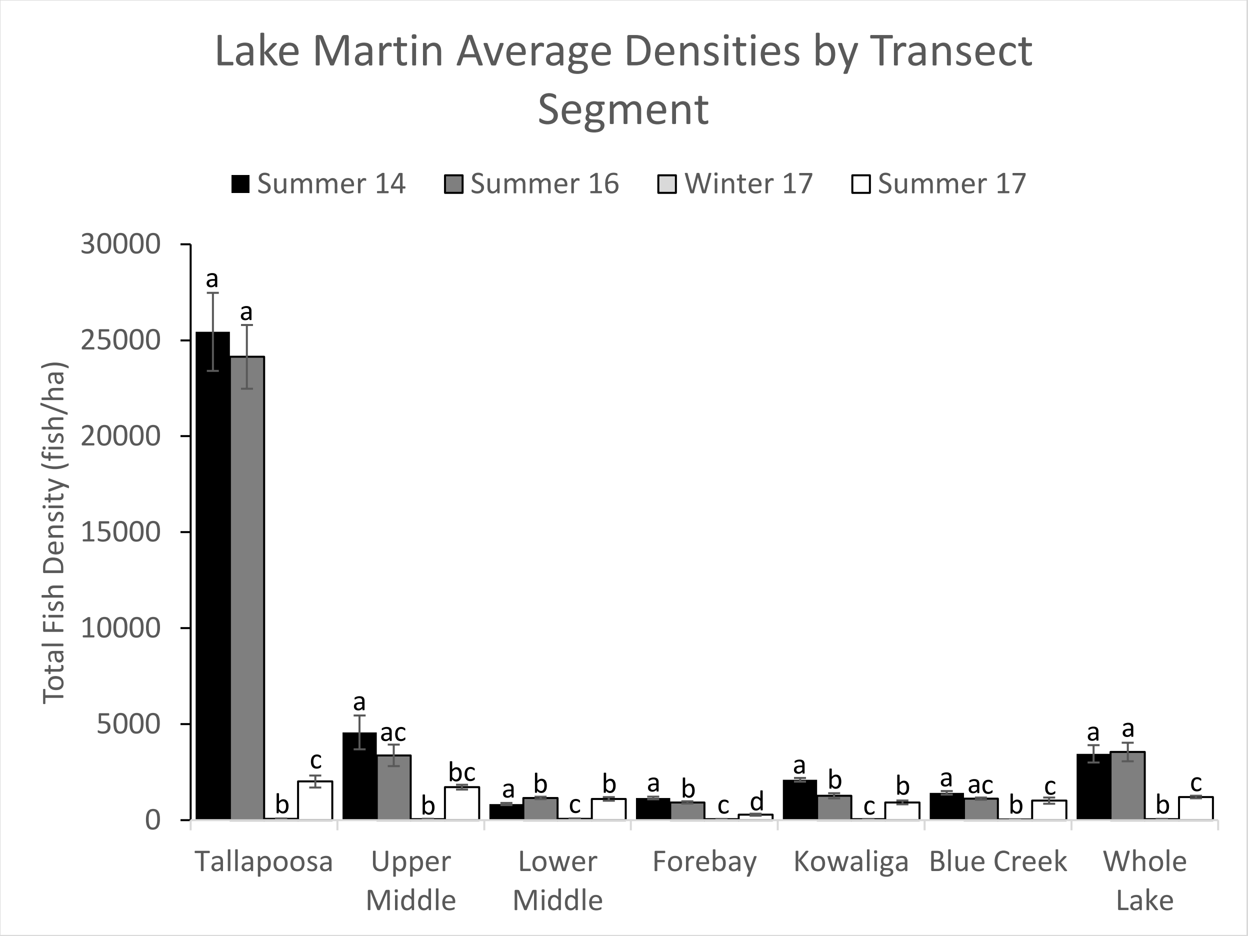


Figure 9. Lake Martin total fish densities (mean ± SE) in the Tallapoosa, Upper Middle, Lower Middle, Forebay, Kowaliga, Blue Creek, and Whole Lake transect segments. Bars with different letters represent significant differences between segment surveys.

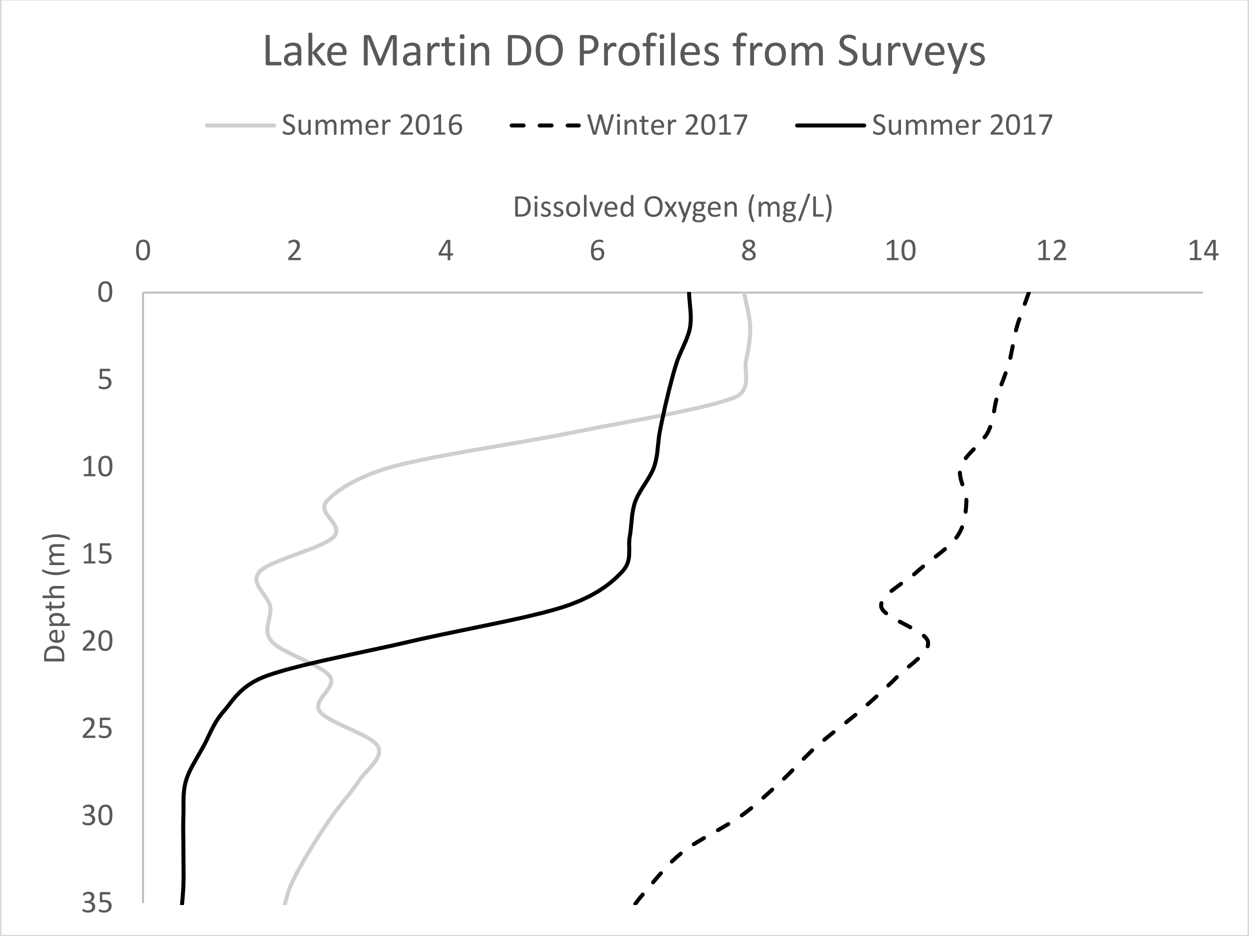


Figure 10. Lake Martin averaged dissolved oxygen profiles across sites for the summer 2016, winter 2017, and summer 2017 surveys.

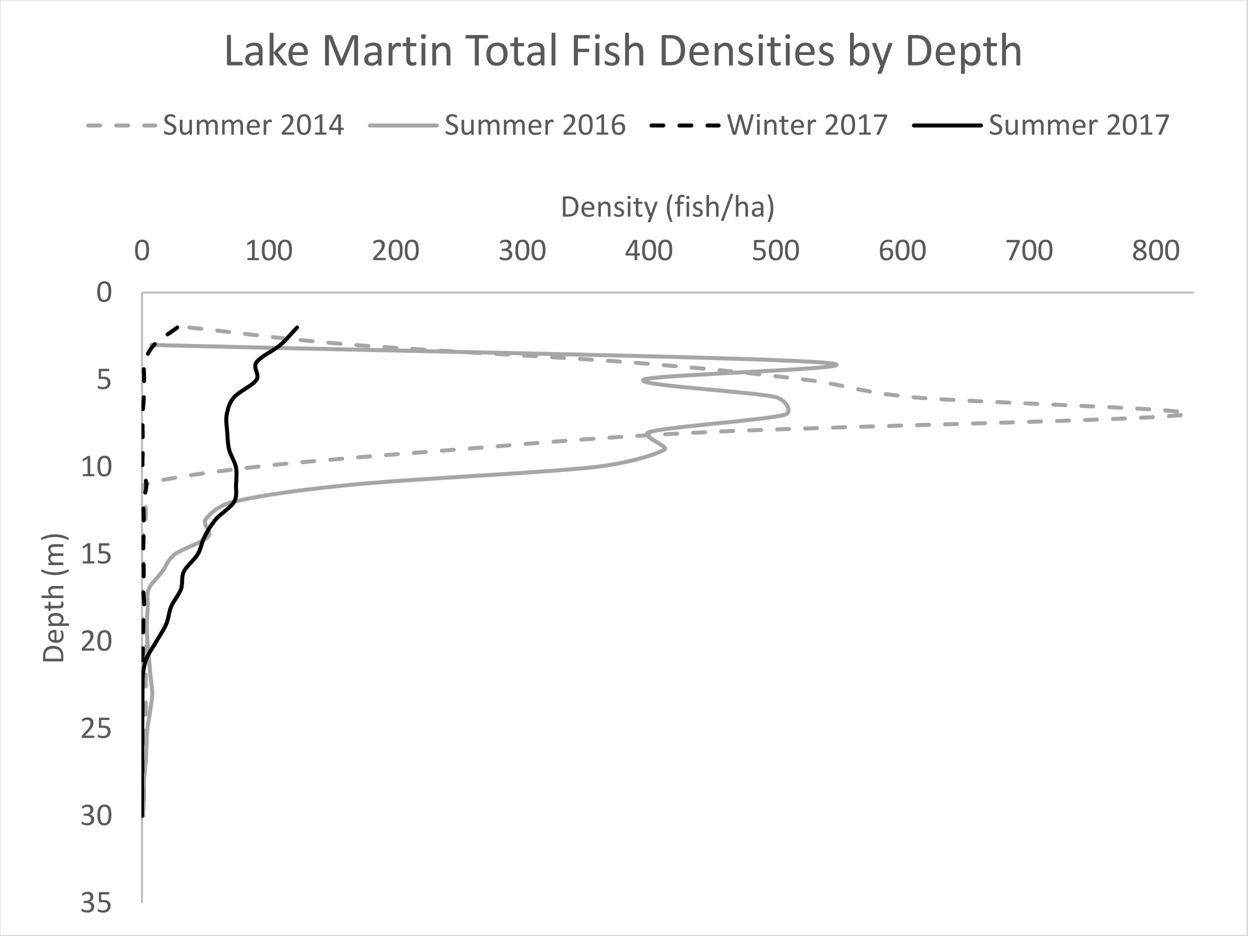


Figure 11. Lake Martin whole lake total fish densities across depths for the summer 2014, summer 2016, winter 2017, and summer 2017 surveys.

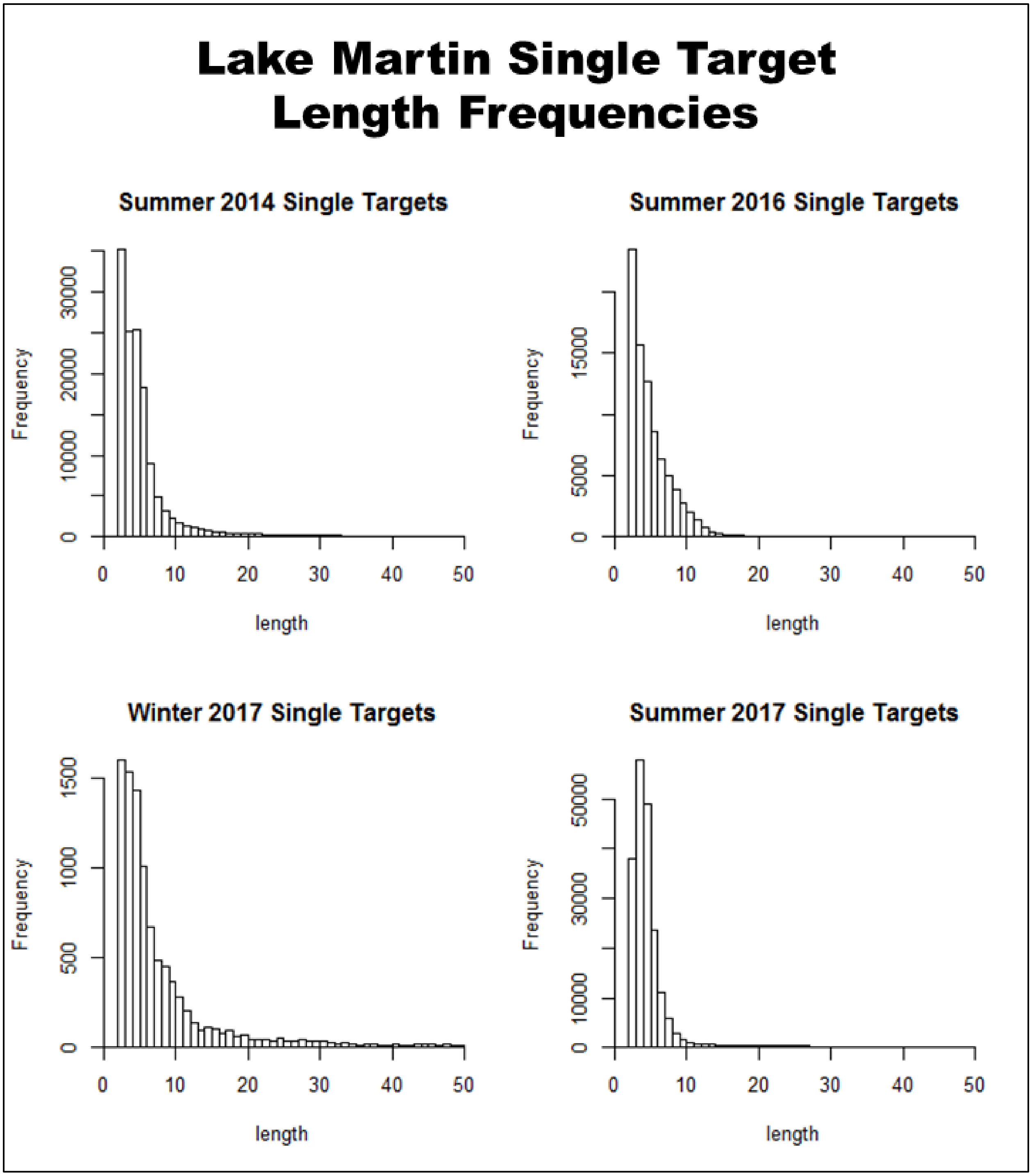


Figure 12. Lake Martin single target length frequency histograms for the summer 2014, summer 2016, winter 2017, and summer 2017 surveys.

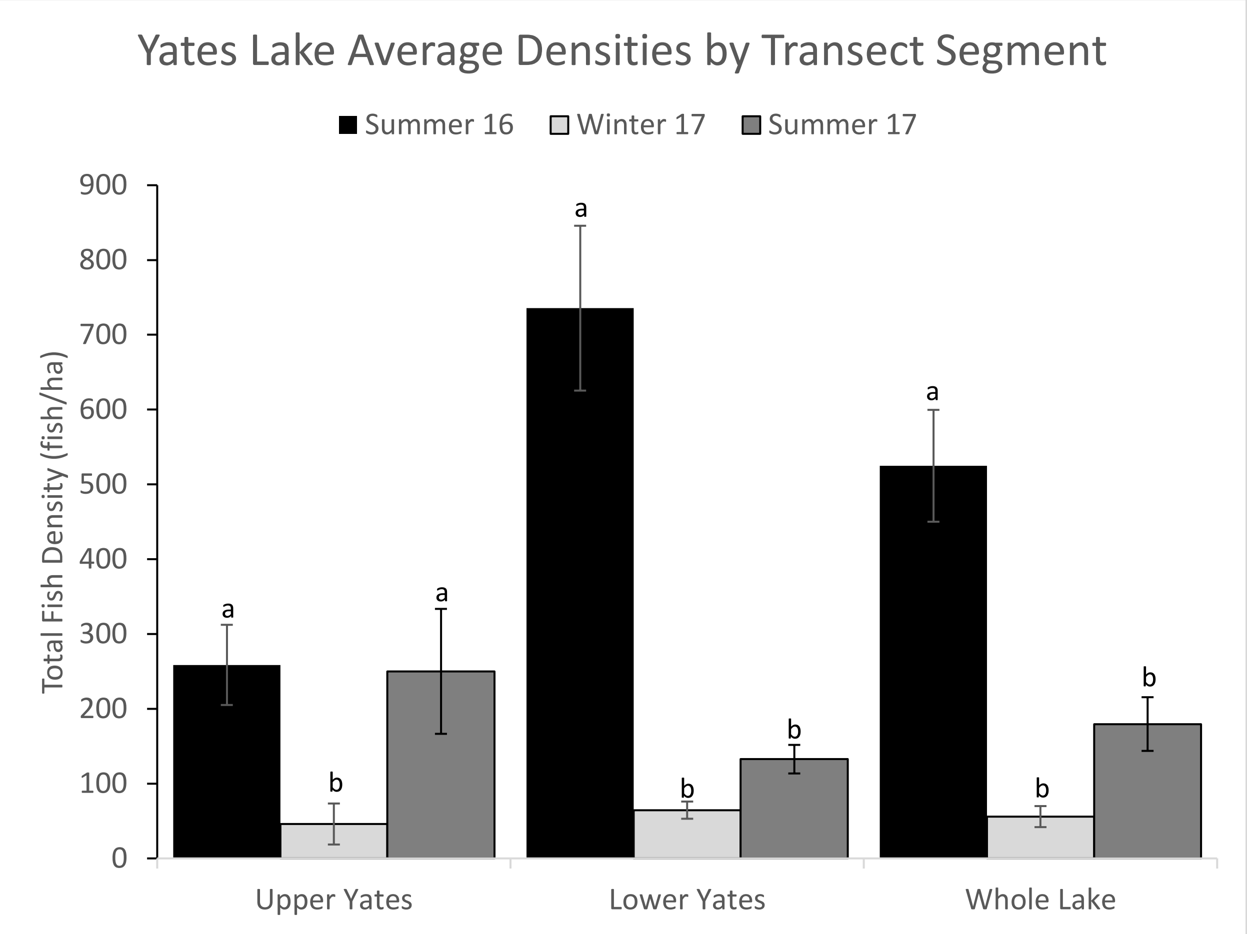


Figure 13. Yates Lake total fish densities (mean ± SE) in the Upper Channel, Lower Channel, and Whole Lake transect segments. Bars with different letters represent significant differences between segment surveys.

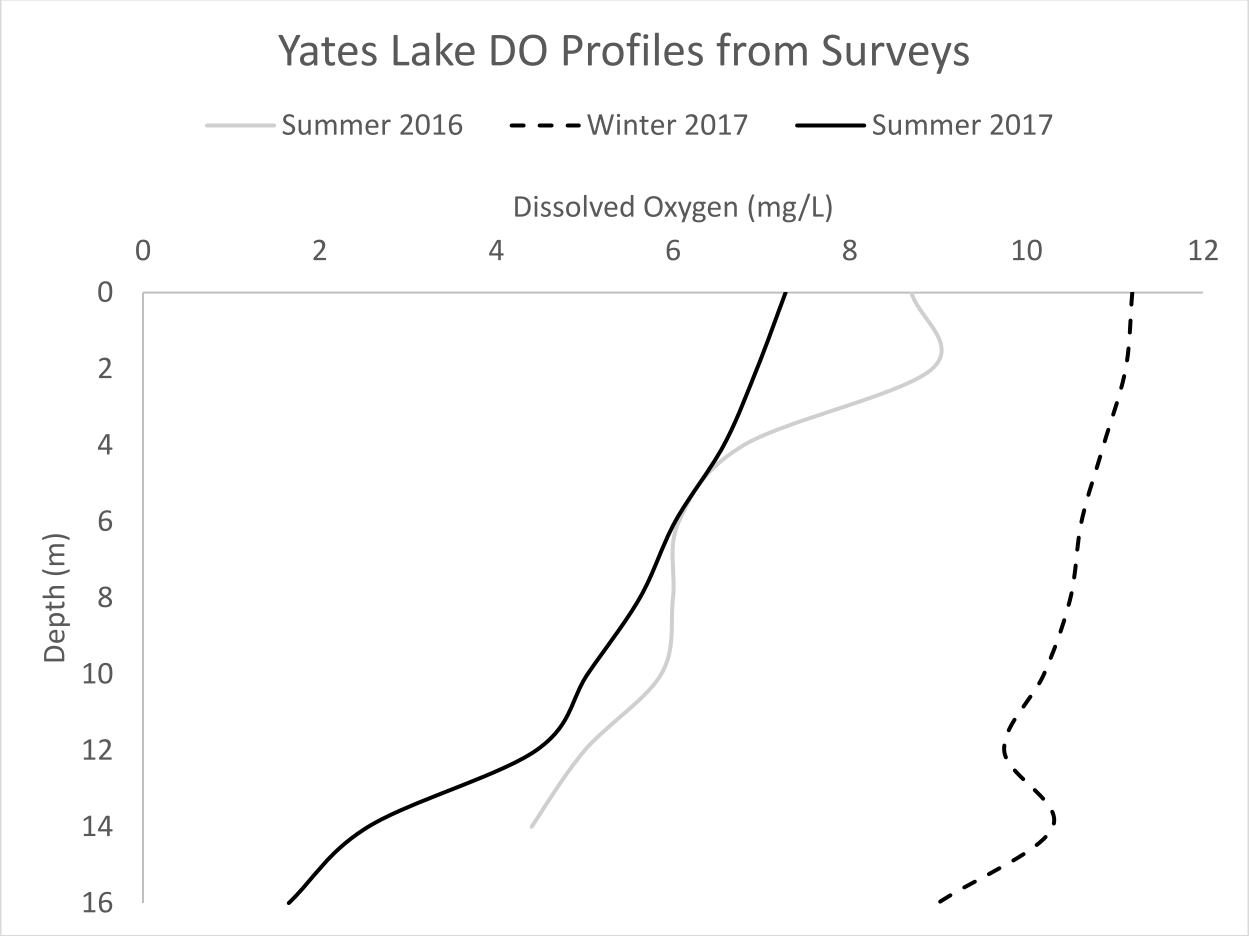


Figure 14. Yates Lake averaged dissolved oxygen profiles across sites for the summer 2016, winter 2017, and summer 2017 surveys.

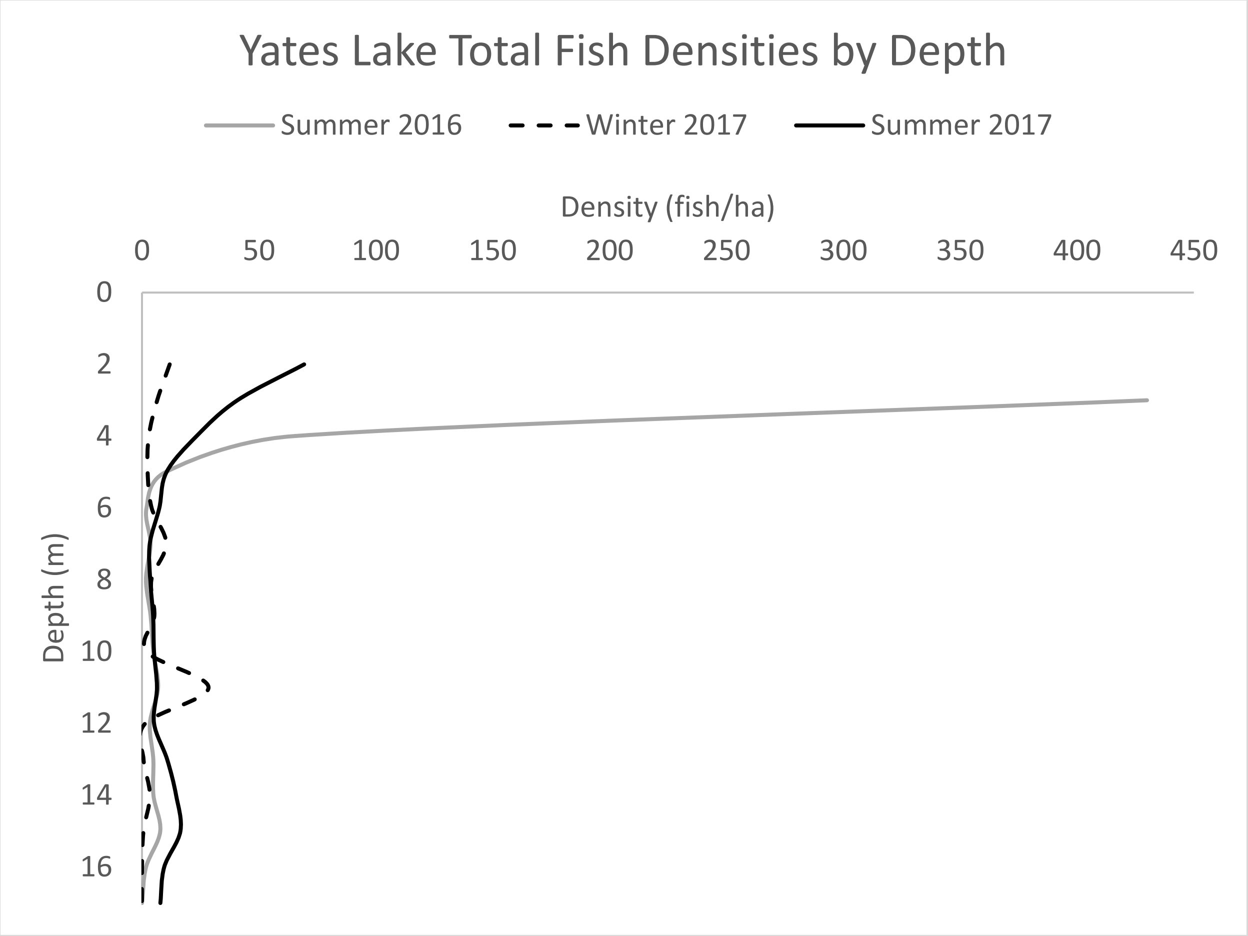


Figure 15. Yates Lake whole lake total fish densities across depths for the summer 2016, winter 2017, and summer 2017 surveys.

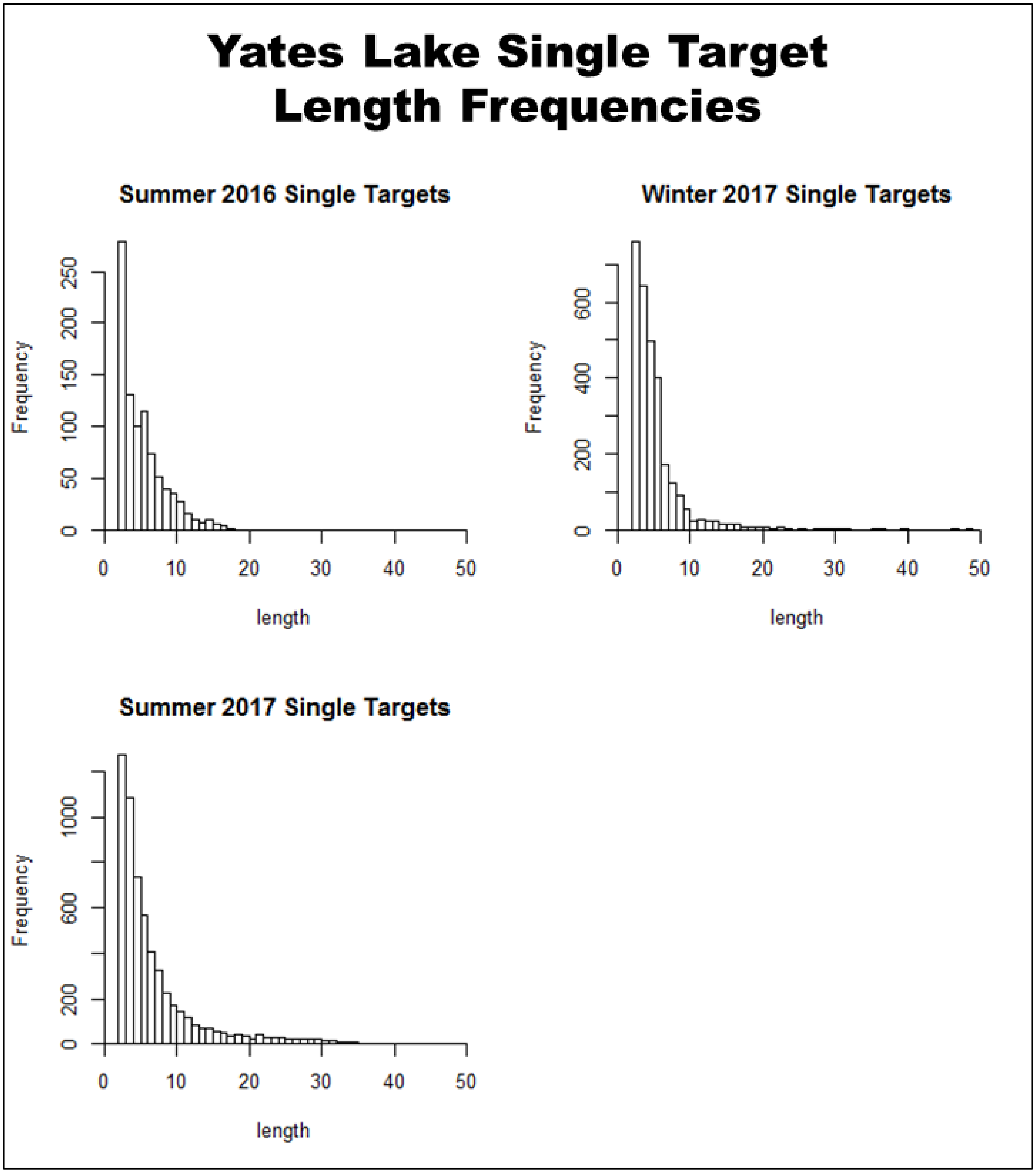


Figure 16. Yates Lake single target length-frequency histograms for the summer 2016, winter 2017, and summer 2017 surveys.

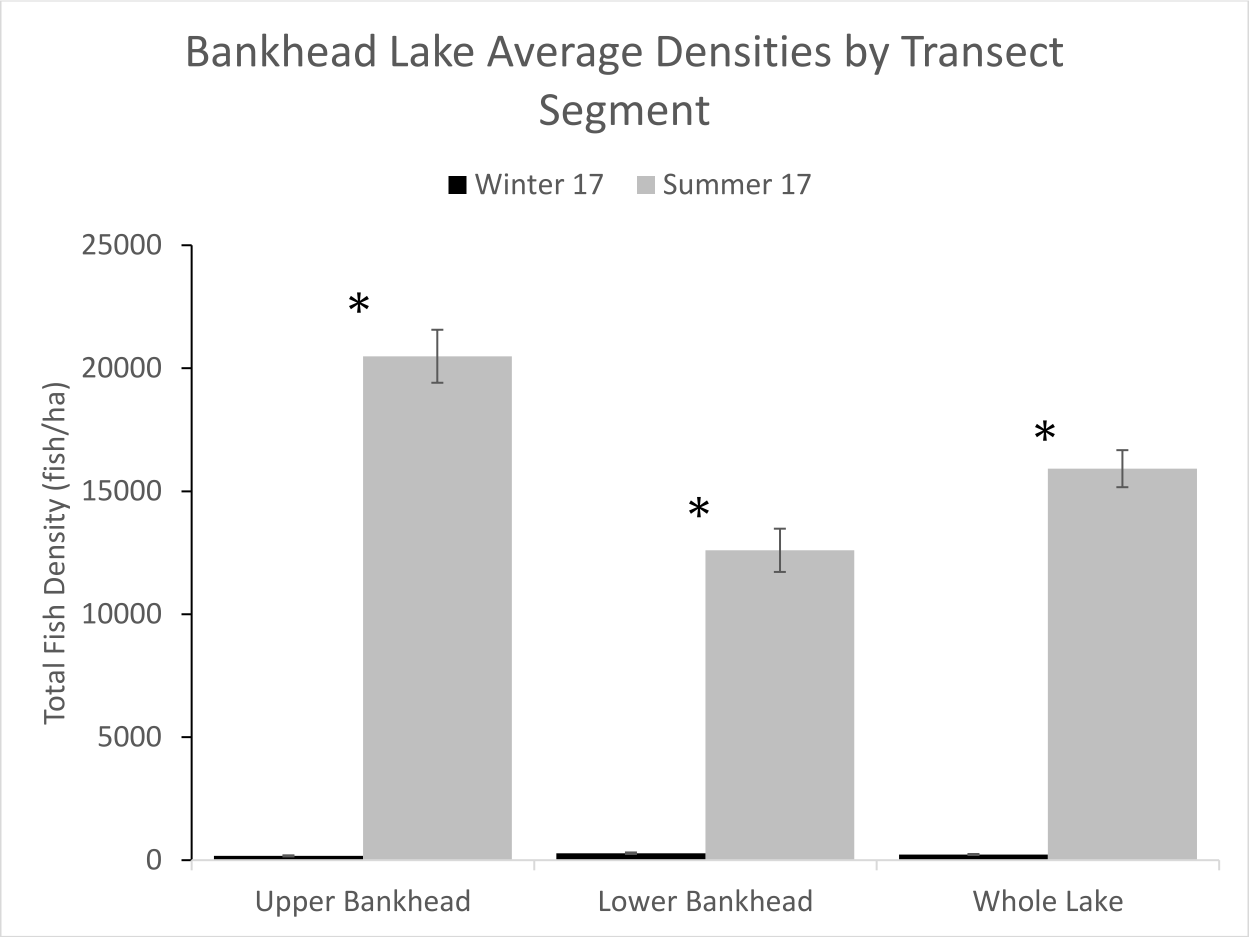


Figure 17. Bankhead Lake total fish densities (mean ± SE) for the Upper Channel, Lower Channel, and Whole Lake. Asterisks represent significant differences between segment surveys.

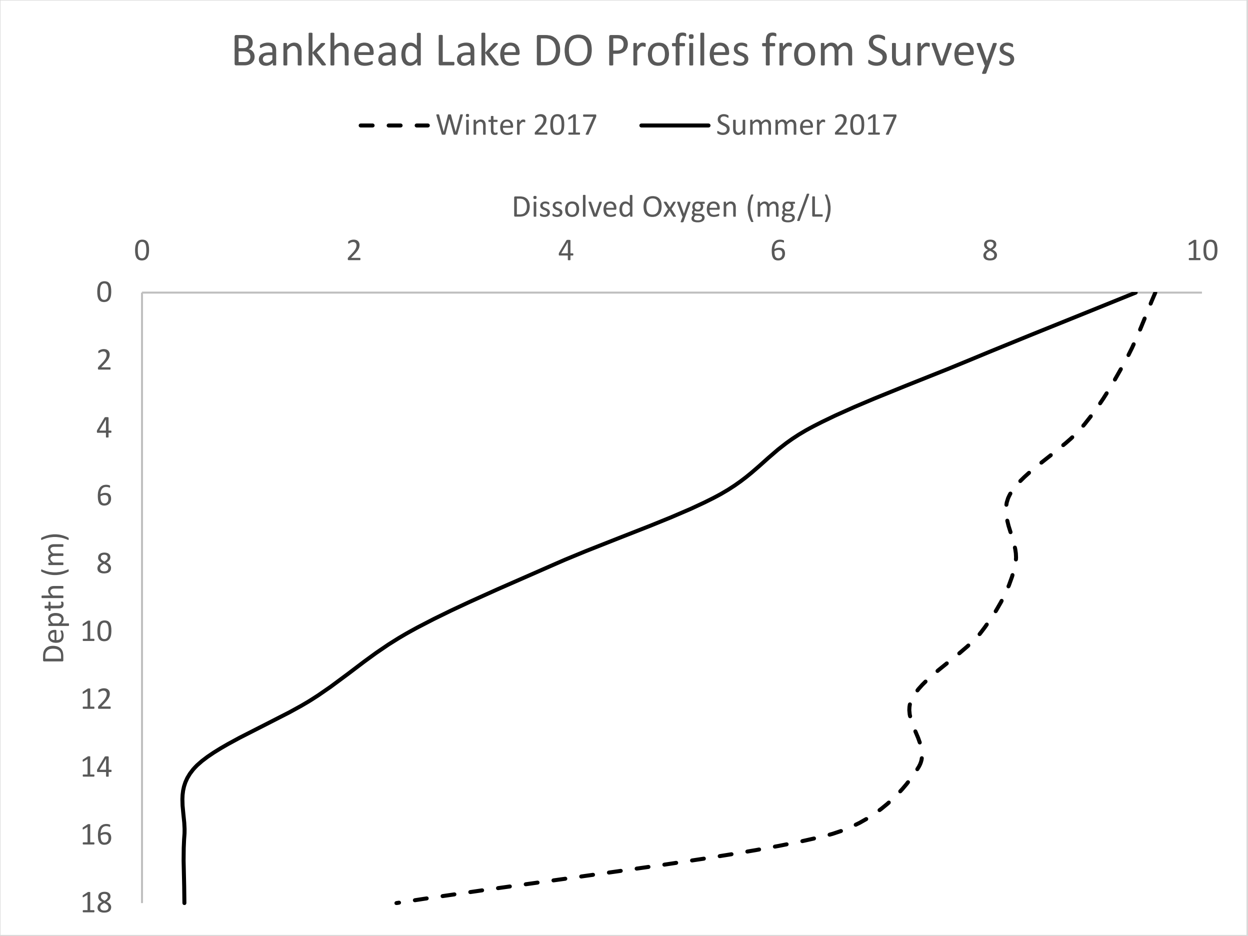


Figure 18. Bankhead Lake averaged dissolved oxygen profiles across sites for the winter 2017 and summer 2017 surveys.

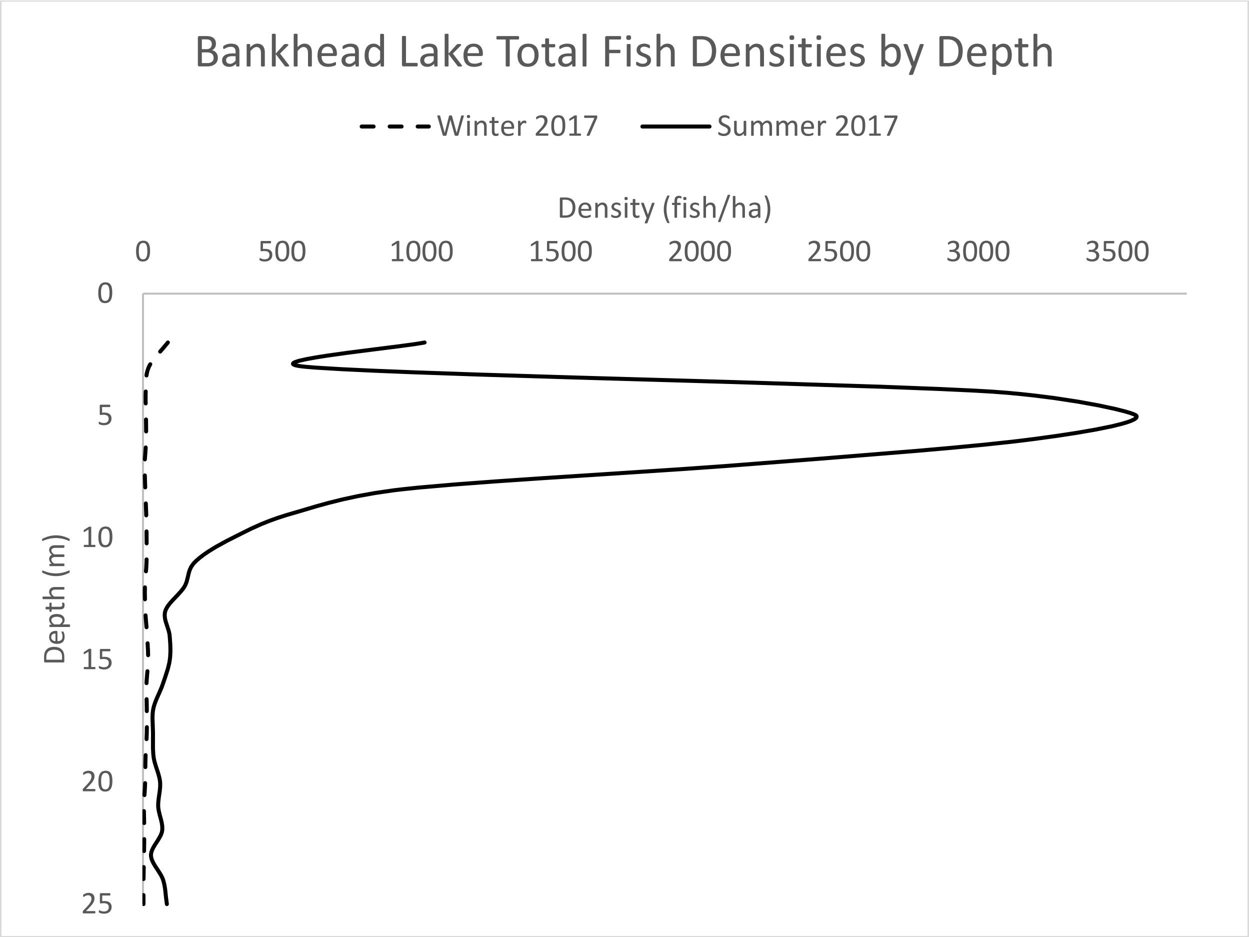


Figure 19. Bankhead Lake whole lake total fish densities by depth for the winter 2017 and summer 2017 surveys.

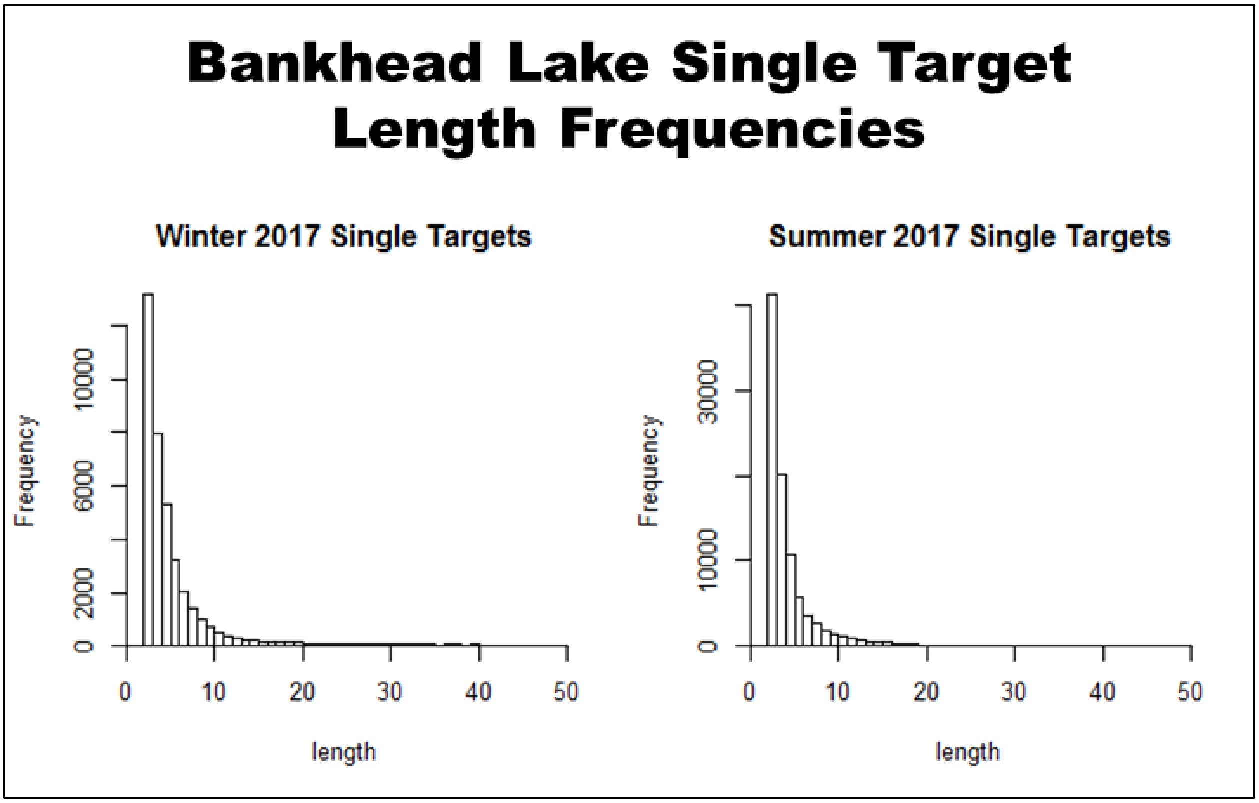


Figure 20. Bankhead Lake single target length frequency histograms for the winter 2017 and summer 2017 surveys.

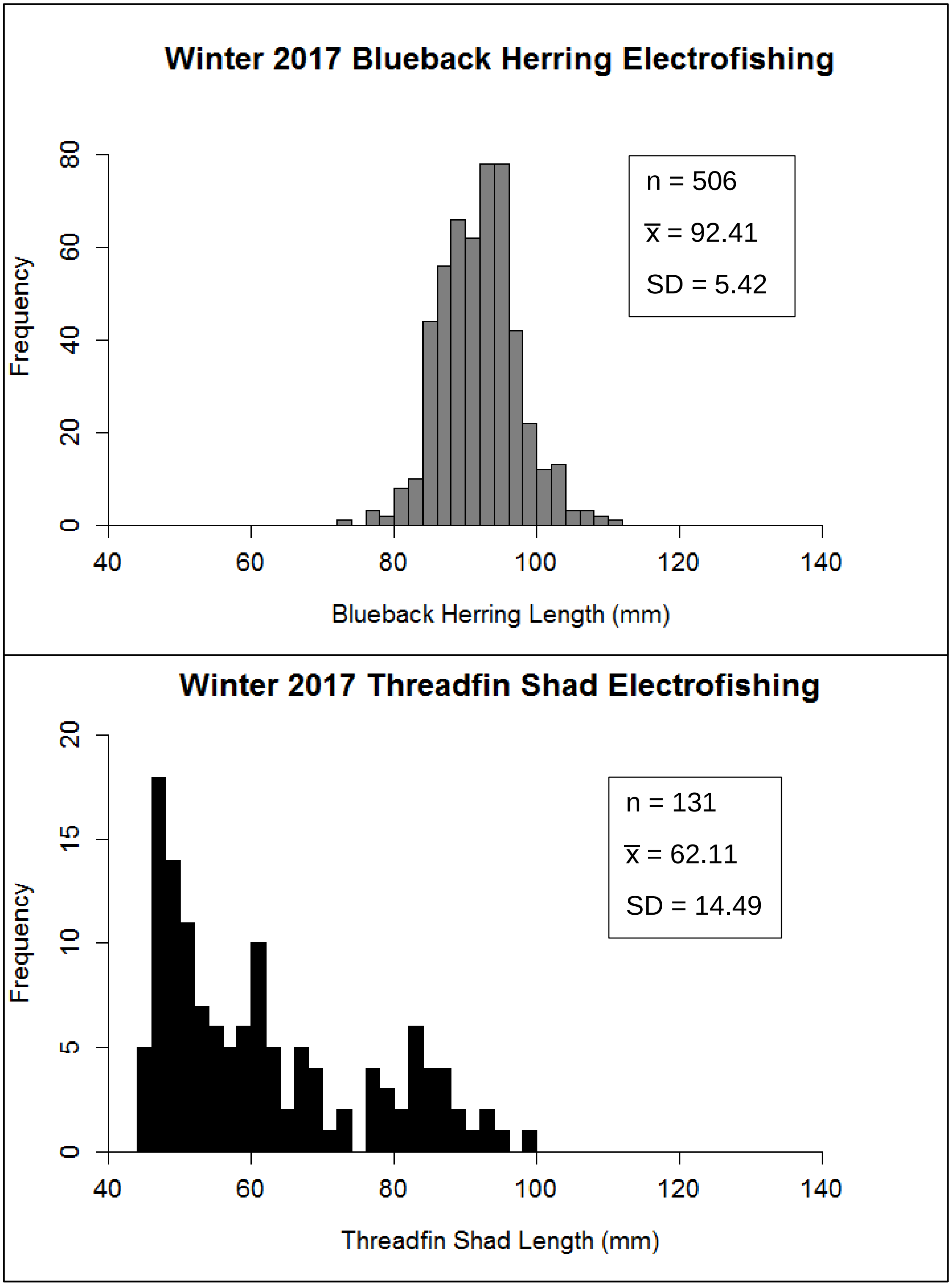


Figure 21. Lewis Smith Lake length-frequency histograms for fish sampled during pelagic electrofishing.

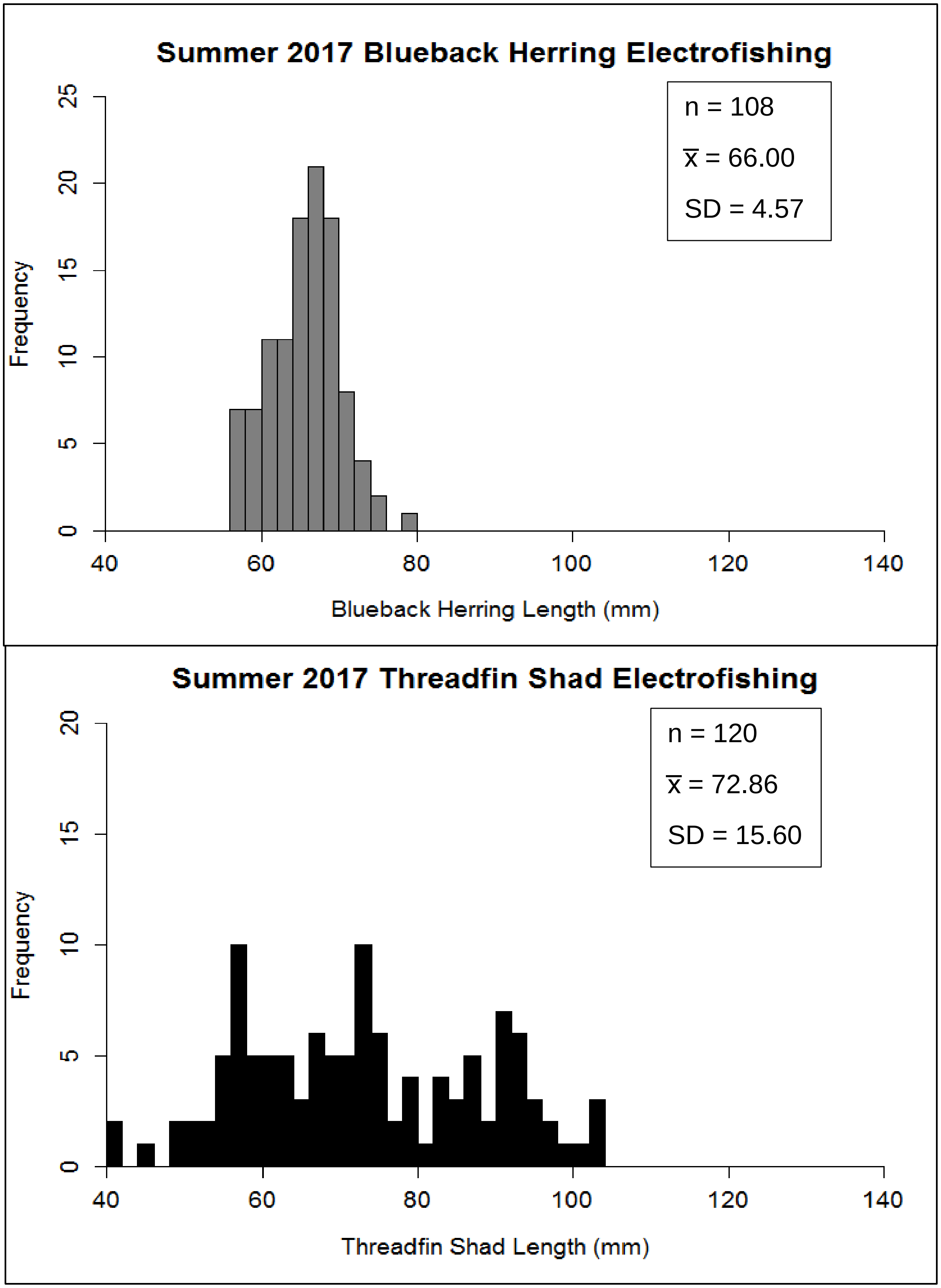


Figure 22. Lewis Smith Lake summer 2017 length-frequency histogram for Threadfin Shad and Blueback Herring collected with pelagic electrofishing.

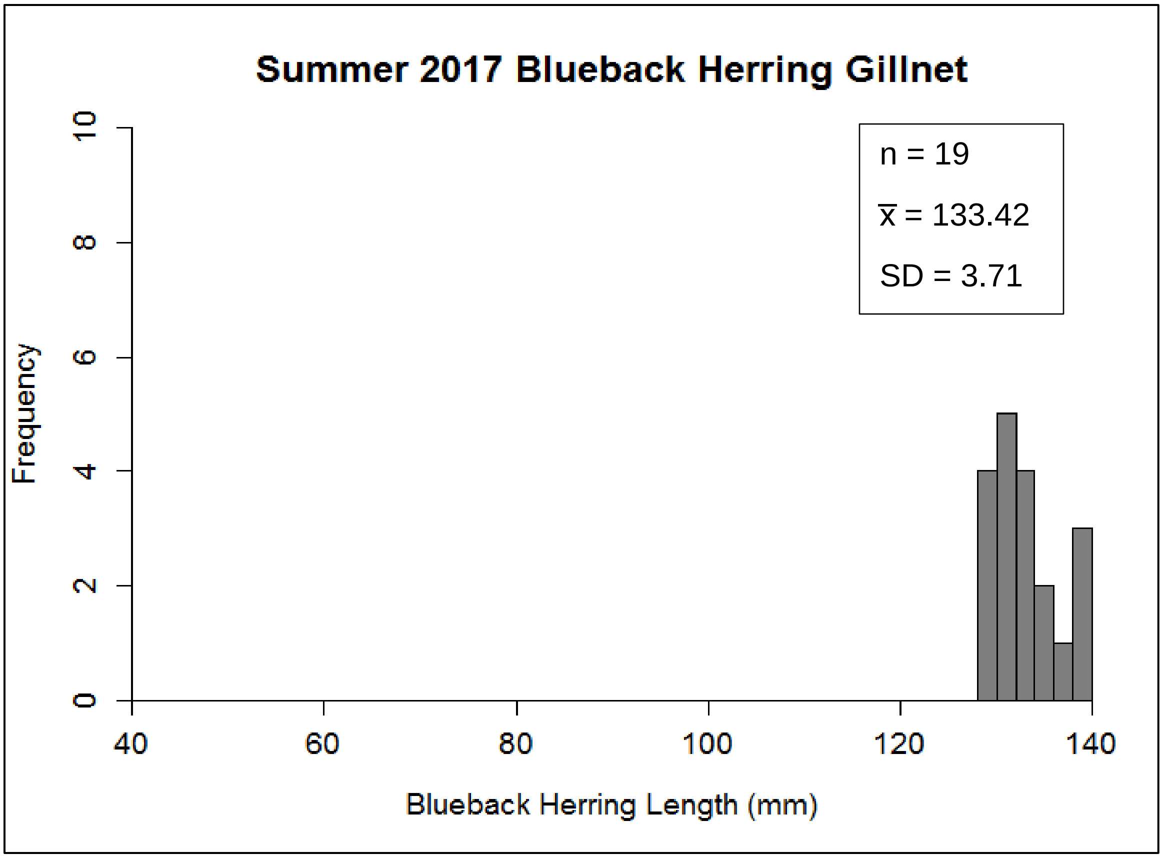


Figure 23. Lewis Smith Lake summer 2017 length-frequency histogram for Blueback Herring collected with gillnets.

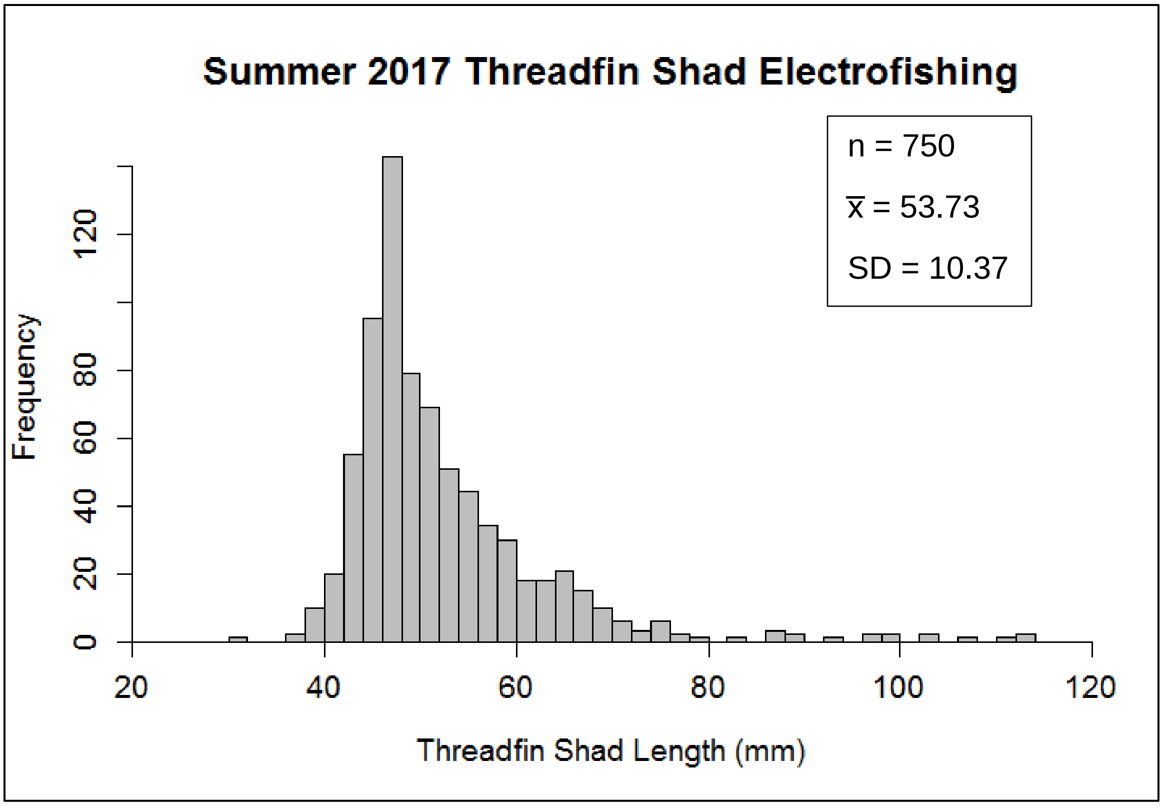


Figure 24. Lake Martin summer 2017 length-frequency histogram for Threadfin Shad collected with pelagic electrofishing.

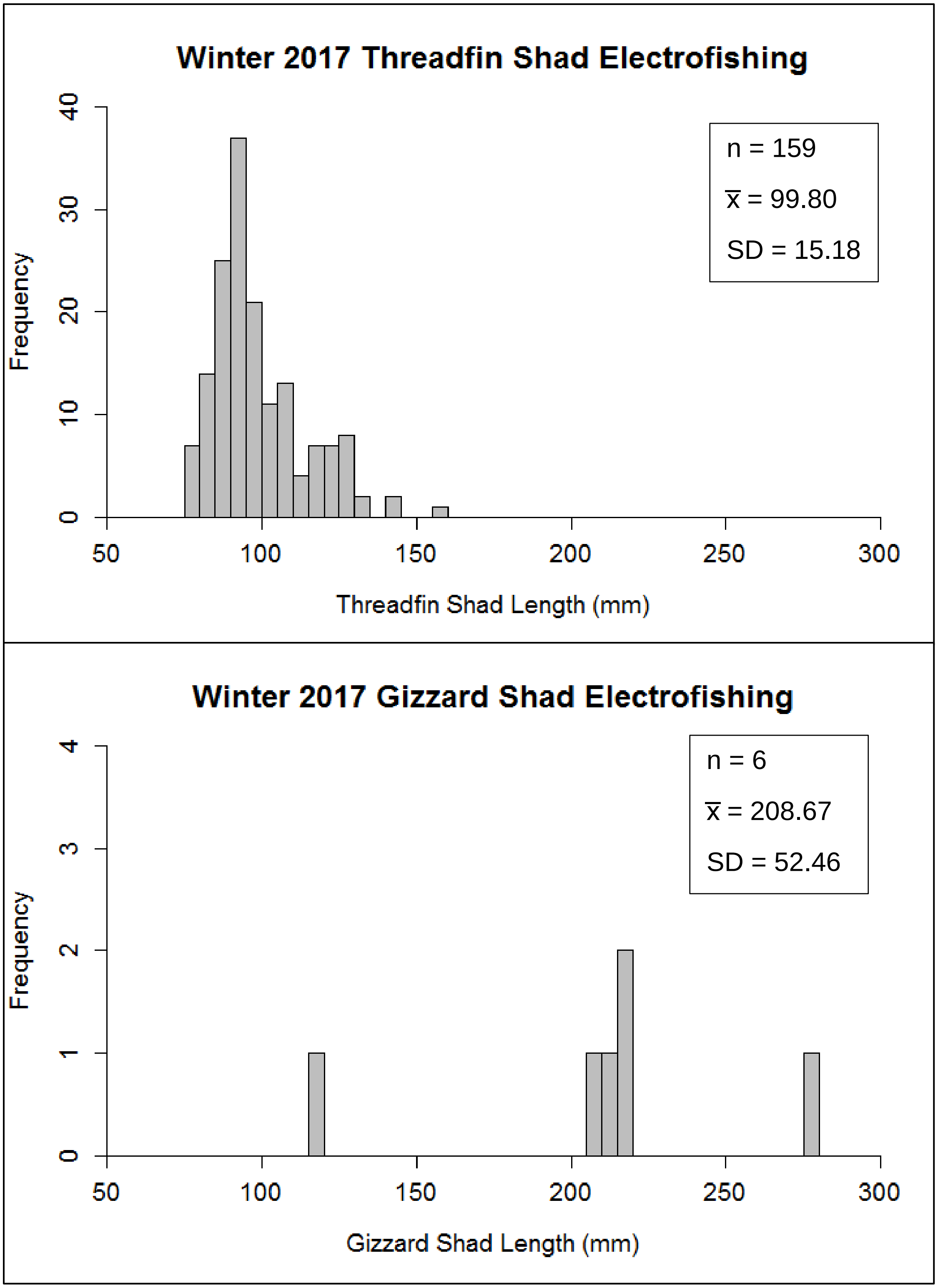


Figure 25. Bankhead Lake winter 2017 length-frequency histogram for Threadfin Shad and Gizzard Shad collected with pelagic electrofishing.

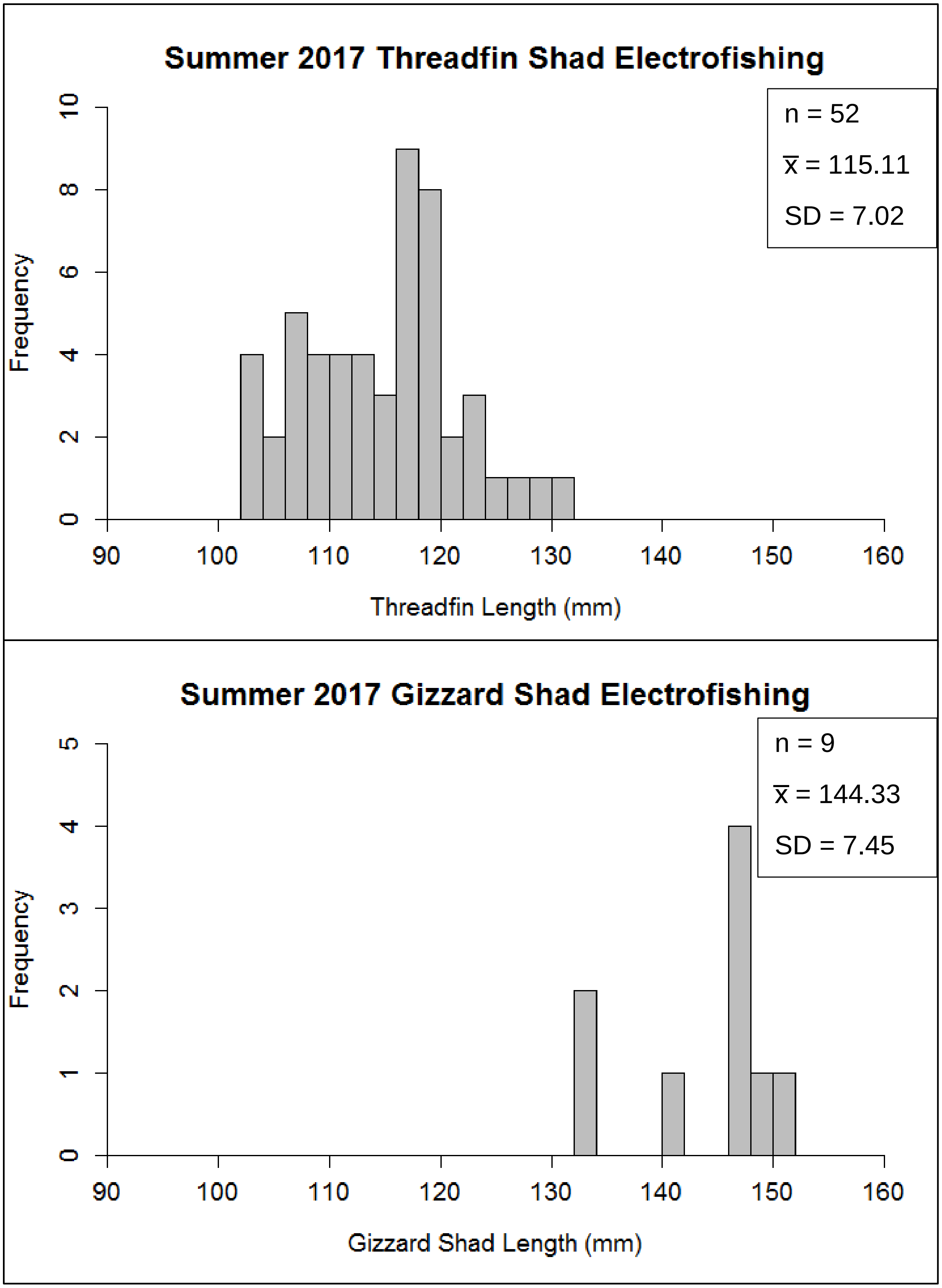


Figure 26. Bankhead Lake summer 2017 length-frequency histogram for Gizzard Shad collected with pelagic electrofishing.

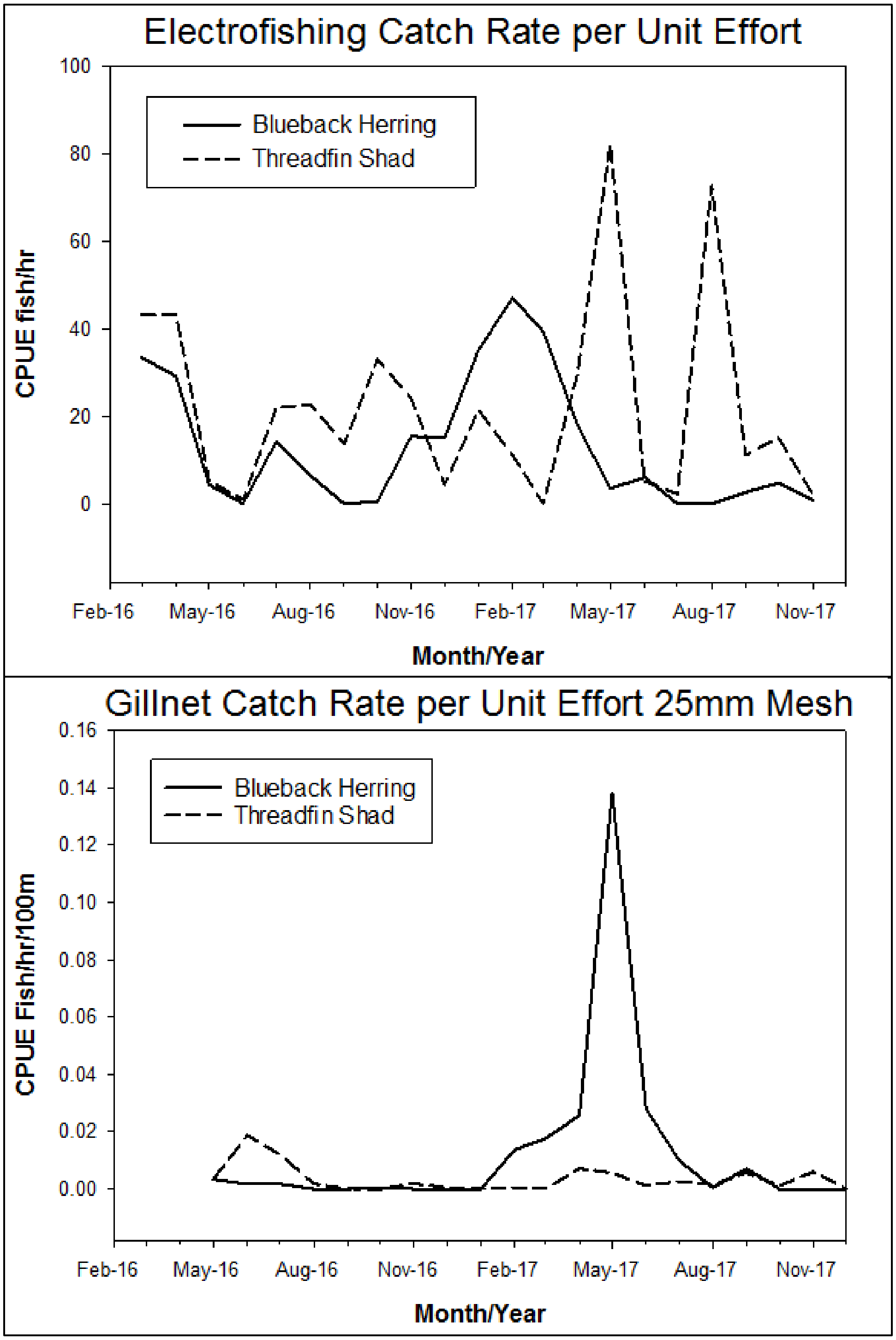


Figure 27. Electrofishing and gill net catch-per-unit-effort for clupeids during the standard sampling in Lewis Smith Lake, Alabama.

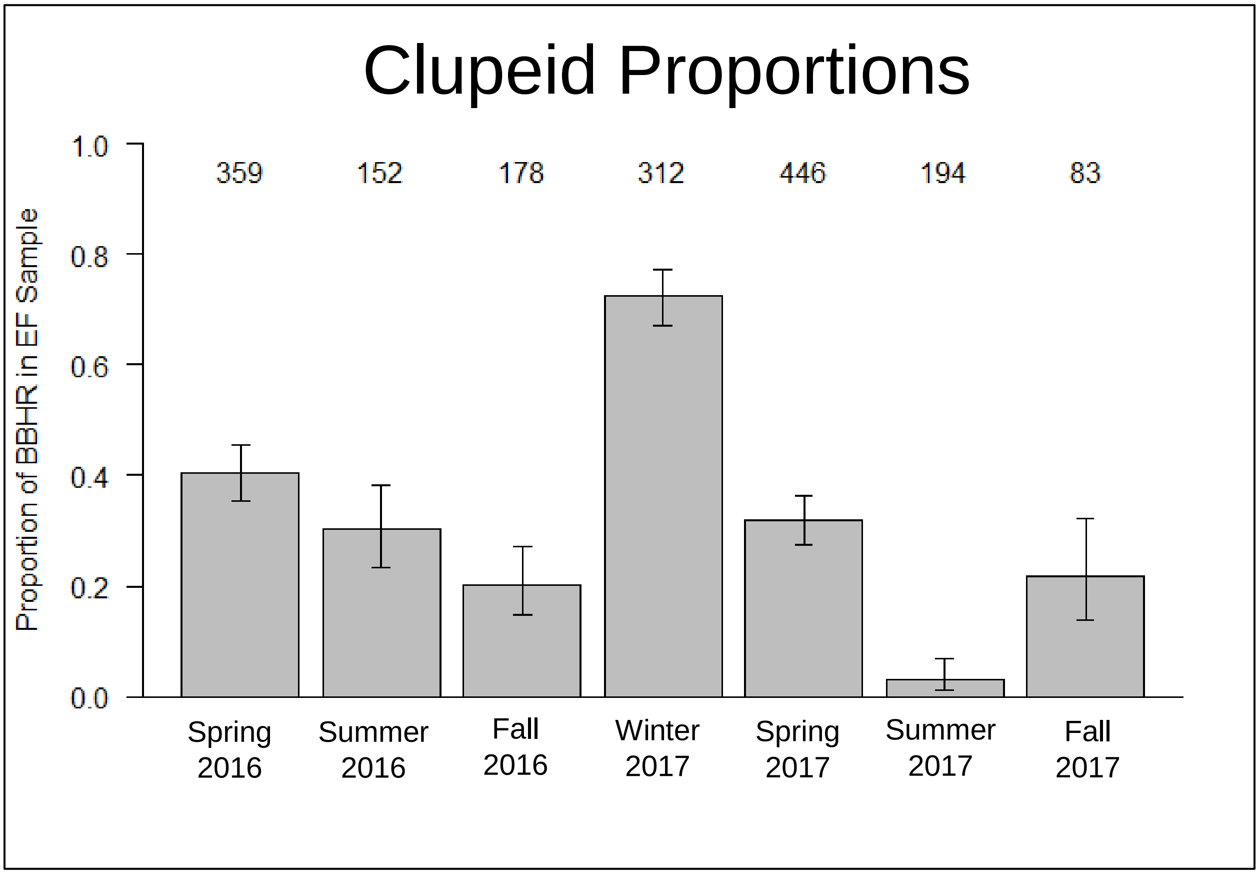


Figure 28. Proportion of electrofished clupeids in Lewis Smith Lake that consisted of Blueback Herring as a function of season and year. Numbers above bars represent combined sample size of Blueback Herring and Threadfin Shad. Error bars represent 95% confidence intervals of Blueback Herring proportions.

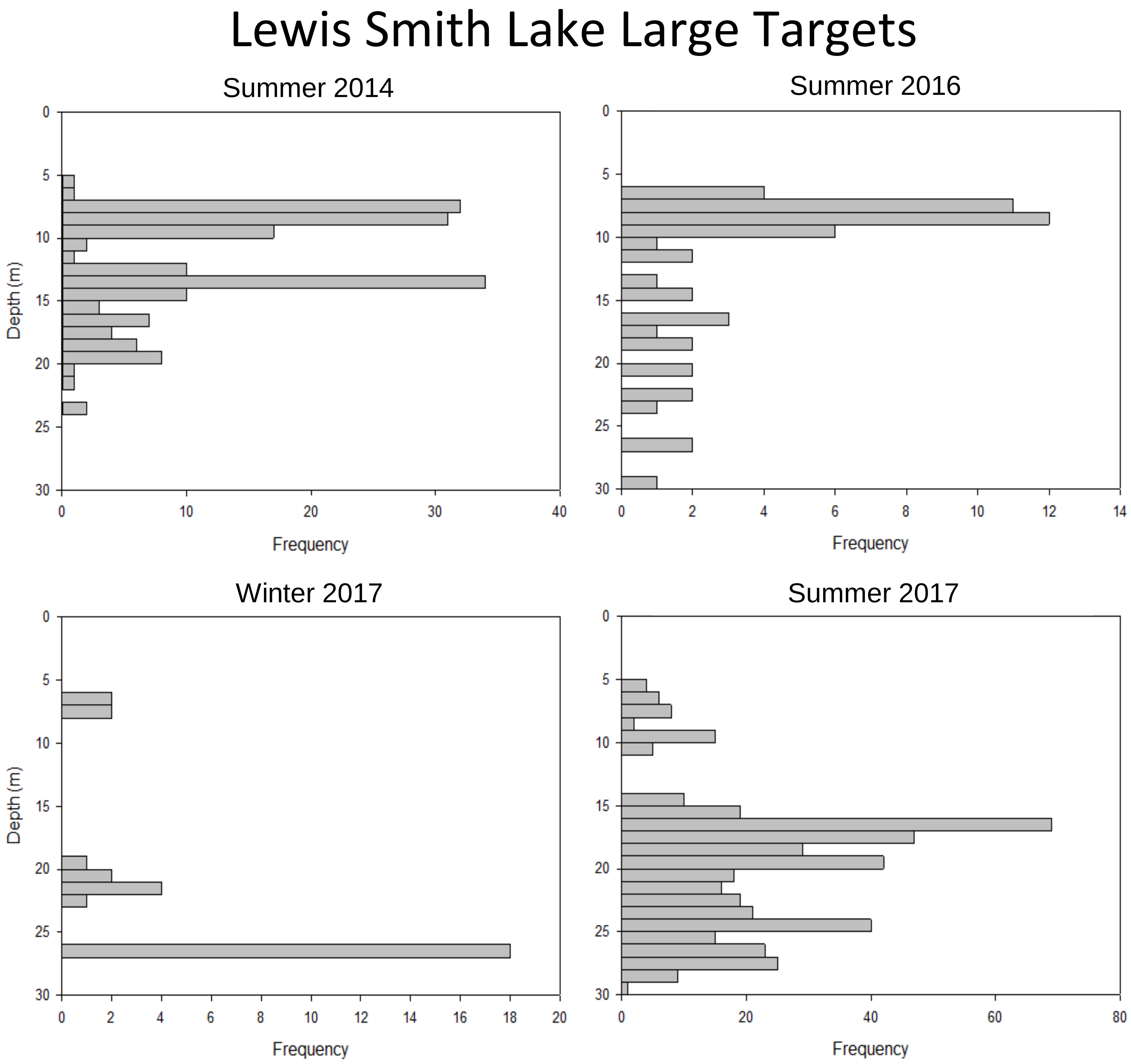
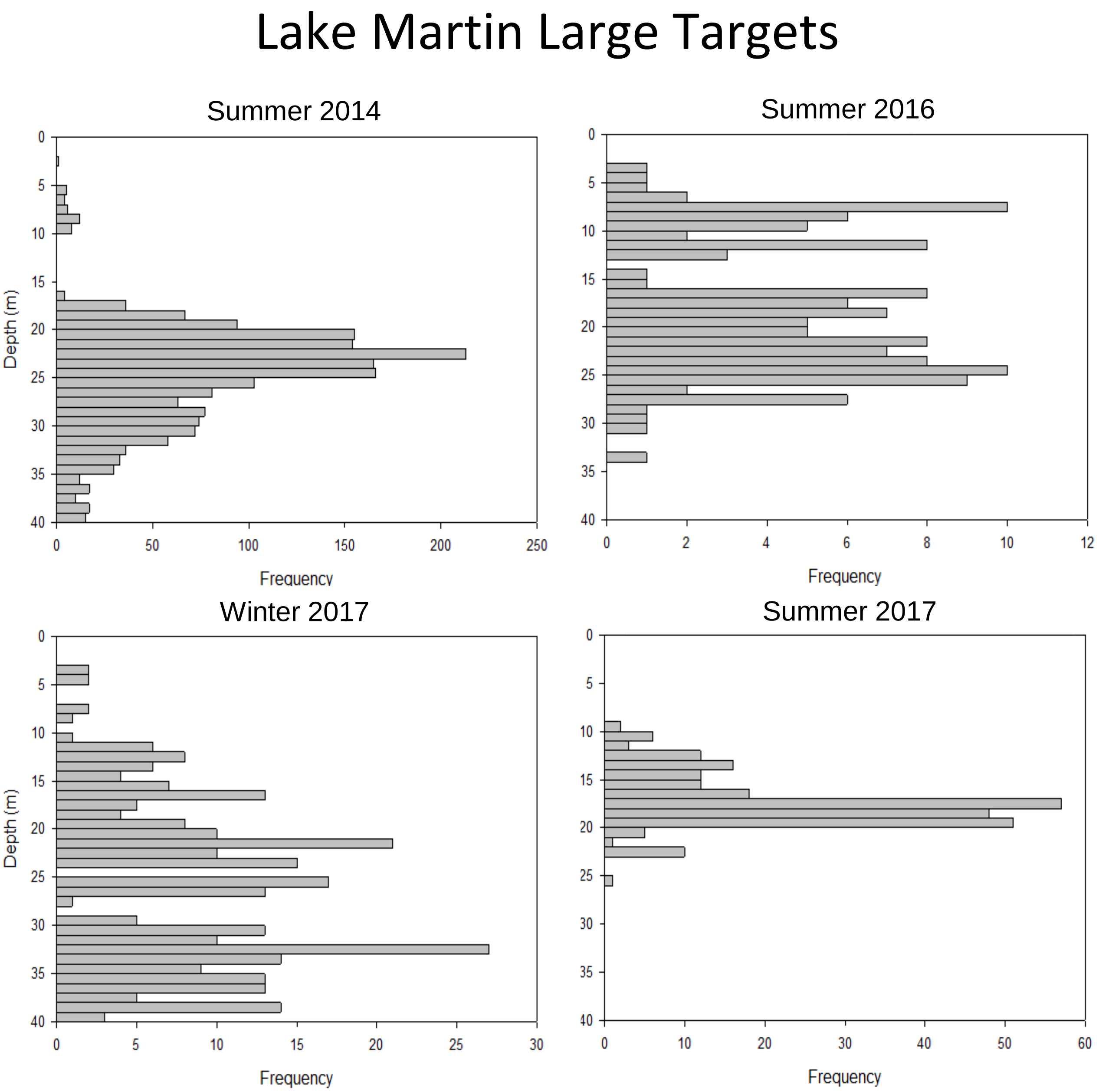


Figure 29. Single targets ≥35 cm as a function of depth for the summer 2014, summer 2016, winter 2017, and summer 2017 surveys in Lewis Smith Lake.


Lake Martin Large Targets

Summer 2014

Summer 2016

Winter 2017

Summer 2017

Figure 30. Single targets ≥35 cm as a function of depth the summer 2014, summer 2016, winter 2017, and summer 2017 surveys in Lake Martin.

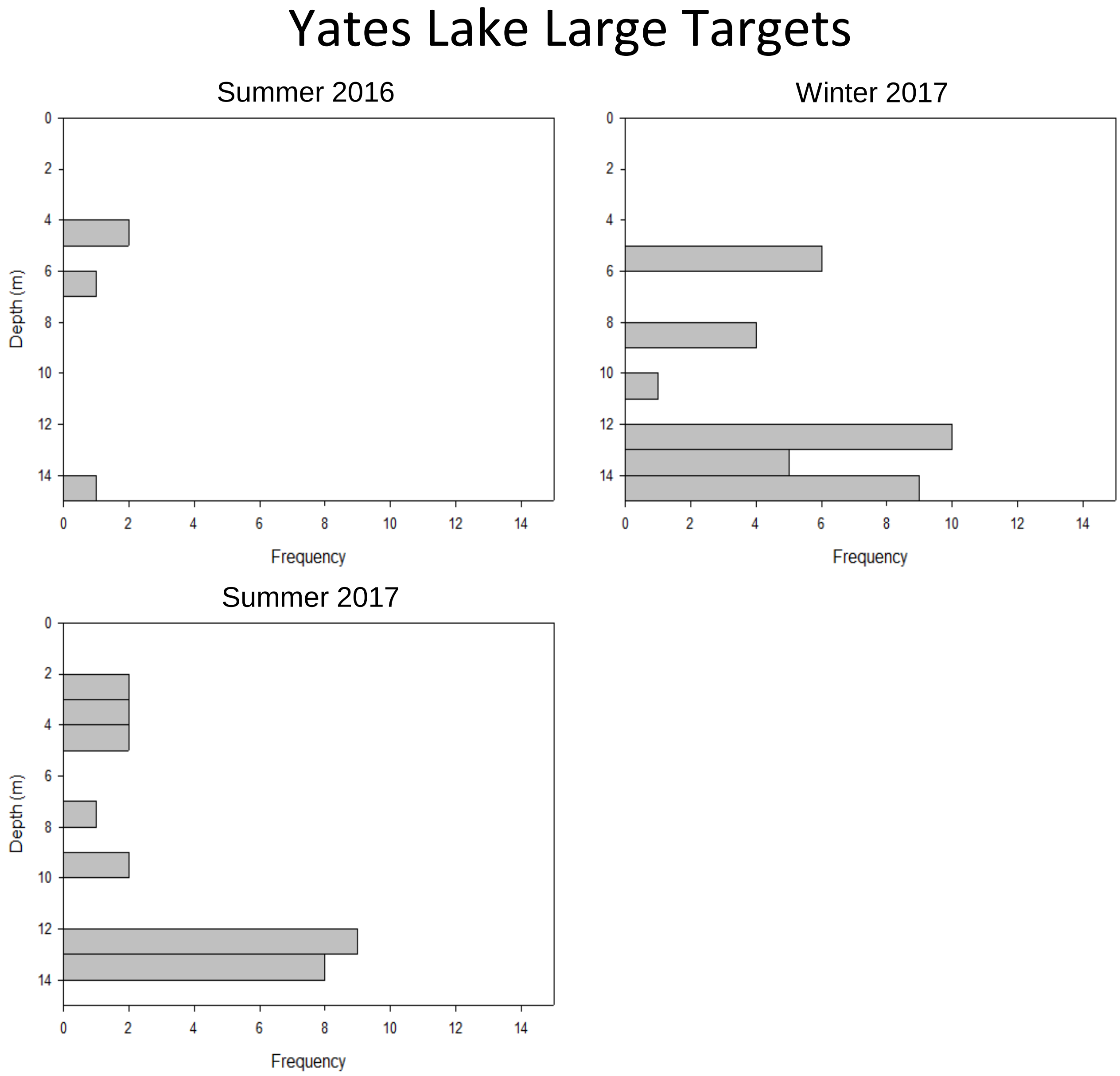


Figure 31. Single targets ≥35 cm as a function of depth for the summer 2016, winter 2017, and summer 2017 surveys in Yates Lake.

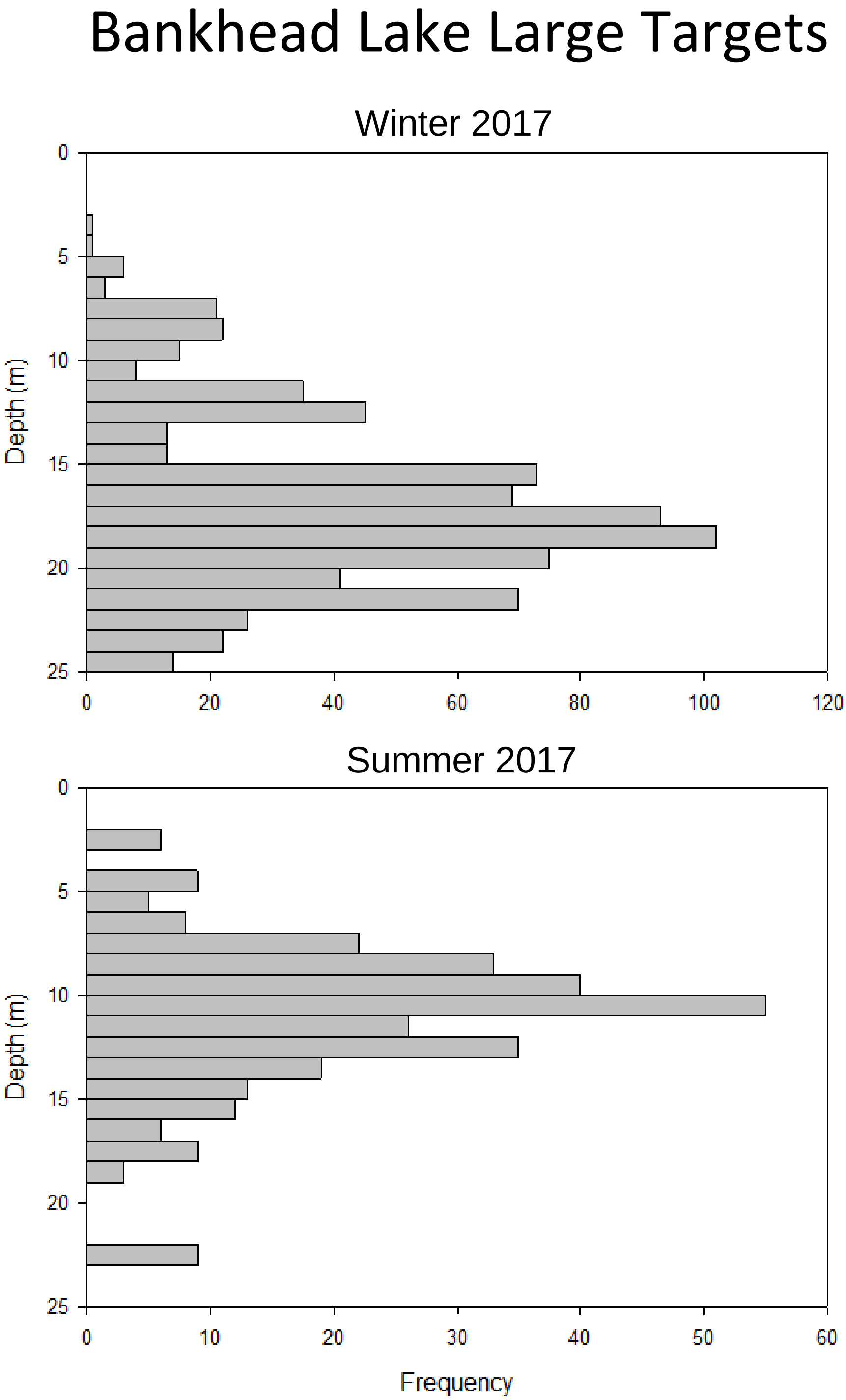


Figure 32. Single targets ≥35 cm as a function of depth for the winter 2017 and summer 2017 surveys in Bankhead Lake.

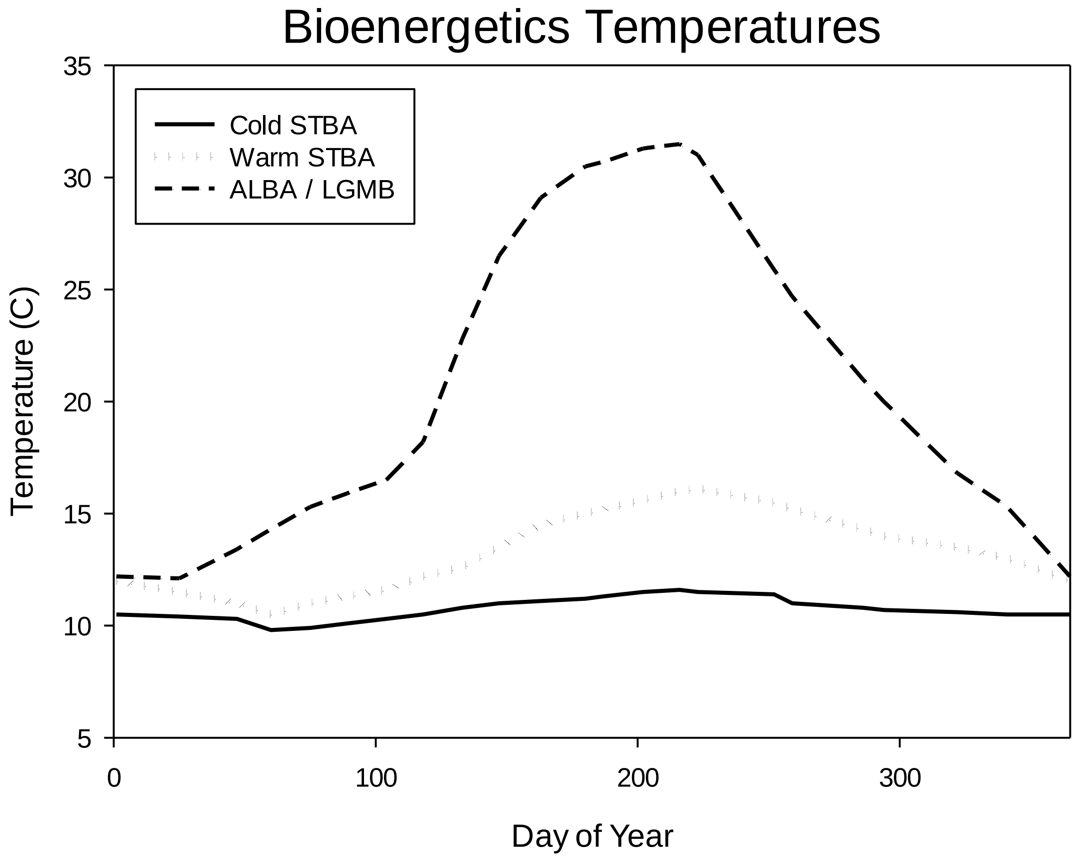


Figure 33. Temperatures incorporated in bioenergetics simulations for each species and scenario. Two temperature scenarios were simulated for Striped Bass (STBA), and one temperature scenario was used for both Largemouth Bass (LGMB), and Alabama Bass (ALBA).

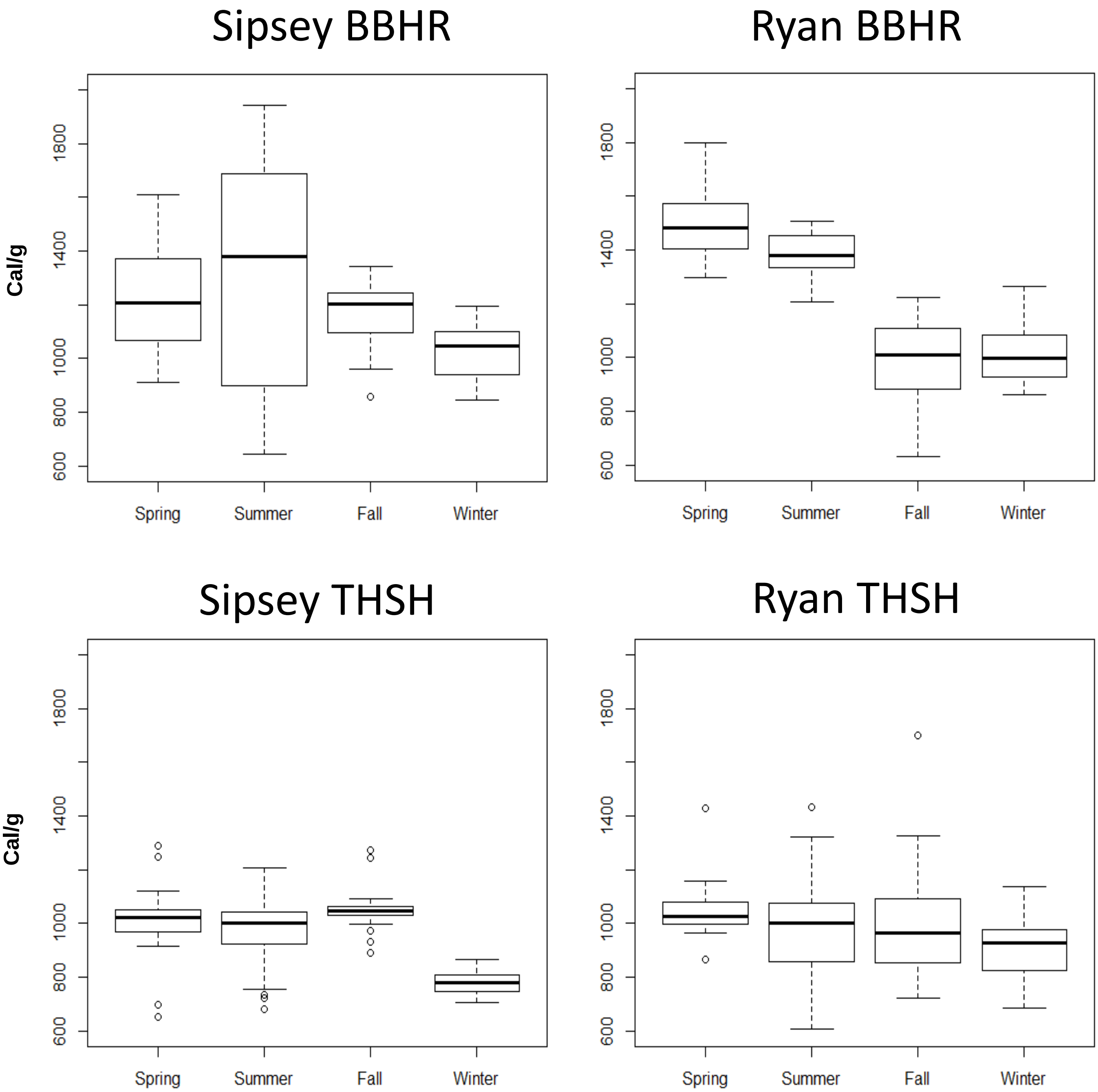


Figure 34. Seasonal caloric densities for Blueback Herring (BBHR) and Threadfin Shad (THSH) by arm.

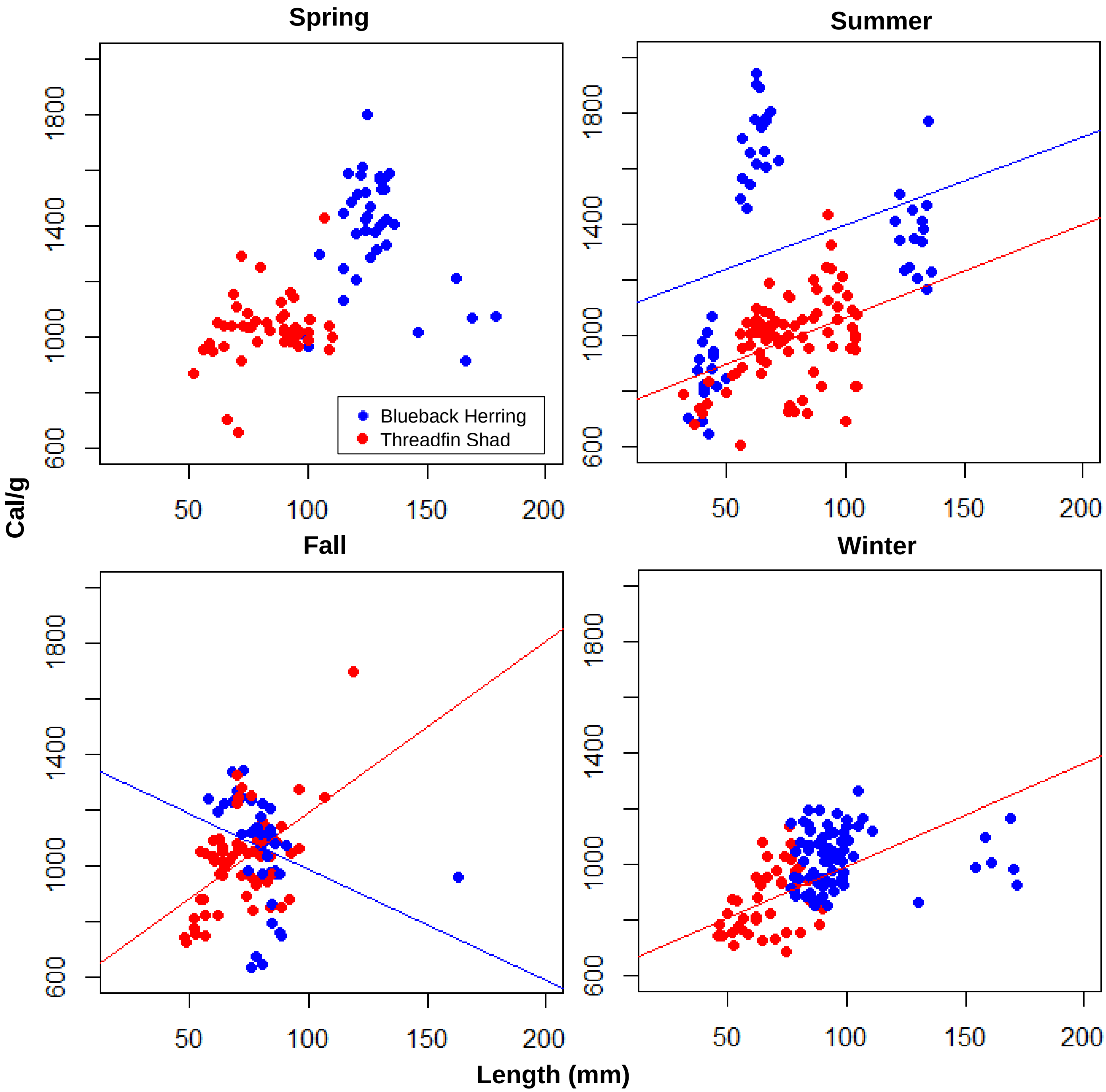


Figure 35. The effect of length on caloric density for Blueback Herring (Blue) and Threadfin Shad (Red) for each season. Significant regression lines are represented on plots.

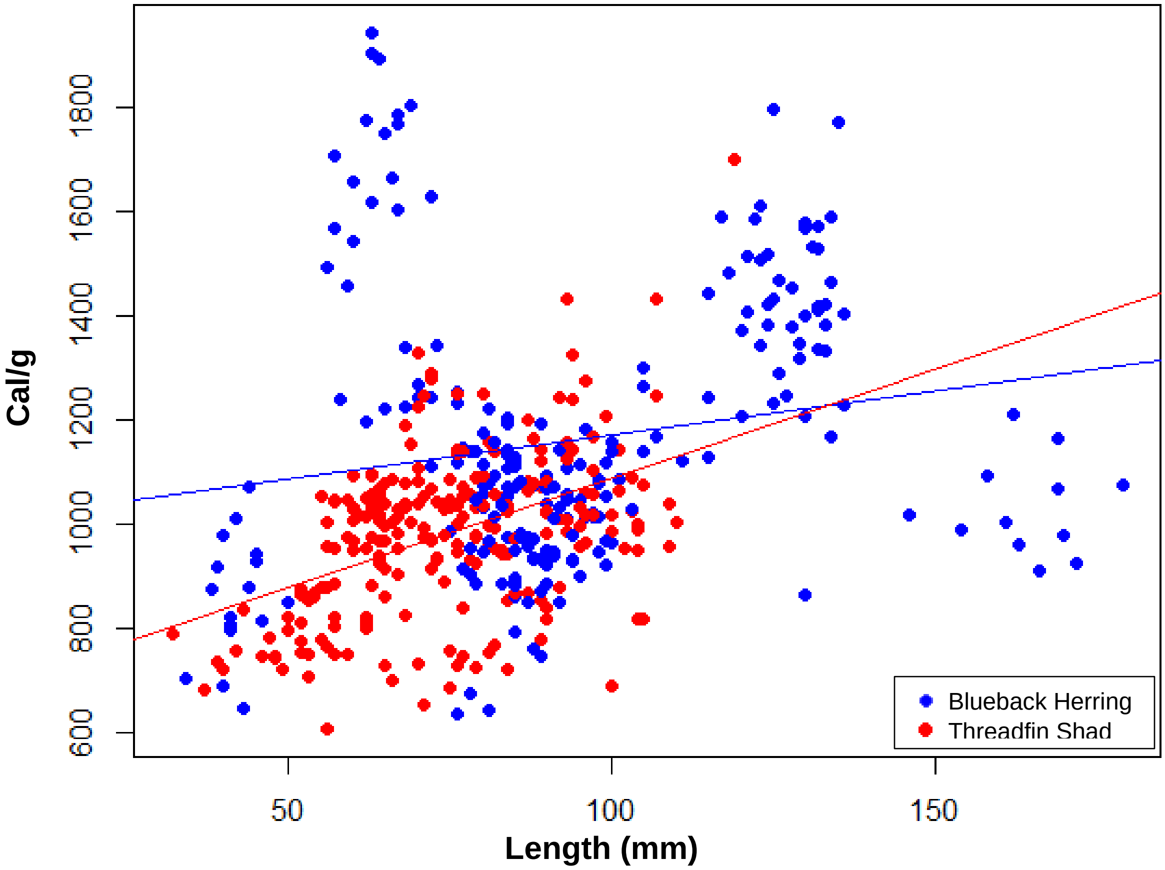


Figure 36. Relationship between caloric density and length for all fish.

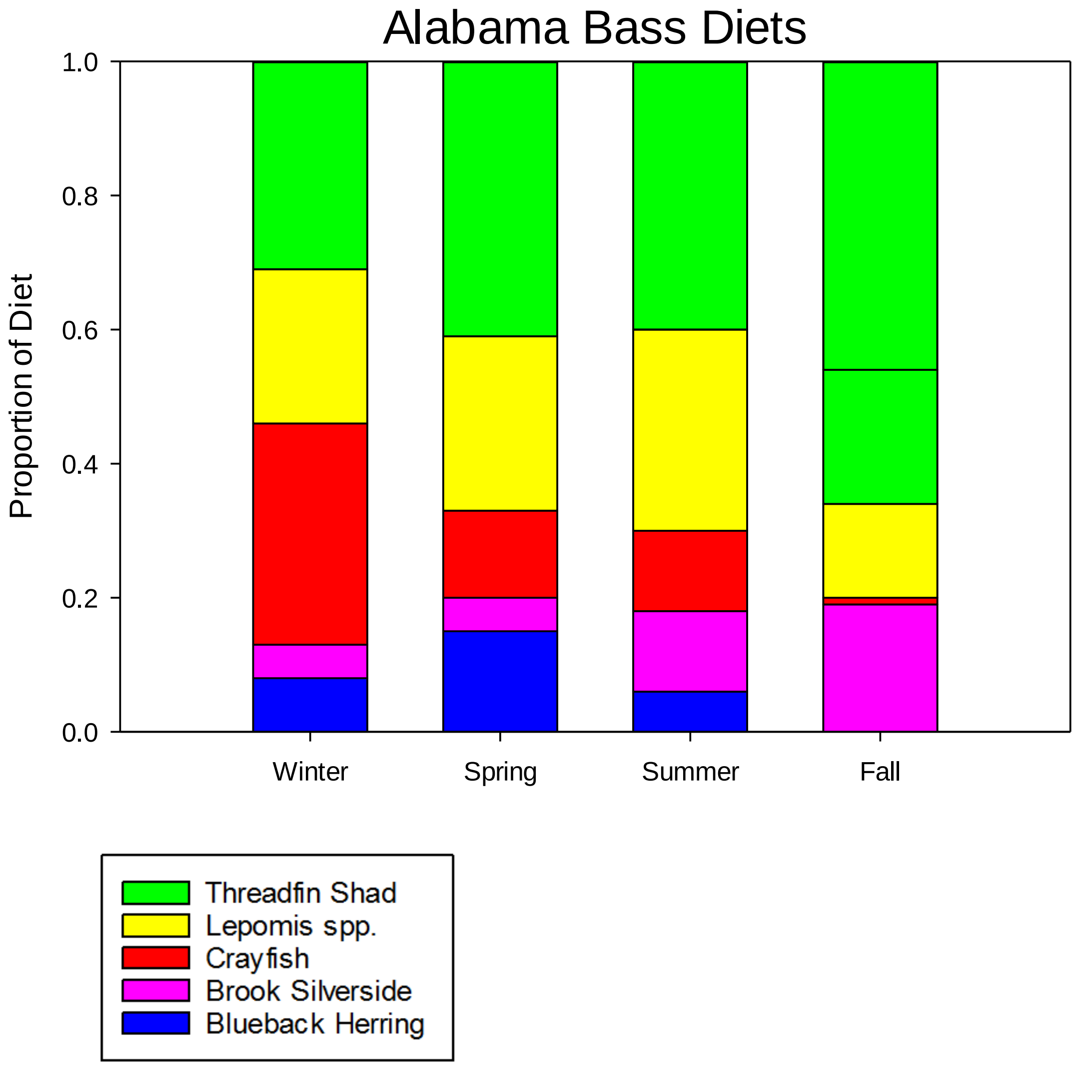


Figure 37. Alabama Bass diet proportions for winter, spring, summer, and fall.

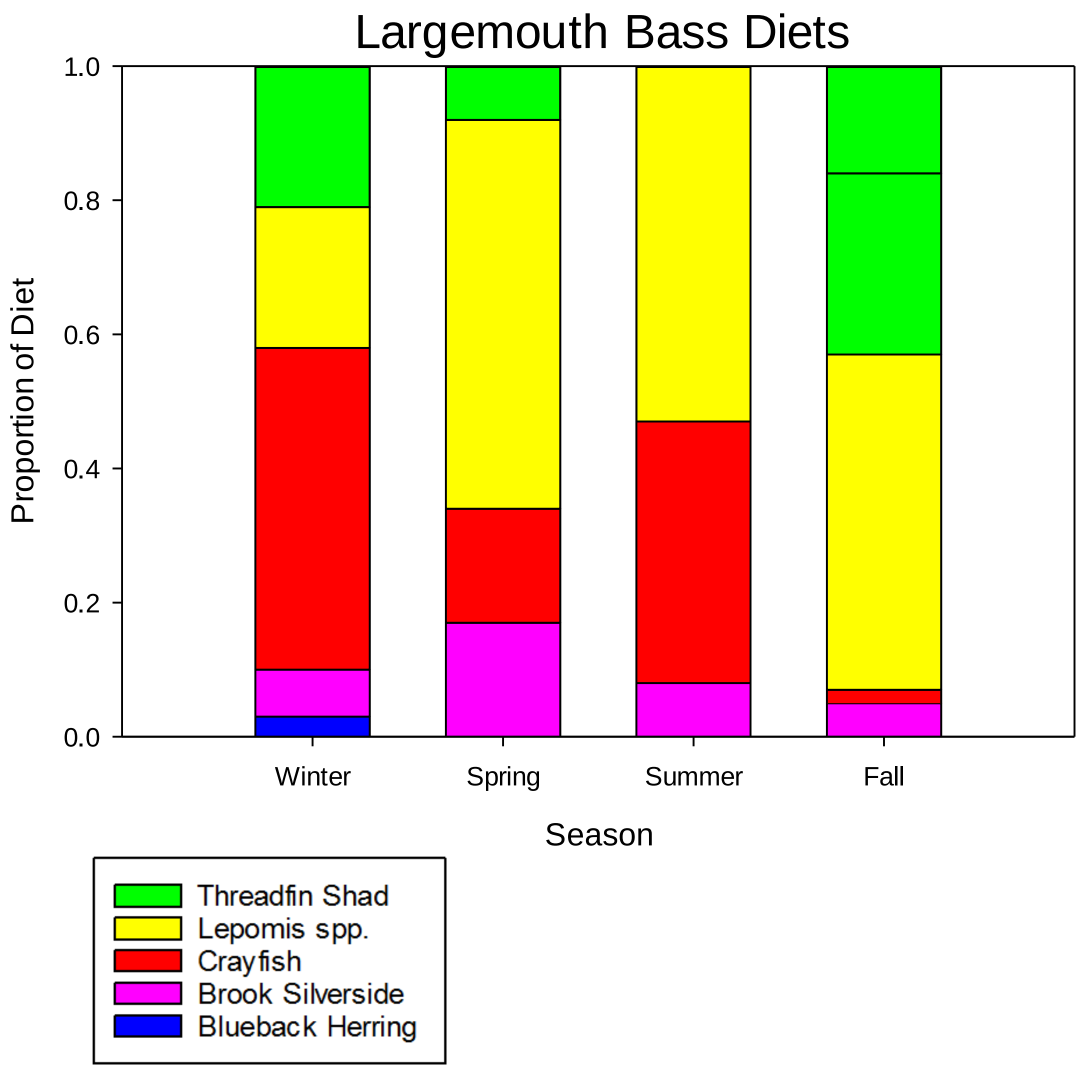


Figure 38. Largemouth Bass diet proportions for winter, spring, summer, and fall

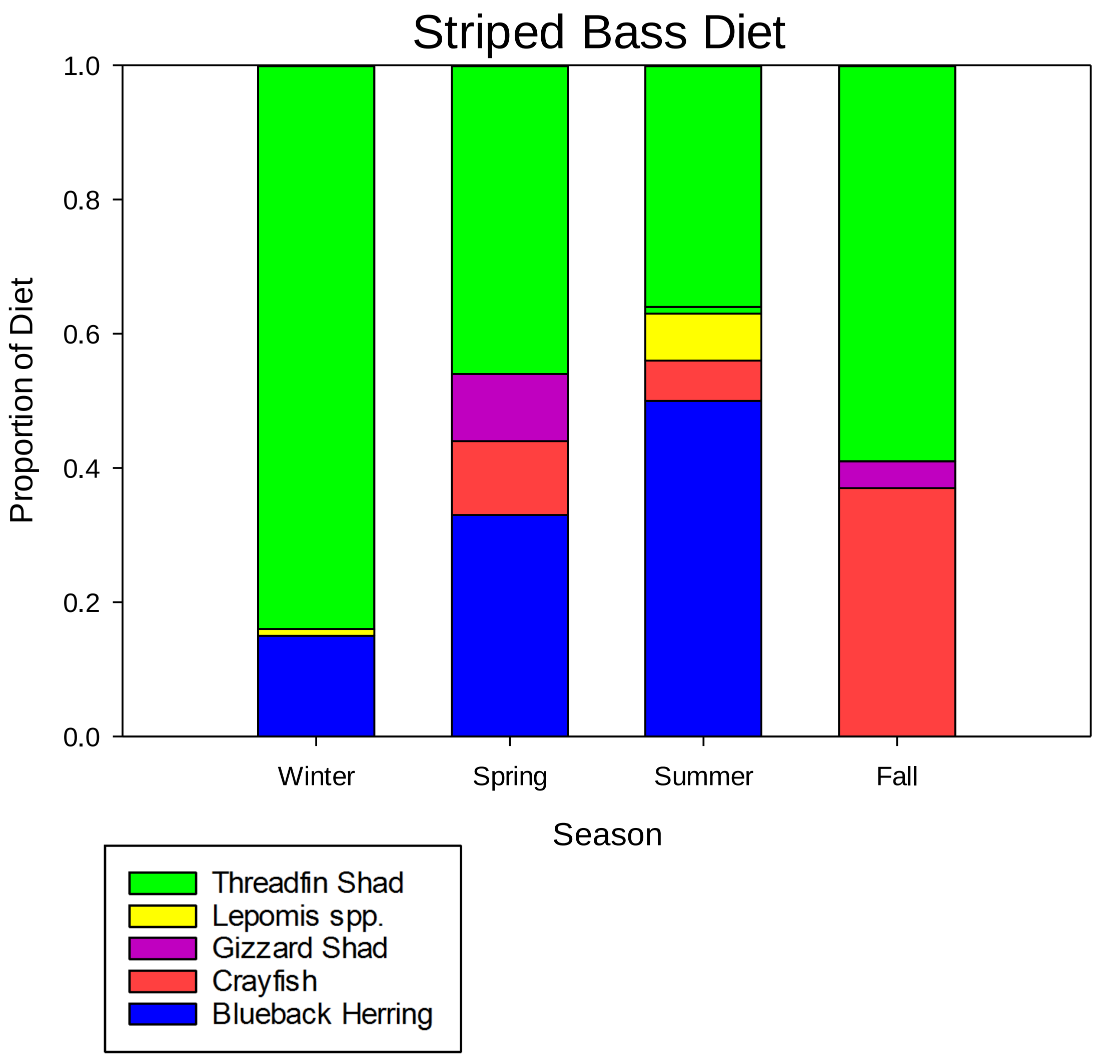


Figure 39. Striped Bass diet proportions for winter, spring, summer, and fall.

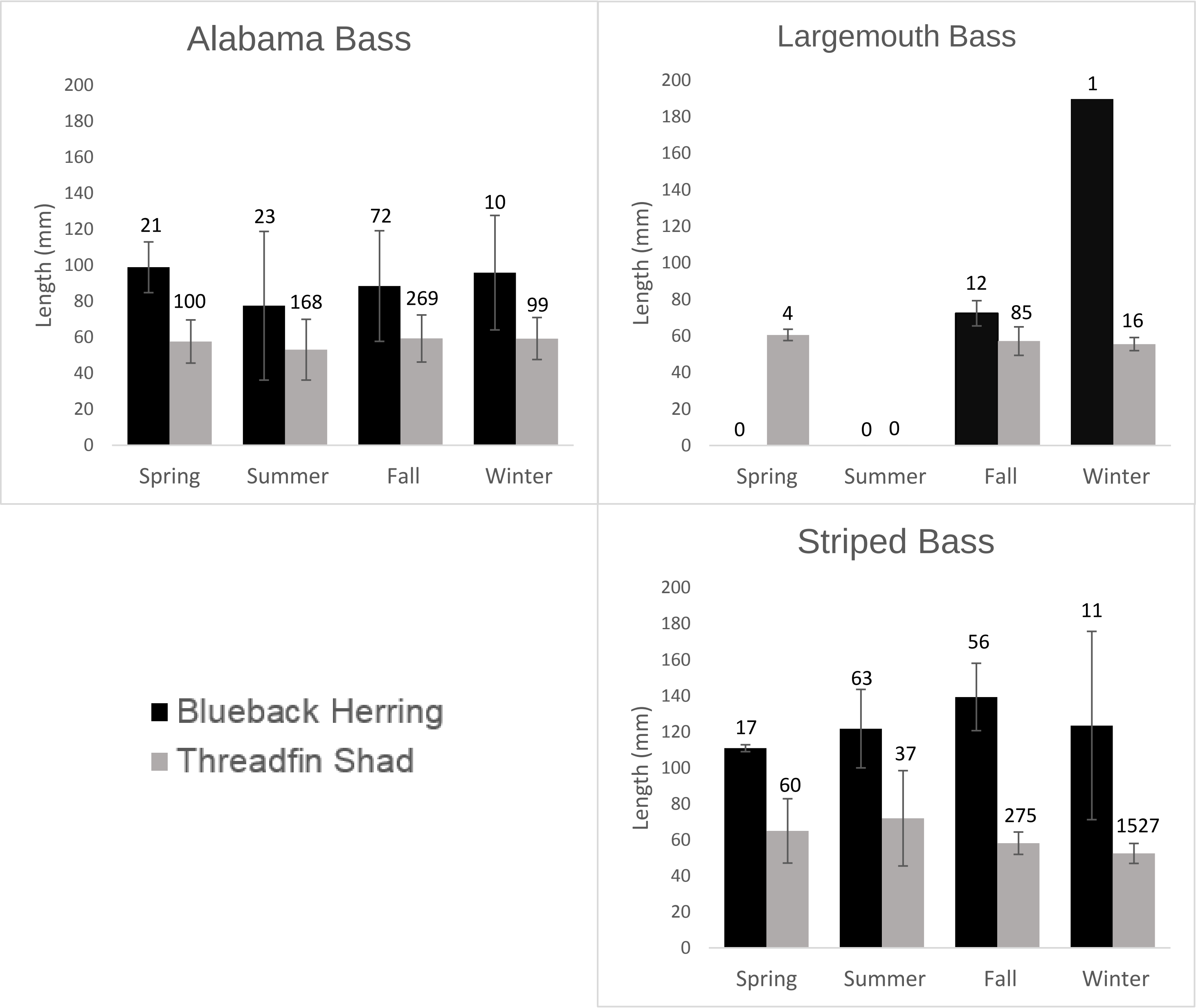


Figure 40. Mean Threadfin Shad and Blueback Herring lengths in piscivore stomachs for each season. Error bars represent standard deviations from the mean. Numbers above bars represent sample sizes in piscivores diets.

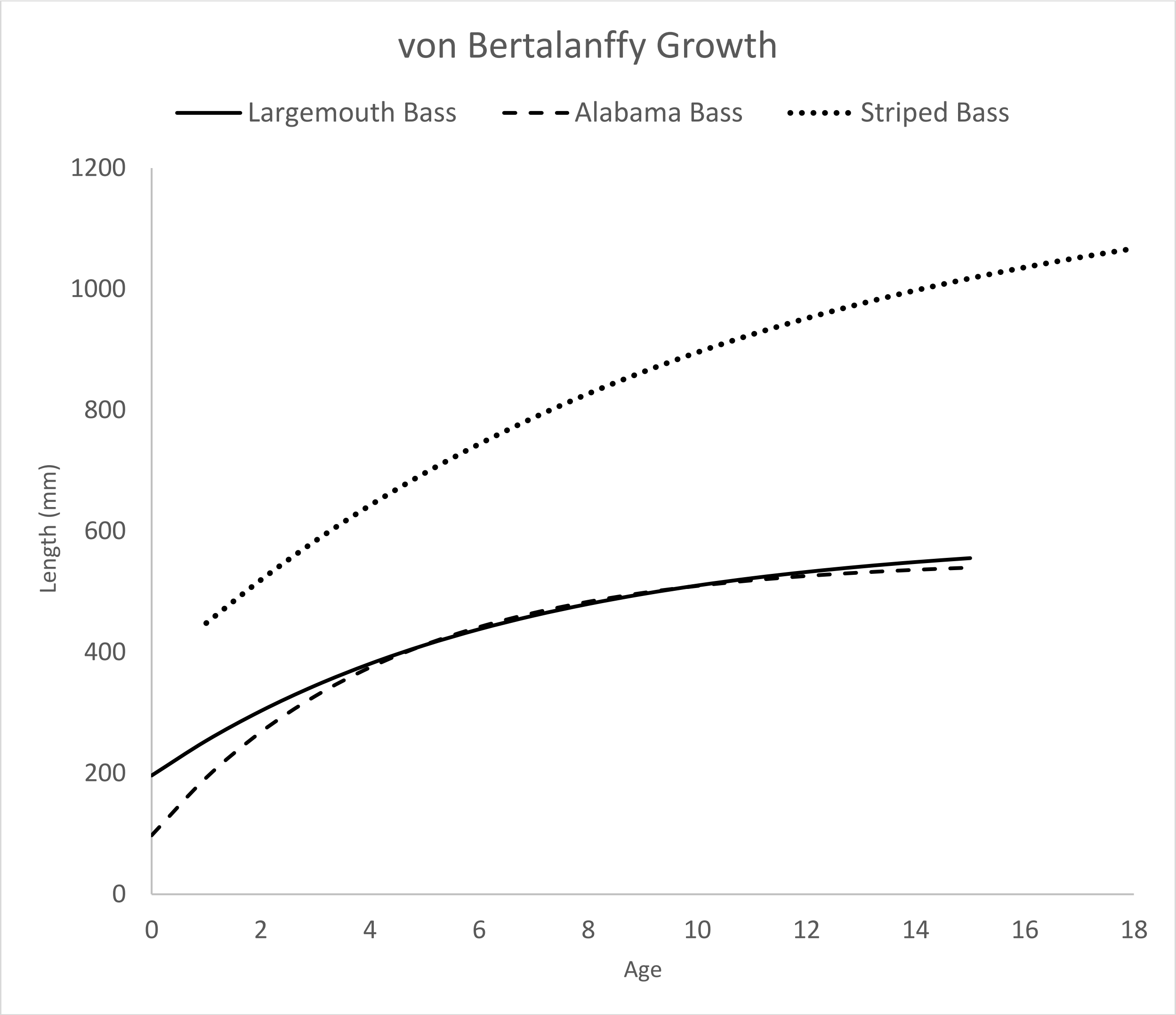


Figure 41. Von Bertalanffy Growth curves for Alabama Bass, Largemouth Bass, and Striped Bass in Lewis Smith Lake, Alabama.

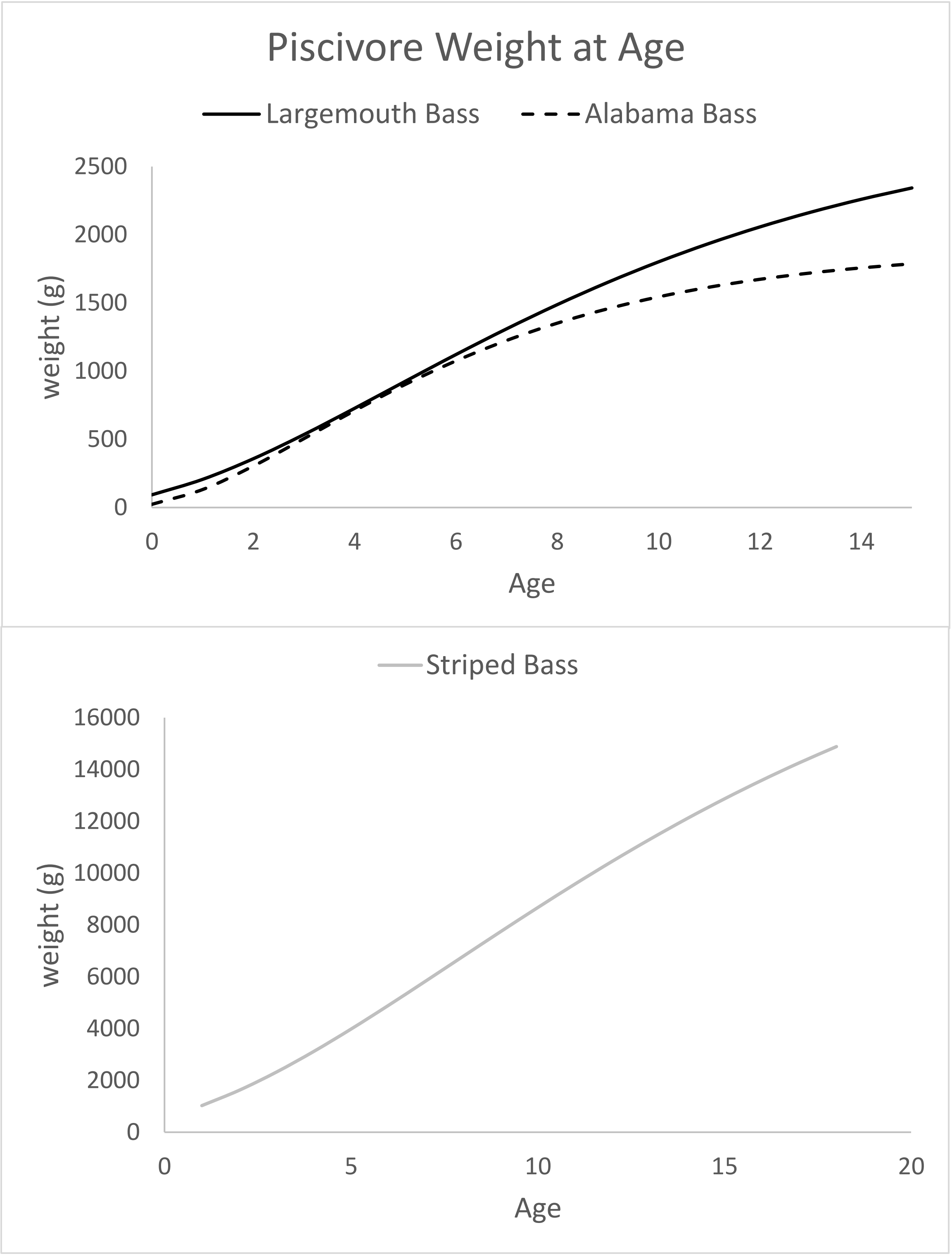


Figure 42. Weight-at-age regressions for Alabama Bass, Largemouth Bass, and Striped Bass in Lewis Smith Lake, Alabama.

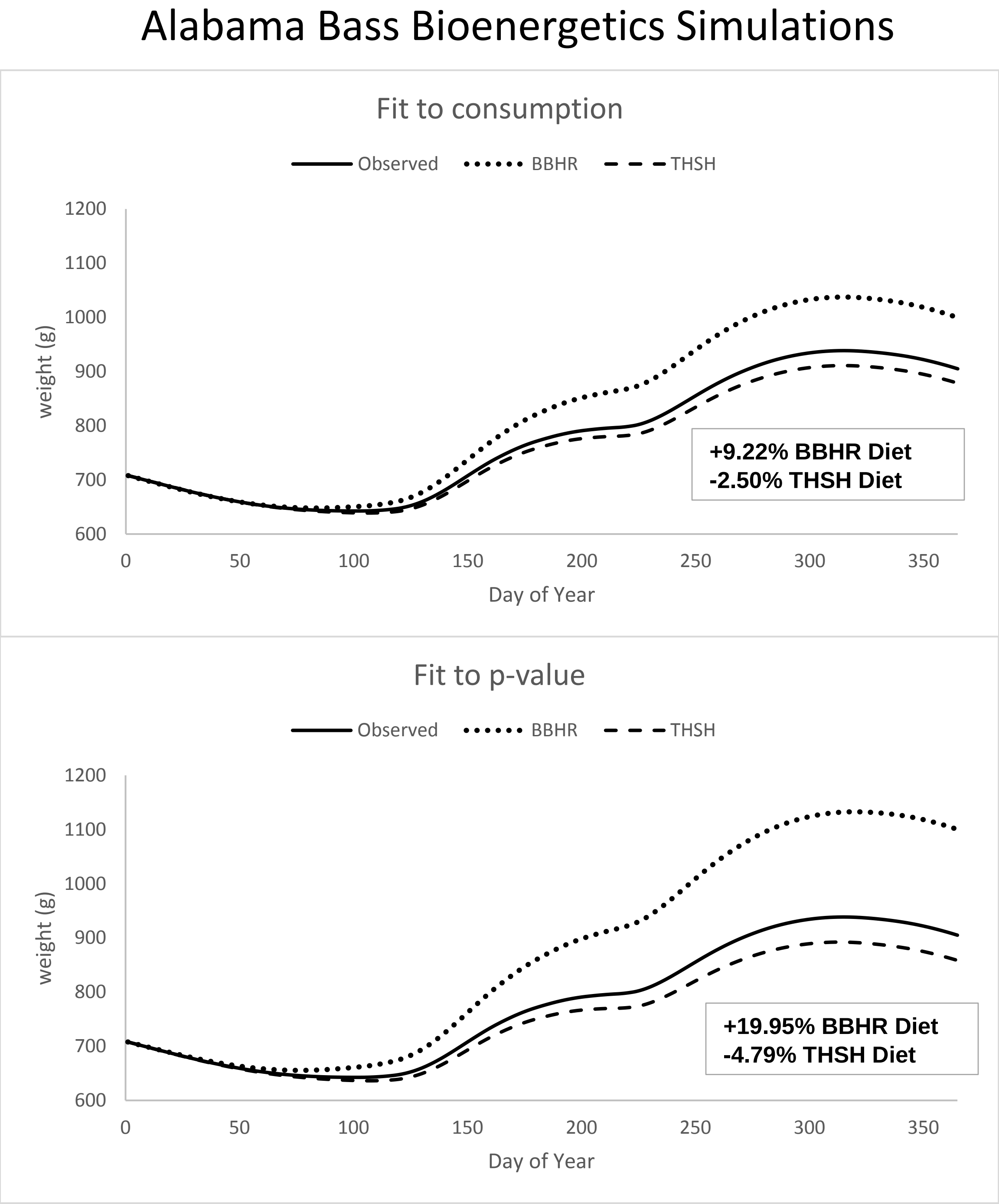


Figure 43. Alabama Bass bioenergetics simulation of growth from age 4 to 5. Solid lines represent simulations with the observed diet proportions. Dotted lines represent simulations with Blueback Herring replacing Threadfin Shad diet proportions, and dashed lines represent Threadfin Shad replacing Blueback Herring diet proportions. Simulations in the upper panel are fit to the model estimated weight of consumed prey from the observed diet proportions while bottom simulations are fit using the model estimated p-value from observed diet proportions. Percentages represent one-year growth differences given a Blueback Herring or Threadfin Shad diet vs. the observed diet.

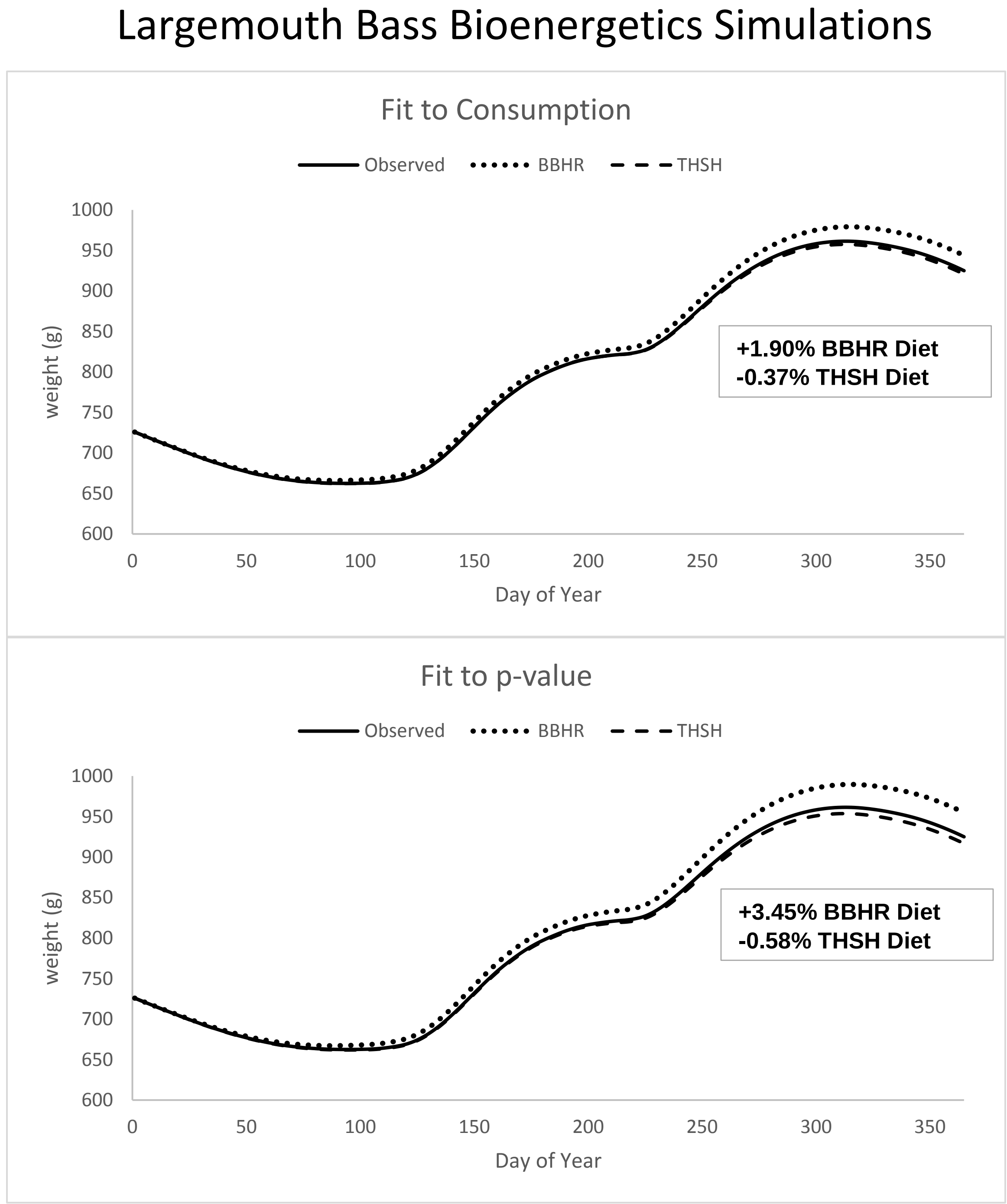


Figure 44. Largemouth Bass bioenergetics simulation of growth from age 4 to 5. Solid lines represent simulations with the observed diet proportions. Dotted lines represent simulations with Blueback Herring replacing Threadfin Shad diet proportions, and dashed lines represent Threadfin Shad replacing Blueback Herring diet proportions. Simulations in the upper panel are fit to the model estimated weight of consumed prey from the observed diet proportions while bottom simulations are fit using the model estimated p-value from observed diet proportions. Percentages represent one-year growth differences given a Blueback Herring or Threadfin Shad diet vs. the observed diet.

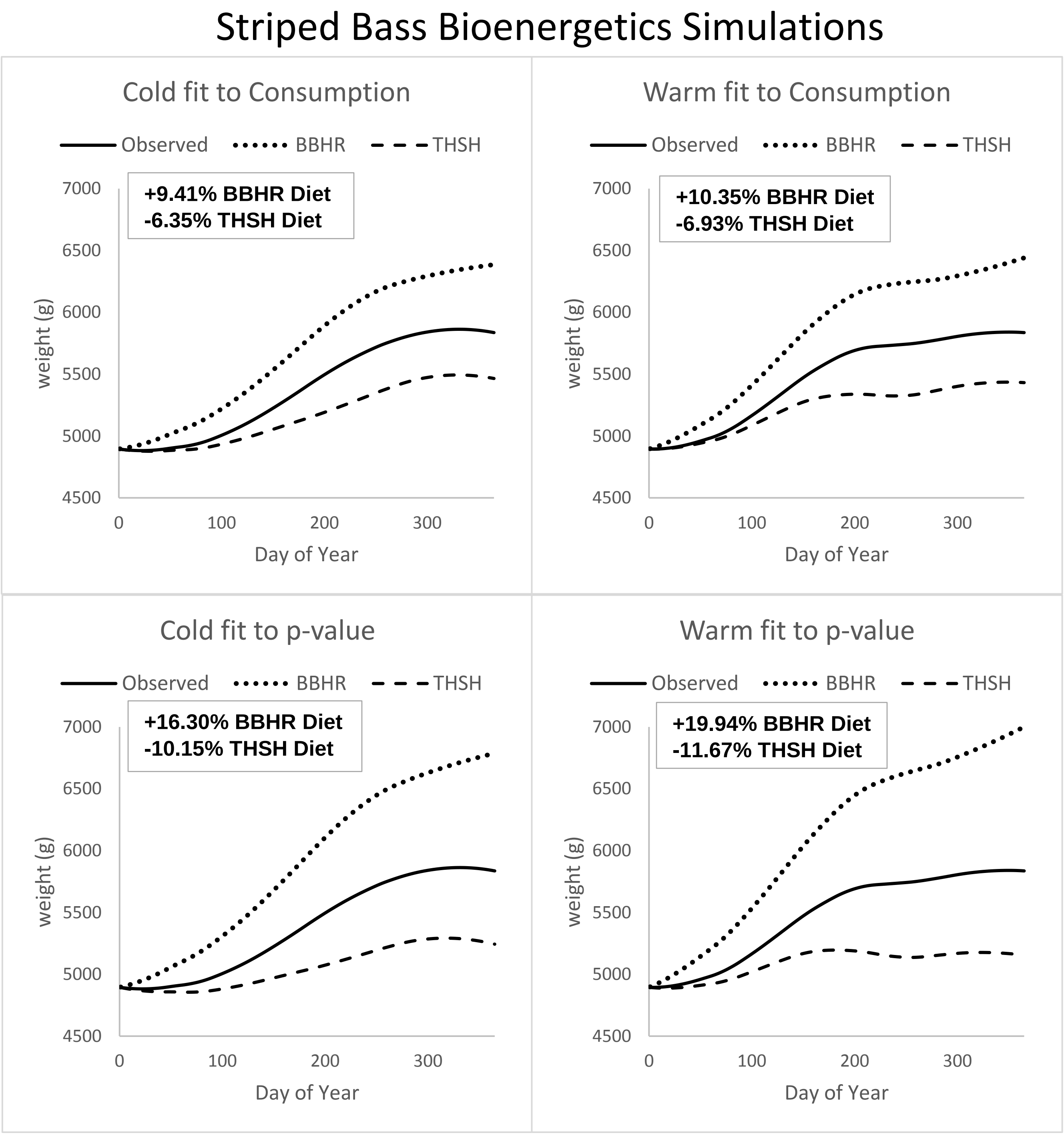


Figure 45. Striped Bass bioenergetics simulations of growth from age 6 to 7. Solid lines represent simulations with the observed diet proportions. Dotted lines represent simulations with Blueback Herring replacing Threadfin Shad diet proportions, and dashed lines represent Threadfin Shad replacing Blueback Herring diet proportions. Simulations in the upper panel are fit to the model estimated weight of consumed prey from the observed diet proportions while bottom simulations are fit using the model estimated p-value from observed diet proportions. Left simulations represent the cold water scenarios while right simulations represent warm water scenarios. Percentages represent one-year growth differences given a Blueback Herring or Threadfin Shad diet vs. the observed diet.