Increasing Largemouth Bass Carrying Capacity Using Destratification: A Case Study

J. Wesley Neal, *Mississippi State University, Box 9690, Department of Wildlife, Fisheries and Aquaculture, Mississippi State, MS 39762*

Abstract: Aeration can circulate waters by disrupting thermal density differences associated with stratification, allowing homogenization of temperature, oxygen, and other physicochemical characteristics within the water body. Use of lake and pond destratification as a management tool has been increasing in recent years, yet data are limited regarding its effects on fish communities. This case study examines the response of a largemouth bass (*Micropterus salmoides*) population to destratification in a 2.4-ha pond over nearly a decade. Biomass (35.8–42.8 kg ha^{-1}) and density (51–93 fish ha^{-1}) of stock-sized (≥200 mm TL) largemouth bass were consistently low prior to installation of the system; however, biomass tripled (129.8 kg ha^{-1}) and density quadrupled (334 fish ha^{-1}) 3.5 years after system initiation and remained high for the duration of the study. In contrast, mean weight and mean relative weight (W_r) of fish declined continuously following population expansion, suggesting that population biomass had exceeded carrying capacity. Although this study was limited to a single waterbody, the rapidity and magnitude of observed changes following years of relative stability suggests that continuous destratification increased carrying capacity for largemouth bass in this case study. These findings may have applications to other ponds characterized by anoxic hypolimnions, and more research on the effects of destratification in small impoundments is warranted. If destratification can increase standing biomass two- or three-fold, while greatly reducing the risks of oxygen-related fish kills, the cost of these systems could be more easily justified by managers of ponds and small lakes.

Key words: aeration, relative weight, biomass, density, ponds

Journal of the Southeastern Association of Fish and Wildlife Agencies 6:1–8

Hypolimnetic oxygen depletion is a limiting factor for many eutrophic lakes and ponds in the southeastern United States during warmer months of the year. During summer stratification, thermally induced density gradients prevent vertical mixing of warm surface water with cooler water from deeper areas. Without contact with the atmosphere or the oxygen-producing phytoplankton found in the shallow layers, oxygen levels in deeper areas decline towards anoxia due to decomposition and biological respiration (Boyd and Boyd 2012). This reduces oxygenated habitat available to fish and other aquatic organisms and increases the risks of summer fish kills.

Aeration is a common method of alleviating dissolved oxygen depletion and has a long history of use in lakes and reservoirs. Specific methods of aeration can be used to add oxygen to the surface layers while leaving stratification mostly intact (e.g., paddlewheels), or it can be used to partially or completely destratify a water body (e.g., diffusers). Artificial destratification disrupts density differences associated with stratification, allowing the prevailing meteorological forces and convection currents to homogenize the water body in terms of temperature, oxygen, and other physicochemical characteristics (Toetz 1981).

A common form of destratification utilizes an air compressor on the shore that pumps air through hoses on the bottom of the pond to diffusers (e.g., Mostefa and Ahmed 2012). Air is released in deeper waters, and a column of small bubbles rises to the surface, expanding and pushing water upward. The bubble plume entrains cold dense water in the hypolimnion and carries it toward the surface where it mixes with warmer epilimnetic water (van Dijk and van Vuuren 2009). Warmer surface waters sink as deeper waters are displaced, creating a circulation effect. Careful consideration of the number of diffusers and their placement can result in near complete thermal homogenization of the water body.

Lake and pond destratification as a management tool has been increasing rapidly in recent years, yet data are limited regarding its effects on fish communities. There has been substantial research on the biological effects of destratification, but this research focused mostly on water quality effects (e.g., Bormans et al. 2015) and lower trophic levels, particularly phytoplankton, zooplankton, and benthic macroinvertebrates (Toetz et al. 1972, Antenucci et al. 2005, Becker et al. 2006, Lehman 2014). A few studies have explored the effect of destratifying lakes on fish populations in terms of reducing fish kills and expanding available habitat (Fast 1966, Fast 1968, Calhoun and Hubbell 1970), but most discussions on increasing fish production via bottom-up trophic effects were theoretical. The lone exception was Johnson (1966), who reported that increased habitat due to destratification improved coho salmon (*Oncorhynchus kisutch*) fry survival from 12.9% to 42.1%; Toetz et al. (1972) subsequently calculated that this resulted in a 328%

increase in annual biomass of smolt migrants. No other attempts to assess the increase in fish production resulting from use of active destratification have been reported.

Aeration and destratification are used effectively to increase production capacity in feed-based commercial aquaculture (e.g., Losordo 1991, Boyd et al. 2018), yet their potential for recreational fisheries has not been assessed. Destratification systems are generally installed in recreational fisheries to prevent oxygen-related fish kills, but there may be additional benefits, as the expansion of habitable pond volume and increased productivity at lower trophic levels associated with destratification may influence sport fish production. This case study describes the response of a largemouth bass (*Micropterus salmoides*) population to destratification and other management activities over nearly a decade in a 2.4-ha research pond under controlled environmental conditions.

Methods

Study Site

This research was conducted on a 2.4-ha impoundment at Mississippi State University that serves as a demonstration pond for teaching and research. The pond averages around 2 m in depth with a maximum depth of nearly 4 m at full pool. The pond was designed with earthen points and humps, deeper channels, shallow stump fields in offshore areas, and a few logs and concrete structures near shore for fish habitat. Some spring growth of filamentous algae (*Pithophora* spp.) and summer growth of water primrose (*Ludwigia* spp.) occurs around the pond edges, but the pond is generally devoid of aquatic plants. The pond is not fertilized, but is naturally fertile due to watershed activities and has experienced oxygen-related fish kills in the past. Summer stratification was strong prior to destratification, with a shallow thermocline around 1.0–1.5 m and near anoxic conditions in deeper waters. The pond was originally stocked with F1 hybrid largemouth bass, bluegill (*Lepomis macrochirus*), redear sunfish (*L. microlophus*), and grass carp (*Ctenopharyngodon idella*) in the mid-1990s and was in relative equilibrium at the time of the study. Fishing is rarely allowed on the pond.

Temperature-Oxygen Sampling

Continuous temperature and oxygen vertical profiles were collected in mid-August 2009 in the study pond to demonstrate the degree of thermal stratification and oxygen depletion present during summer months. A Hydrolab DS5 multiparameter data sonde (OTT Hydromet, Inc., Loveland, Colorado) was lowered and raised from a boat at five random sites in the lower section of Blackjack Pond where water depths exceeded 2.5 m. These methods were repeated in August 2018 to demonstrate thermal and oxygen patterns present with continuous destratification system operation. For each period, values at 0.25-m intervals from 0.5 m to the pond bottom were averaged across the five stations and standard deviations were calculated as a measure of variance.

Largemouth Bass Sampling

Mark-recapture population estimates for largemouth bass were conducted annually since 2010, with the exceptions of 2011 and 2014. Boat-mounted electrofishing standardized to an output of 3500–4000 W (Burkhardt and Gutreuter 1995) was used. The entire shoreline and available offshore habitats were electrofished at least once and often multiple times in mid-spring and all largemouth bass were collected, measured (TL, mm) and weighed (g); stock-size fish (≥200 mm total length) were marked by clipping a pelvic fin prior to release.

Marked fish were given sufficient time to reintegrate into the population prior to recapture about four weeks later. A similar sampling protocol was used for recapture sampling efforts, and all largemouth bass were collected, measured, and examined for marks. Population size (N) of stock-size largemouth bass was estimated using Chapman's modification of the Petersen index (Chapman 1951), with a target 95% confidence interval of ±25% of N (Robson and Regier 1964). Estimated N of stock-size largemouth was multiplied by their mean weight to estimate total stock-size biomass. Body condition was determined using the relative weight (W_r) index (Henson 1991, Neumann et al. 2012).

Habitat Alterations

In 2012, Vertex Water Features (Pompano Beach, Florida) donated a destratification system to Mississippi State University. The system was designed by their engineers specifically for the study pond and consisted of five diffusers strategically placed in the bottom of the pond and supplied by a compressor cabinet on the shore. This system was installed and activated in November 2012. The system was operated continuously for the remainder of the study with the exception of temporary power outages and scheduled maintenance which generally lasted less than 24 h.

In late spring 2014, Mossback Fish Habitat (Springdale, Arkansas) donated nine artificial fish attractors to Mississippi State University, which were placed in the study pond. These consisted of three small nursery attractors (Root Wad kit) in the shallow upper end of the pond, three larger habitat complexes (Safe Haven kit) in deeper water, and three paired horizontal habitat complexes (Trophy Tree kit) on the dam. The artificial habitats accounted for about 0.5% of the pond's total surface area. No other habitat changes were made to the pond over the course of the study.

Statistical Analysis

The study covered nine consecutive years, with three years before destratification and six years with ongoing destratification. The data suggested a three-year transitional period prior to largemouth bass population response, so I analyzed the data using a *post-hoc* time series approach of 2010–2012 (pre-installation), 2013–2015 (transitional), and 2016–2018 (response). Statistical comparisons among the three periods used analysis of variance (ANOVA) for biomass, density, and mean weight. Relative weight served as a measure of condition, but for statistical comparison, slopes of weight-length regressions were compared between years using analysis of covariance (ANCOVA, PROC GLM) on $\log_{10}$-transformed data. Alpha was set at $P = 0.05$.

Results

The study pond was strongly stratified during summer months prior to system installation, with a clear thermocline establishing between 0.75 and 1.75 m and dissolved oxygen levels less than 2.0 mg L^{-1} at 1.25 m and below (Figure 1). Following installation, continuous system operation created a pond environment with nearly homogenous temperature in the water column. Dissolved oxygen remained stratified despite thermal destratification, although deeper areas maintained habitable oxygen levels, exceeding 2.0 mg L^{-1} at all but the deepest depth strata (2.75 m). This more than doubled the habitable depth of the pond during summer.

Biomass of stock-sized largemouth bass during the pre-installation period ranged from 35.8 to 42.8 kg ha^{-1} (Table 1) and remained low during the transitional period (range = 42.3–42.7 kg ha^{-1}). Estimated density of stock-sized largemouth bass was also consistently low during the pre-installation and transitional periods, with estimates ranging from 51 to 93 fish ha^{-1}. Beginning

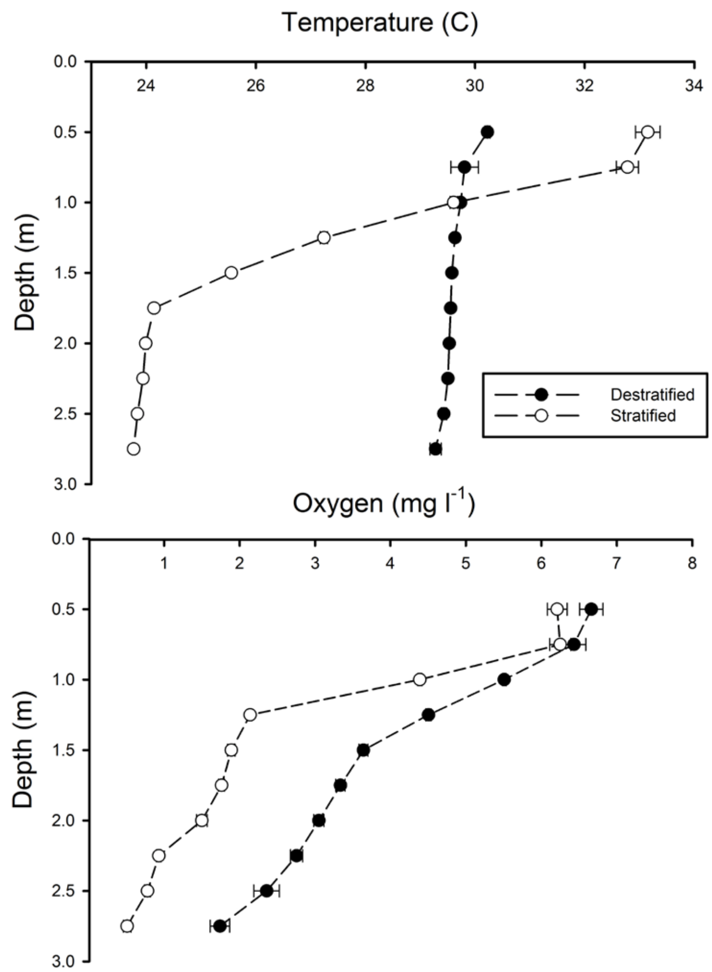

Figure 1. Mid-summer temperature (Top) and oxygen (Bottom) conditions in Blackjack Pond from before (14 August 2009, Stratified) and after (30 August 2018, Destratified) system installation. Error bars represent one standard deviation of the mean ($n = 5$).

Table 1. Population estimate parameters and population statistics from Blackjack Pond from 2010 to 2018. Data are separated into pre-installation (2010–2012, no destratification), transitional (2013–2015, first three years of destratification), and response (2016–2018, observed largemouth bass population changes) periods. Number of fish marked (M), captured (C), and recaptured (R) were used to estimate stocked-size largemouth bass abundance (N). 95% confidence bounds (CB) for N are presented.

Response period	Year	Pop. est. parameters M	C	R	N	95% CB (±N)	Density (fish ha^{-1})	Mean weight (kg)	Biomass (kg ha^{-1})	Mean W_r
Pre-installation	2010	54	48	11	224	42.0%	93	0.459	42.8	80.1
	2011	–	–	–	–	–	–	–	–	–
	2012	70	27	15	123	27.6%	51	0.685	35.8	81.7
Transitional	2013	58	60	20	170	27.3%	71	0.602	42.7	79.3
	2014	–	–	–	–	–	–	–	–	–
	2015	80	81	33	194	19.4%	81	0.522	42.3	87.6
Response	2016	124	230	35	801	25.0%	334	0.389	129.8	83.9
	2017	180	244	49	886	20.9%	369	0.324	119.4	80.5
	2018	140	266	42	875	22.6%	364	0.273	99.5	73.8

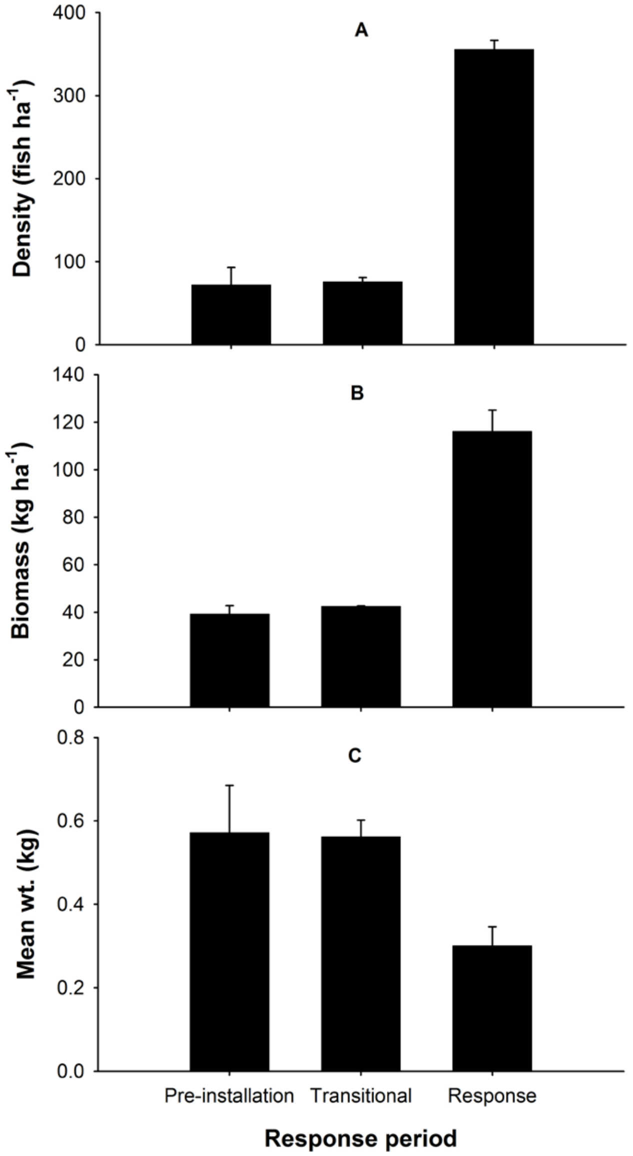

Figure 2. Estimates of (A) stock-sized (≥200 mm TL) largemouth bass density (N ha^{-1}), (B) biomass (kg ha^{-1}), and (C) mean weight (kg) in the study pond from 2010 to 2018. Error bars are one standard error.

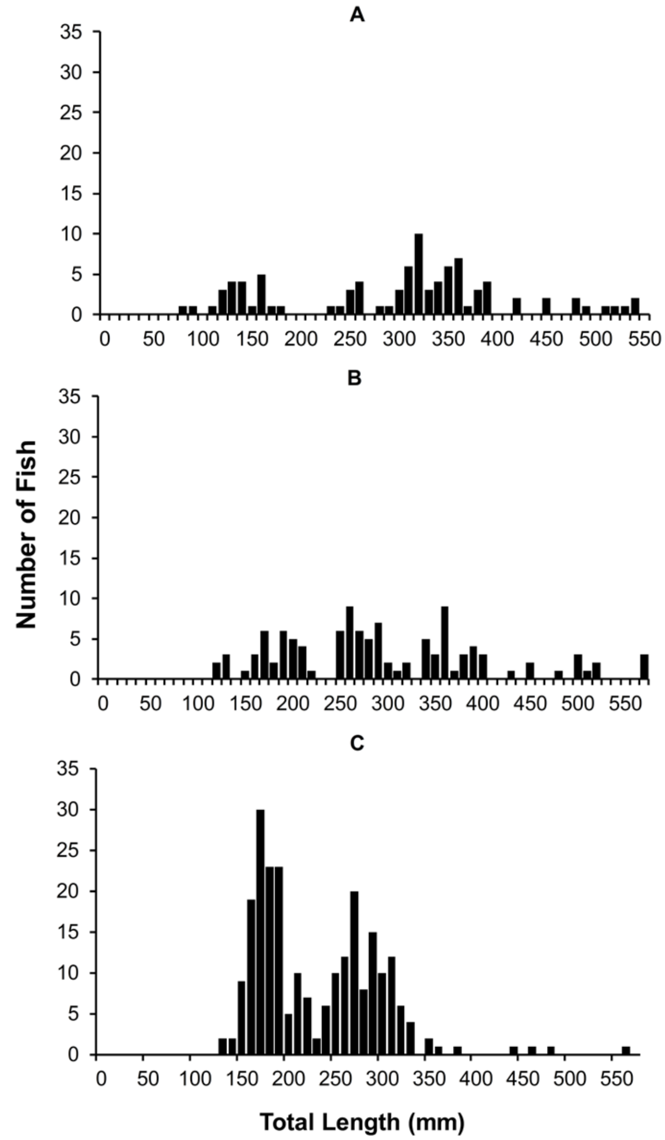

Figure 3. Largemouth bass length distributions in Blackjack Pond (A) prior to destratification system installation (2012), (B) during the transitional period with destratification (2015), and (C) following population response to destratification (2018).

in the fourth year following installation, largemouth bass populations appeared to respond to environmental changes. Biomass tripled in 2016 compared to previous estimates and average density more than quadrupled (Table 1); both remained high in 2017 and 2018. Mean density (F = 165.01, df = 2, 6; P < 0.001) and biomass (F = 39.06, df = 2, 6; P = 0.002) were higher in the response period than in the other two periods (Figure 2).

Concurrent to the increases in biomass and abundance, mean weight of individual fish declined precipitously. Prior to the population increase, mean weight of individual stock-sized largemouth bass peaked at 0.685 kg just before the destratification system was installed, and subsequently dropped every year to 0.274 kg in 2018 (Table 1). Mean weight (F = 5.49, df = 2, 6; P = 0.071) was marginally higher in the response period than in the other two periods (Figure 2), and there was a significant negative trend in mean weight post-destratification (r^2 = 0.95; P = 0.027). Relative weight also declined following the population expansion, peaking near 88 in 2015 just before biomass and abundance increased, then falling each year to a low near 74 in 2018 (Table 1). Length-weight regression slopes were significantly different between years (F = 9315.25, df = 13,959, P < 0.001), with slopes declining post-destratification.

Increases in largemouth bass abundance and biomass appeared to be largely a function of increased recruitment (Figure 3). Length distributions prior to system installation and during the transitional period were protracted, with low numbers of fish but consistent representation in all size classes. In 2016, a strong year class began to recruit, resulting in 82% of the population measuring less than 300 mm total length by the March sample. In 2017, another strong year class recruited, and 93% of the population was less than 350 mm. By 2018, about 97% of the population was less than 350 mm.

In response to increases in abundance and declining mean weight and condition, biomass removals were conducted following the population estimates in 2017 and 2018. In 2017, 24% of the stock-sized biomass was removed, and another 21% was removed in 2018. Only largemouth bass 200–330 mm total length were removed in an attempt to reduce abundant smaller fish. Because removals occurred after estimate completion, only the 2018 estimate would have been influenced by removing largemouth bass (the 2017 removal). However, the 2018 population estimate was nearly identical to the 2017 estimate (density differed by 5 fish ha^{-1}), suggesting compensatory effects of the harvest on natural mortality.

Discussion

The population of largemouth bass in this 2.4-ha pond significantly expanded following destratification and habitat alterations. Although lack of replication precludes conclusive assignment of causation, the sheer magnitude of the observed changes following at least six years of relative stability suggests that some combination of destratification or habitat augmentation may have been responsible. Biomass and abundance increases of 300% or more can be achieved via implementing a proper fertilization program (Neal and Kröger 2012, Stone et al. 2012), but changes of this magnitude without a causative factor are inconsistent with ecological carrying capacity theory (e.g., Chapman and Byron 2018), even in relatively dynamic fisheries such as southern ponds.

It is unlikely that the addition of artificial habitats was responsible for the observed changes. Although there is some evidence that structural habitat may increase productivity by serving as an attachment surface for periphyton and other biofilms (Pardue 1973, Neal and Lloyd 2017), the reported changes were usually only a fraction of the overall carrying capacity. Further, the ratio of structural coverage to lentic surface area in previous studies was considerably greater than in the current study, where less than 1% of the pond area contained artificial habitat. Several studies have hypothesized that the addition of structural habitat may increase carrying capacity and biomass, but these studies limited their hypothesis to areas in the immediate vicinity of the structures and did not extrapolate biomass increases system-wide (Bortone et al. 1994, Polovina 1994, McCann et al. 1998, Miranda 2017). Thus, it is likely that the habitat in the study pond served more to concentrate fish around habitats rather than increase biomass or abundance, and any increases associated with biofilms or higher recruitment should have been minimal.

In addition, it is also implausible that the structures added in spring 2014 increased juvenile largemouth survival and recruitment. Studies have shown that complex habitat can decrease largemouth bass predation rates and increase abundance of juvenile largemouth bass (Savino and Stein 1982, Bettoli et al. 1992, Irwin et al. 1997). However, the relative amount of artificial habitats installed during this study was small compared to other habitat available to the bass, and the habitats were designed to attract predators, not conceal prey (i.e., large interstitial spaces allowing free movement of predators). The exception were the Root Wad kit attractors, designed to serve as nursery habitat. These habitats were electrofished periodically during the study and few juvenile bass were collected; juvenile bass were much more common along the shoreline of the pond.

Another factor which hypothetically could have affected results might be the continued introgression of the F1 largemouth bass originally stocked. Heterosis, also known as hybrid vigor, is manifested when Florida bass (*M. s. floridanus*) are crossed with northern largemouth bass (*M. s. salmoides*). However, as F1 individuals interbreed over subsequent generations, the qualities present due to heterosis are lost and the resulting population will be inferior to both the F1 cross and the original parental stock (e.g., Burke and Arnold 2001). This outbreeding depression would indeed be a concern in a newly stocked impoundment; however, the study pond was established more than 20 years before this research and any outbreeding depression influences on the population had likely stabilized before this research was initiated.

The most plausible explanation is that destratification increased habitable volume and overall productivity of this pond. Artificial destratification may affect physicochemical function of lentic waters that, in turn, can influence biological productivity. Research has demonstrated destratification effects on thermal and oxygen profiles (e.g., Bryan 1964, Irwin et al. 1966), oxidation and organic matter (e.g., Knoppert et al. 1970), alkalinity and pH dynamics (Leach and Harlin 1970, Malueg et al. 1971, Balangoda 2016), and nutrient dynamics (e.g., Hooper et al. 1953, Ridley et al. 1966, Haynes 1971, Balangoda 2016). Phosphorus is normally the limiting nutrient in southern ponds (Stone et al. 2012), and destratification can certainly influence phosphorus dynamics, although predicting specific effects is quite complicated (Toetz et al. 1972, James 2015, Balangoda 2016). Artificial destratification should promote precipitation and sorption of phosphorus and prevent the release of phosphorus from sediments, but the same aerobic conditions may accelerate phosphorus release by increasing temperature at the sediments and increasing rates of phosphorus turnover and uptake (Toetz et al. 1972).

Biologically, destratifying waterbodies can reduce cyanobacteria (Haynes 1971, Maleug et al. 1971) and encourage growth of beneficial green algae and flagellates (Robinson et al. 1969, Lackey 1971, Sipaúba-Tavares et al. 1999). Fast (1971) found that a measure of productivity (mg C fixed m^{-3} over 4 h) was three times higher in lakes when aerated, but he reported no controls for comparison. Increased production should be transferred up the trophic web. For example, Fast (1971) reported that the total number of benthic invertebrates almost doubled with aeration.

Destratification through physical aeration maintains higher oxygen levels in deeper waters during the summer months. This not only reduces the risk of oxygen-related fish kills, it increases oxygenated deeper habitat to benefit most heterotrophs, including macroinvertebrates and fish (Fast 1966, 1968, 1971). Johnson (1966) demonstrated that destratification increased coho salmon fry survival in natural systems, and his data suggested that biomass production tripled following destratification (Toetz et al. 1972). Aquaculture producers have long understood the value of aeration to fish production (e.g., Boyd 1998, Tucker and Hargreaves 2004), and oxygen is possibly the most important factor affecting fish survival and growth (Brett 1979, Hargreaves and Tomasso 2004). Aeration enables aquaculture to produce much greater biomass of fish than would be possible under natural conditions.

In the current study, biomass of stock-sized largemouth bass tripled following installation of a destratification system. Similar to the Johnson (1966) study, this increase in biomass appears to be largely driven by recruitment. Strong year classes, as indicated by length distributions, recruited each year from 2016 to 2018. This skewed the population towards smaller size classes, and resulted in mean weight of stock size largemouth bass decreasing about 25% when biomass increased. Recruitment may explain why the rapid increase in biomass of stock-size fish occurred 3.5 years after the installation occurred. The system was installed after spawning season in November 2012 when the pond was already destratified, so the first period of summer destratification would have occurred in summer 2013, after the 2013 spawning season. Thus, the first spawning event following full destratification would have occurred in 2014, and these fish would have recruited to the stock-size population by 2016. Further exploration of this lag effect is warranted.

The increase in biomass may have exceeded carrying capacity in this pond. Mean relative weight increased considerably in 2015, suggesting that prey production had begun to increase following destratification, but after density of largemouth bass increased the following year, relative weight began to decline rapidly. Despite substantial harvest (24% of stock-sized biomass) of smaller largemouth bass in 2017, recruitment continued and density remained high. Mean relative weight reached its lowest level of 73.8 in 2018, suggesting that the largemouth bass population is limited by available prey resources. A more liberal harvest strategy may be needed to reduce largemouth bass density and concentrate available prey resources into fewer, yet healthier fish (Schramm and Willis 2012).

This research was conducted in a single pond and replication was not feasible, so results are presented as a case study and caution is warranted when extrapolating these results to other systems and drawing larger conclusions. The results suggested that continuous destratification may increase carrying capacity for largemouth bass in eutrophic southern ponds, but it is unclear what the magnitude might be once the system reaches a new equilibrium. The tripling of stock-size largemouth bass biomass in the current study may have exceeded the increase in carrying capacity, as suggested by declines in relative weight. Regardless, these results emphasize the need for additional research on the effects of destratification on fish communities in small impoundments. If destratification can indeed double or triple standing biomass while greatly reducing the risks of oxygen-related fish kills, the cost of these systems could be more easily justified by managers of ponds and small lakes.

Acknowledgments

The author would like to thank the many graduate and undergraduate students that volunteered their time and labor to sample Blackjack Pond over the course of this study. There are too many to list, but special thanks goes to my graduate students N. Peterson, C. Lloyd, K. Olivieri, C. Fox, J. Bies, B. Haley, and Z. Moran. S. Miranda provided statistical guidance. The destratification system used in this study was designed and donated by S. Cruz of Vertex Water Features, and R. Stewart and D. King of Mossback Fish Habitat donated the artificial habitats.

Literature Cited

Antenucci, J. P., A. Ghadouani, M. A. Burford, and J. R. Romero. 2005. The long-term effect of artificial destratification on phytoplankton species composition in a subtropical reservoir. Freshwater Biology 50:1081–1093.

Balangoda, A. 2016. Artificial destratification effects on nitrogen and phosphorus dynamics in a eutrophic impoundment in the northern Great Plains. Environmental Monitoring and Assessment 188:469–488.

Becker, A., A. Herschel, and C. Wilhelm. 2006. Biological effects of incomplete destratification of a hypertrophic freshwater reservoir. Hydrobiologia 559:85–100.

Bettoli, P. W., M. J. Maceina, R. L. Noble, and R. K. Betsill. 1992. Piscivory in largemouth bass as a function of aquatic vegetation abundance. North American Journal of Fisheries Management 12:509–516.

Bormans, M., B. Marsálek, D. Jancula. 2015. Controlling internal phosphorus loading in lakes by physical methods to reduce cyanobacterial blooms: a review. Aquatic Ecology 50:407–442.

Bortone, S. A., T. Martin, and C. M. Bundrick. 1994. Factors affecting fish assemblage on a modular artificial reef in a north Gulf of Mexico estuary. Bulletin of Marine Science 55:319–332.

Boyd, C.E. 1998. Pond water aeration systems. Department of Fisheries and Allied Aquacultures, Auburn University, Auburn, Alabama 18(1):9–40.

——— and C.A. Boyd. 2012. Physiochemical characteristics of ponds. Pages 49–112 *in* J.W. Neal and D.W. Willis, editors. Small impoundment management in North America. American Fisheries Society, Bethesda, Maryland.

———, E.L. Torrans, and C.S. Tucker. 2018. Dissolved oxygen and aeration in ictalurid catfish aquaculture. Journal of the World Aquaculture Society 49:7–70.

Brett, J.R. 1979. Environmental factors and growth. Pages 599–675 *in* W.S. Hoar, D.J. Randall, and J.R. Brett, editors. Fish physiology. 8th edition. Academic Press, New York.

Bryan, J.G. 1964. Physical control of water quality. British Waterworks Association Journal 46(383):1–19.

Burke, J.M. and M.L. Arnold. 2001. Genetics and the fitness of hybrids. Annual Review of Genetics 35:31–52.

Burkhardt, R.W. and S. Gutreuter. 1995. Improving electrofishing catch consistency by standardizing power. North American Journal of Fisheries Management 15:375–381.

Calhoun, A. and P. Hubbell. 1970. Effects of artificial destratification of bottom organisms in El Capitan Reservoir. Inland Fisheries Branch, California Department of Fish and Game, Fisheries Bulletin 148.

Chapman, D.G. 1951. Some properties of the hypergeometric distribution with applications to zoological sample censuses. University of California Publications in Statistics 1:131–160.

Chapman, E.J. and C.J. Byron. 2018. The flexible application of carrying capacity in ecology. Global Ecology and Conservation 12: available online at: <https://www.sciencedirect.com/science/article/pii/S2351989417301427>. Accessed 4 September 2018.

Fast, A.W. 1966. Artificial destratification of lakes and its significance in fisheries management. California Department of Fish and Game Administrative Report No. 66-16.

———. 1968. Artificial destratification of El Capitan Reservoir by aeration: part 1—effects of chemical and physical parameters. California Department of Fish and Game, Fisheries Bulletin 141.

———. 1971. The effects of artificial aeration on lake ecology. Ph.D. dissertation, Michigan State University, East Lansing.

Hargreaves, J.A. and J.R. Tomasso, Jr. 2004. Environmental biology. Pages 36–68 *in* C.S. Tucker and J.A. Hargreaves, editors. Biology and culture of channel catfish. Elsevier B.V., Amsterdam, Netherlands.

Haynes, R. 1971. Some ecological effects of artificial circulation on a small, eutrophic New Hampshire lake. Ph.D. dissertation. University of New Hampshire, Durham.

Henson, J.C. 1991. Quantitative description and development of a species-specific growth form for largemouth bass, with application to the relative weight index. Master's thesis. Texas A&M University, College Station.

Hooper, F.F., R.C. Ball, and H.A. Tanner. 1953. An experiment in the artificial circulation of a small Michigan lake. Transactions of the American Fisheries Society 82:222–241.

Irwin, E.R., R.L. Noble, and J.R. Jackson. 1997. Distribution of age-0 largemouth bass in relation to shoreline landscape features. North American Journal Fisheries Management 17:882–893.

Irwin, W.H., J.M. Symons, and G.G. Robeck. 1966. Impoundment destratification by mechanical pumping. Pages 251–274 *in* J.M. Symons, editor. Water quality behavior in reservoirs: a compilation of published research papers. Public Health Service Publication No. 1930, Cincinnati, Ohio.

James, W.F. 2015. Managing internal phosphorus loading and vertical entrainment in a weakly stratified eutrophic lake. Lake and Reservoir Management 31:292–205.

Johnson, R.C. 1966. The effect of artificial circulation on production of a thermally stratified lake. Fisheries Research Papers, Washington Department of Fisheries 2(4):5–15.

Knoppert, P.L., J.J. Rook, T. Hofker, and G. Oskam. 1970. Destratification experiments at Rotterdam. Journal of the American Water Works Association 62:448–454.

Lackey, R.Y. 1971. Effects of artificial destratification of a lake ecosystem. Colorado Division of Game, Fish, and Parks, Dingell-Johnson Job Completion Report, Project F-46-R, Job No. 2.

Leach, L.E. and C.C. Harlin, Jr. 1970. Induced aeration of small mountain lakes. National Water Quality Control Research Program, Region VI, Ada Environmental Protection Agency, Water Quality Office Program #16080.

Lehman, J.T. 2014. Understanding the role of induced mixing for management of nuisance algal blooms in an urbanized reservoir. Lake and Reservoir Management 30-63-71.

Losordo, T.M. 1991. Stratification and mixing in shallow aquaculture ponds. Pages 396–435 *in* D.E. Brune and J.R. Tomasso, editors. Aquaculture and water quality. World Aquaculture Society, Baton Rouge, Louisiana.

Maleug, K., J. Tilstra, D. Schults, and C.F. Powers. 1971. The effect of induced aeration upon stratification and eutrophication processes in an Oregon farm pond. International Symposium of Man-Made Lakes, Knoxville, Tennessee. May.

McCann, K., A. Hastings, and G.R. Huxel. 1998. Weak trophic interactions and the balance of nature. Nature 395:794–798.

Miranda, L.E. 2017. Reservoir fish habitat management. Lightening Press, Totowa, New Jersey.

Mostefa, G. and K. Ahmed. 2012. Treatment of water supplies by the technique of dynamic aeration. Procedia Engineering 33:209–214.

Neal, J.W. and R. Kröger. 2012. Sportfish pond fertilization. Pages 245–276 *in* C.C. Mischke, editor. Aquaculture pond fertilization: impact of nutrient impact on production. John Wiley & Sons, Hoboken, New Jersey.

——— and M.C. Lloyd. 2017. Response of fish populations to floating streambed wetlands. Journal of the Southeastern Association of Fish and Wildlife Agencies 5:64–70.

Neumann, R.M., C.S. Guy, and D.W. Willis. 2012. Length, weight, and associated indices. Pages 637–676 *in* A.V. Zale, D.L. Parrish, and T.M. Sutton, editors. Fisheries techniques, 3rd edition. American Fisheries Society, Bethesda, Maryland.

Pardue, G.B. 1973. Production response of the bluegill sunfish, *Lepomis macrochirus*, Rafinesque, to added attachment surface of fishfood organisms. Transactions of the American Fisheries Society 3:622–626.

Polovina, J.J. 1994. Function of artificial reefs. Bulletin of Marine Science 55:13–49.

Ridley, J.E., P. Cooley, and J.A.P. Steel. 1966. Control of thermal stratification in Thames Valley reservoirs. Proceedings of the Society for Water Treatment and Examination 15:225–244.

Robinson, E.L., W.H. Irwin, and J.M. Symons. 1969. Influence of artificial destratification on plankton populations in impoundments. Transactions of the Kentucky Academy of Science 301:1–18.

Robson, D.S. and H.A. Regier. 1964. Sample size in Petersen mark-recapture experiments. Transactions of the American Fisheries Society 93:215–226.

Savino, J.F. and R.A. Stein. 1982. Predator-prey interactions between largemouth bass and bluegills as influenced by simulated, submersed vegetation. Transactions of the American Fisheries Society 111:255–266.

Schramm, H.L. and D.W. Willis. 2012. Assessment and harvest of largemouth bass-bluegill ponds. Pages 181–214 *in* J.W. Neal and D.W. Willis, editors. Small impoundment management in North America. American Fisheries Society, Bethesda, Maryland.

Sipaúba-Tavares, L. H., A. M. Freitas, and F. M. S. Braga. 1999. The use of mechanical aeration and its effects on water mass. Revista Brasileira de Biologia 59(1):33–42.

Stone, N., J. E. Morris, and B. Smith. 2012. Managing the pond environment. Pages 113–154 *in* J. W. Neal and D. W. Willis, editors. Small impoundment management in North America. American Fisheries Society, Bethesda, Maryland.

Toetz, D. W. 1981. Effects of whole lake mixing on water quality and phytoplankton. Water Research 15:1205–1210.

———, J. Wilhm, and R. Summerfelt. 1972. Biological effects of artificial destratification and aeration in lakes and reservoirs—analysis and bibliography. Bureau of Reclamation Report REC-ERC-72-22, U.S. Department of Interior, Denver, Colorado.

Tucker, C. S. and J. A. Hargreaves. 2004. Pond water quality. Pages 215–273 *in* C. S. Tucker and J. A. Hargreaves, editors. Biology and culture of channel catfish. Elsevier B. V., Amsterdam, Netherlands.

Van Dijk, M. and S. J. van Vuuren. 2009. Destratification induced by bubble plumes as a means to reduce evaporation from open impoundments. Water South Africa 35(2):158–167.