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Comparison of Abiotic and Biotic Factors Between Coves of Varying Connection to Harlan County Reservoir, Nebraska

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COMPARISON OF ABIOTIC AND BIOTIC FACTORS BETWEEN COVES OF VARYING CONNECTION TO HARLAN COUNTY RESERVOIR, NEBRASKA

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ABSTRACT

Sediment berms of various heights have developed in the mouths of several coves within Harlan County Reservoir due to a combination of sediment deposition and lateral drift of eroded sediments. These berms can isolate coves from the main reservoir if the berm height is greater than the water elevation of the reservoir. Disconnection of coves may impact water quality, zooplankton, and fish within the isolated reservoir habitats. This study will examine differences in water quality parameters, zooplankton communities, and fish assemblages between disconnected coves, connected coves and the main reservoir.

Water quality parameters related to water clarity and productivity differed between habitat types, with disconnected coves having higher turbidity and relative chlorophyll *a* readings, and lower secchi depths compared to the main reservoir. Connected coves had intermediate values between disconnected coves and the main reservoir. Disconnected coves also had increased densities of zooplankton. Zooplankton assemblages differed between habitats. Both cove types had increased abundance of rotifers within their assemblages, however this was more pronounced in disconnected coves. The main reservoir had increased *Daphnia* spp. compared to disconnected coves. Ordination of the zooplankton assemblages showed high community consistency within the main reservoir, compared to more variability within connected and disconnected coves.

Connected coves had increased species richness and Shannon's diversity compared to disconnected coves. Jaccard's similarity index indicated that habitats were similar in species composition, however their assemblages had low similarity based on Renkonen Similarity Index. Ordination of the fish community showed connected coves had distinct fish assemblages compared to disconnected coves. Disconnected coves had large variability in fish assemblages. White Bass, Freshwater Drum, Channel Catfish, Gizzard Shad, River Carp Sucker, Walleye, Shortnose Gar, Northern Pike, Western Mosquitofish, Largemouth Bass, and Red Shiner were indicator species for connected coves. Disconnected coves had no species that were indicators.

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CHAPTER 1:

INTRODUCTION AND LITERATURE REVIEW

Brian E. Mason

Reservoir Characteristics

The damming of rivers by humans has occurred over several millennia; the first known dam was built in ancient Egypt in 4,000 BCE (Kornijów 2009). As of 2009, approximately 800,000 dams are in operation around the world, and 91,468 (11.4%) are within the United States (Kornijów 2009). Approximately 6,400 (7.0%) U.S. dams are considered large impoundment structures (dams >15 m in height; Namy 2007). The building of dams creates an artificial barrier within riverine systems, and reservoirs are the resulting waterbodies that form within the historic river valley. The majority of reservoirs that are created from the construction of dams provide water for domestic use, hydroelectrical generation, flood control, agricultural irrigation, and recreational benefits (Namy 2007; Kornijów 2009; Zarfl et al. 2015).

Ecological consequences on rivers downstream of dams is well documented. Flows downstream of these barriers are relatively more uniform with reduced high magnitude flow events (Moyle and Mount 2007). Additionally, fine sediments within streambeds are often vulnerable to erosion below dams as the released water have a reduced sediment load, leading to an excess of potential energy expended on the suspension of small particulates (Williams and Wolman 1984; Kondolf 1997). The reduction of high flow events and the release of sediment-lean water often results in increased channelization and bed down-cutting of river systems below dams; thus, rivers downstream become deeper, narrower channels of coarse substrate with limited lateral movement of sediments within the floodplain (Kondolf 1997; Graf 2006; Juracek 2015).

Changes in hydrologic regimes also reduce channel complexity and eliminate backwaters that provide habitat for aquatic organisms (Moyle and Mount 2007; Juracek 2015).

Reservoirs formed above dams may be divided into riverine, transitional, and lacustrine zones from the upstream to downstream ends based on longitudinal changes in water velocity (Figure 1). This zonation is a result of subsequent changes in many physical and biological characteristics (Thornton et al. 1981; Kimmel et al. 1990; Thornton 1990; Olds et al. 2011). The location where inflow meets standing water is called the riverine zone which typically has high turbidity, coarse sediment deposition, high levels of available nutrient and allochthonous production (Thornton et al. 1981; Kimmel et al. 1990). As the inflowing water loses velocity, finer sediments are then deposited and the turbidity decreases, while dissolved nutrients remain (Kimmel et al. 1990). This combination of increased light penetration and nutrient load can result in the transitional zone having high primary productivity (Bernot et al. 2004; Thornton et al. 1981). The lacustrine zone lies downstream of the transitional zone where water velocity is the slowest, resulting in the deposition of fine sediment and clays, further reducing water turbidity (Thornton et al. 1981; Kimmel et al. 1990).

In addition to longitudinal gradients within reservoirs, stratification by water depth can also occur within reservoirs during the summer months due to vertical differences in water temperatures and resulting changes in water densities (Kennedy and Walker 1990; Ashby and Kennedy 1993). During these months, high sediment and nutrient accumulation within some reservoirs can increase nutrient loading and primary productivity, potentially promoting hypoxic conditions within the reservoirs hypolimnion

(Patton and Lyday 2008; Lucchesi et al. 2019). Stratification, however, does not typically occur in reservoirs with shallow depths and that experience frequent mixing from strong winds, such as Harlan County Reservoir in Nebraska (Koupal and Peterson 2008, Maline et al. 2011).

Biotic communities within reservoirs may differ substantially compared to those of un-impounded rivers. Dams create barriers for movement of aquatic organisms, which may limit movement to feeding and spawning grounds and reduce genetic exchange between subpopulations (Larinier 2000; Li et al. 2013). Subsequent reductions in recruitment and recolonization rates may lead to declines and local extirpations of native species (Larinier 2000; Kornijów 2009). Changing from a lotic environment to one resembling a lacustrine system may result in a species composition shift due to changing habitat; riverine species may be lost while lacustrine species thrive (Benson 1982; Li et al. 2013; Juracek 2015). In addition to fish community changes caused by modifying the original riverine habitat, desirable fish species are often introduced, and their populations supported by reservoirs managers. Due to this management, reservoirs are often sources of non-native species that originate due to stocking programs, designed to increase angling opportunity, or by accidental introduction (Martinez et al. 1994; Miranda and Bettoli 2010; Havel et al. 2015). Dams can also impact zooplankton communities of impounded river systems. Reservoirs with top release dams been found to act as sources of zooplankton for downstream river segments, especially during dry seasons or droughts when river backwaters are unavailable (Zhao et al. 2017). Conversely, bottom release dams can act as a barrier to zooplankton movement downstream, as the majority of zooplankton biomass occurs near the top of the water column and would avoid entrainment (Zhao et al. 2017).

Aging Process of Reservoirs and Resulting Cove Isolation

Both reservoirs and natural lakes undergo aging processes as these water bodies undergo succession, but the rate of aging may be more rapid in reservoir systems compared to natural lakes (Wetzel 1990). Reservoirs often have larger watersheds compared to natural lakes and are strategically placed on the landscape at locations to maximize water capture potential and holding capacity (Benson 1982; Kimmel and Groeger 1986). The majority of reservoirs within the United States are located between 30° and 41° latitude where few natural lakes occur and soils in many of the associated ecoregions are often highly erodible to wind and water interactions (Canfield and Bachmann1981; Kennedy and Walker 1990). Increased erosion and sedimentation rates can further increase reservoir turbidity, reduce water storage capacity, degrade and homogenize littoral habitat, and reduce abundance and diversity of benthic invertebrates (Muncy et al. 1979; Miranda 2017). Irrigation or flood-control reservoirs may experience high levels of intra- and inter-annual variations in water levels, thus exacerbating erosion rates (Miranda 2017). Because of these continuous aging processes, reservoirs may experience changes over time from their original form or desired purpose.

Reservoir construction in the United States increased substantially during the early to mid-1900s, but rates of new impoundment construction have slowed between the 2000s to the present. The median age of U.S dams reported in 2016 was 66 years

(Miranda 2017). Most dams have a designed life span of 50 – 100 years (Juracek 2015), thus, the majority of reservoirs in the United States are approaching or are beyond their intended life expectancy. Reservoir chronological age, however, does not necessarily correlate to the degree of senescence (Miranda and Krogman 2015). The rate in which individual reservoirs experience the effects of aging is highly variable and dependent on factors such as climate, geography, reservoir morphology, and watershed land cover (Miranda and Krogman 2015). According to a study by Miranda and Krogman (2015) examining rates of reservoir aging, impoundments in areas with a high percentage of cultivated land tended to have higher function age scores compared to other parts of the country. The increase in aging rate within areas of heavy agriculture land use, such as the Great Plains region, is likely due to increased sediment runoff (Hargrove et al. 2010). As more impoundments reach their expected functional lifespan, management of reservoir habitats become increasingly focused on reducing erosion, removing existing sediments, and preventing future sedimentation (Miranda and Krogman 2015; Miranda 2017).

One feature of reservoirs that could be impacted by the reservoir aging process is coves. Coves are common within many reservoirs, often formed by the back filling of water in steeper banked valleys created by former inflowing tributaries and intermittent drainages to the historic river valley (Miranda et al. 2014). Coves offer valuable habitat for reservoirs and may differ markedly in their physical, chemical, or biological attributes compared to the main reservoir (Meals and Miranda 1991). The calm waters found in coves can support the establishment of aquatic macrophytes when water levels are relatively stable (Marsh and Langhorst 1988; Slipke et al. 2005; Dagel and Miranda

2012). Coves and ephemeral backwaters have also been shown to harbor unique zooplankton assemblages with higher species richness compared to the main reservoir (Marsh and Langhorst 1988), likely supported, in part, by the increased aquatic macrophyte coverage compared to the main lake (Bergström et al. 2000; Geraldes and Boavida 2004). Fish assemblages have also been found to differ between main reservoir and cove environments, and many fish species use coves for spawning and nursery habitat (Copp and Penaz 1988; Meals and Miranda 1991, Nicolas and Pont 1997; Slipke et al. 2005).

Coves can become disconnected from the main reservoir over time by the accumulation of sediment berms formed at the opening between these two areas (Marsh and Langhorst 1988; Mueller 1995; Slipke et al. 2005; Slipke and Maceina 2007; Patton and Lyday 2008; Miranda 2017). The process of berm development may differ between different reservoir systems. Coves in relatively narrow, fast-moving reservoirs with high sediment loads are more prone to sediment backfill due to particles settling from suspension within the cove (Slipke et al. 2005; Slipke and Maceina 2007). This type of cove isolation has been observed in Demopolis Reservoir, Alabama (Slipke et al. 2005; Slipke and Maceina 2007; Figure 2). Larger reservoirs with heavy wind and wave action and highly erodible soils may form spits of eroded shoreline sediments that develop across the mouth of small bays and coves (Marsh and Langhorst 1988; Mueller 1995). Spit formation is commonly observed in ocean sediments distributed by currents, wave energy, and tidal forces (Evans 1942; Hoyt 1967). Although not as commonly observed in inland reservoirs, spit formations have been documented in Lake Mohave, Nevada –

Arizona (Marsh and Langhorst 1988; Mueller 1995; Figure 3). Isolated habitats can also be created by the formation of deltas at the confluence of streams within reservoirs, however these habitats differ from coves as they were once a part of the historic reservoir basin, cut off over time by sediment levees (Kaemingk et al. 2007; Patton and Lyday 2008; Figure 4).

Defining cove disconnection can be difficult and is likely dependent on the particular focus of a given study. The interaction between water elevations in the reservoir relative to the height of the isolating sediment berm could influence surface water connection in coves, depending on depth of connection (e.g., few centimeters to several meters). A cove isolated by a sediment berm could also exchange water with the main reservoir via groundwater connection (Marsh and Langhorst 1988; Mueller 1995). Different organisms (e.g., zooplankton v. fish) will require different water depths over the berm in order to move between the main reservoir and the cove. Slipke et al. (2005) based their determination of connectivity on the ability to have boat access throughout the year. By definition, Slipke et al. (2005) included coves with shallow surface water connection as "disconnected", but the exchange of aquatic organisms between the two systems may still have been possible. Using that same definition, Slipke and Maceina (2007) found similar movement rates of Largemouth Bass (*Micropterus salmoides*) and White Crappie (*Pomoxis annularis*) in and out of a "nearly disconnected" cove (0.3 m approximate connecting depth) compared to a "connected cove". Thus, defining disconnection between coves may need information on the depths needed by the organisms of interest rather than depths required for recreational use.

Potential Abiotic and Biotic Effects of Cove Disconnection

Little research has been conducted on the effects of cove disconnection within reservoirs (Miranda 2017), leading to difficulty in understanding of changes associated with cove isolation. Coastal lagoon habitats have been shown to have higher water elevation and unique water quality characteristics compared to the nearby ocean environment due to limited percolation through isolating sediment berms (Gordon. 1991; Hanslow et al. 2000). Within reservoirs, however, similar hydrological and water quality patterns have been observed between disconnected coves and the main reservoir, suggesting some level of water connectivity occurs between the separating sediment berm (Marsh and Langhorst 1988; Mueller 1995). Similarities in water quality between the main reservoir and disconnected coves is likely dependent on the permeability of the sediment, the length of time of disconnection, and the extent of surface water exchange over the berm. Although some subsurface connection is possible, surface disconnection of coves could lead to changes in water quality within the cove. For example, water temperature of disconnected coves on Lake Mohave were slightly lower during the winter and slightly elevated during the summer compared to the parent reservoir (Marsh and Langhorst 1988). Additionally, disconnected coves in Demopolis Reservoir, Alabama, were found to be warmer during spring than coves with surface connection to the main reservoir (Maceina and Slipke 2003; Slipke et al. 2005). Water temperatures in coves may vary more between seasons than in main reservoirs due to the shallower depths.

Turbidity within coves is likely highly variable as well, as many factors affect water turbidity. Reduced landscape runoff, wind protection, and submergent aquatic vegetation have been associated with reduced turbidity within a cove (Scheffer 1997; Hargreaves 1999). If these factors are present within coves, a positive feedback for reduced turbidity could be established, whereby clearer water would allow for increased vegetation, further stabilizing the substrate and allowing for continued vegetation growth (Scheffer 1997). The opposite could also occur in coves susceptible to factors leading to higher turbidity. A negative feedback may occur with increased runoff and sediment disturbance from wind, reducing vegetation growth and leading to increased turbidity (Scheffer 1997). Additionally, the biotic community may increase turbidity in coves. For example, benthivorous fish such as common carp (*Cyprinus carpio*) may roil sediments and decrease aquatic macrophyte establishment through their feeding and reproductive behaviors (Zambrano et al. 2001). Turbidity within disconnected coves may also be influenced by isolation over time. Turbidity in oxbow lakes of the Mississippi River has been shown to increase as the time of disconnection increases, likely due to agricultural runoff and wind-driven sediment resuspension (Miranda 2005; Knight et al. 2008). Similar processes to those found in oxbow lakes are likely to occur within shallow, disconnected reservoir coves that receive runoff (Slipke et al. 2005; Miranda et al. 2014). Connected coves may be less susceptible in comparison due to more frequent flushing events of suspended particles to the main reservoir.

Primary productivity within coves is likely variable and dependent on nutrient availability, surrounding land use, and sediment resuspension. Disconnected coves have been noted to have higher concentrations of available chlorophyll *a* than connected coves in Demopolis Reservoir, Alabama (Maceina and Slipke 2003; Slipke et al. 2005).

Additionally, coves influenced heavily by agricultural land use, can receive inputs of phosphorus and nitrogen that can increase phytoplankton production (Hillbricht-Ilkowska 1993; Izydorczyk et al. 2008). Izydorczyk et al. (2008) found that wind-sheltered coves fed by streams with high nutrient run-off had increased algal growth that eventually led to large main reservoir algal blooms. Shallow coves may also have increased primary production due to the resuspension of sediments by wind, increasing available nutrients that were previously sequestered in the substrate (Mosley 2015; Abirhire et al. 2019).

The difference in primary productivity between connected and disconnected coves would likely affect dissolved oxygen, as differential rates of respiration and decay in productive conditions leads to hypoxic conditions (Kimmel et al. 1990; Ashby and Kennedy1993; Lucchesi et al. 2019). Higher coverage of aquatic vegetation within disconnected coves (Maceina and Slipke 2003; Slipke et al. 2005) may reduce dissolved oxygen concentrations, especially following plant senescence (Miranda and Hodges 2000). Additionally, collapses of algal colonies during summer and declines in photosynthetic activity in winter due to snow and ice conditions may lower dissolved oxygen levels further during these time periods (Stefan and Fang 1997; Posch et al. 2012). As disconnected coves are typically smaller enclosed systems, they may be more susceptible to variations in dissolved oxygen compared to the main reservoir. Even temporary reductions of dissolved oxygen could be problematic for biotic communities within disconnected coves, as the sediment berm would restrict or eliminate the ability to find refugia and avoid the hypoxic water.

Difference in water quality between the main reservoir, connected coves, and disconnected coves may impact zooplankton populations. *Daphnia* spp. are known to be intolerant to suspended solids and high water turbidity (Arruda 1983; McCabe and O'Brien 1983), but other species such as rotifers are more tolerant of turbid waters (Bernot et al. 2004). Further, the lack of *Daphnia* spp. may support higher rotifer densities due to decreases in competition (Wolfinbarger 1999). These differences indicate that turbid coves would harbor higher abundances of rotifers while assemblage in the less turbid main reservoir would have increased abundances of *Daphnia* spp. Differences in the amount of aquatic vegetation between the main reservoirs, connected coves, and disconnected coves may also lead to differences in zooplankton communities as higher densities of aquatic vegetation have been found to harbor distinct zooplankton assemblages. For example, abundance of *Chydorus* and *Cyclopoida* species appear to be higher in vegetated areas (Geraldes and Boavida 2004; Olson et al. 2004). These species would likely have increased abundance within disconnected coves where higher densities of aquatic vegetation have been reported (Slipke et al. 2005). Furthermore, the higher primary productivity of disconnected coves (Maciena and Slipke 2003; Slipke et al. 2005) would support a higher abundance of total zooplankton (Canfield and Watkins 1984; Dodson 1992; Canfield and Jones 1996; Shuter and Ing 1997).

Differences in water quality between connected and disconnected coves may also favor more tolerant and less diverse fish communities. Disconnected coves of Demopolis Reservoir, Alabama, have lower species richness of larval and juvenile fish compared to connected coves and the reservoir mainstem (Slipke et al. 2005). Furthermore, disconnected coves and oxbow lakes with reduced water quality have been shown previously to support smaller, more tolerant centrarchid species (Slipke and Maciena 2005; Miranda et al. 2005). Reduced fish species richness could also be related to the lack of reintroduction of lacustrine fish species, as the absence of surface water connection can restrict movement of fish between disconnected coves and the main reservoir (Marsh and Langhorst 1988; Mueller 1995). Disconnected coves had significantly lower catch per unit effort of Largemouth Bass, crappies (*Pomoxis* spp.), and Threadfin Shad (*Dorosoma petenense*) compared to connected coves in Demopolis Reservoir, Alabama, indicating that connectivity could be important for continued persistence of these species within coves (Slipke and Maceina 2005). The lack of reintroduction could further reduce the diversity and species richness of disconnected coves, potentially resulting in an assemblage of species better adapted to isolated environments. Additionally, degradation and loss of coves could be detrimental to population sustainability of many fish species within the main reservoir, because of the importance of these habitats for their reproduction, growth, and survival (Meals and Miranda 1991).

Study Area

Harlan County Reservoir was constructed between 1946 and 1952 by the U.S. Army Corps of Engineers. The reservoir is located adjacent to the towns of Republican City and Alma, Nebraska (Figure 5). The primary purposes of the reservoir were to control flooding along the Republican River and provide water for irrigation (USACE 1991). The dam impounding the Republican River is constructed of earthen materials and stands approximately 32 m above the streambed (USACE 1991). The surface area of Harlan County Reservoir is approximately 5,400 ha, with a total holding capacity of over one billion m^3 (USACE 2011). Water elevations within the reservoir vary annually and seasonally due to river inflows and irrigation demands (Figure 6), but typically increase in the spring due to local precipitation and snowmelt, decrease during the summer as irrigation needs increase, and remain low throughout the fall and winter as the limited precipitation is locked as snow until spring (Diffendal et al. 2002). The reservoir and the surrounding area also provide recreational benefits such as fishing, hunting, camping, wildlife watching, boating, and water sports (Diffendal et al. 2002).

Sediment berms of various heights have developed in the mouths of several coves within Harlan County Reservoir due to a combination of sediment deposition and lateral drift of eroded sediments. Some berms have minimal sediment accumulation and are often submerged during median mid-summer reservoir water elevation (e.g., Bone and Prairie Dog; Figure 7); others are as high as or higher than these same water elevations (e.g., Methodist, Tipover and Indian Hill; Figure 7). Because of the high variation of water elevation within Harlan County Reservoir, it is difficult to define cove disconnection to the main reservoir. Shorter berms increase the likelihood for connection to the main reservoir within and between years (Figures 6 and 7). The highest berms, however, reduce or eliminate surface water exchange between the two areas for longer periods of time (Figures 6 and 7). Routine dredging for navigational purposes has allowed Patterson and Gremlin coves to maintain consistent connectivity over time, but no actions have been undertaken to maintain connections between other coves and the main reservoir (USACE 1991; Diffendal et al. 2002). Groundwater connection likely occurs to varying degrees through sediment berms as water is exchanged through the porous sediment (Marsh and Langhorst 1988; Mueller 1995). Although restricting biotic movement, groundwater exchange between the main reservoirs could potentially influence the water quality in disconnected coves, depending on the cove's hydrology. For the purpose of my study, disconnected coves are defined as coves having <1 m of water connection at the berm saddle point (i.e., the lowest point atop the berm crest; Hanslow et al. 2000) at any time throughout the study. All other coves are classified as connected. Depth of connection is based on measurements of the sediment berms for each cove, taken by Flatwater Group, Incorporated in 2017 (Figure 7), compared to the main reservoir water elevation recorded on the dam spillway (Figure 6). Connection was continually examined for every cove during the timeframe of this study; 2017 and 2018, however, all study coves exhibited no change from their original designation. Connected coves included Bone, Gremlin, Patterson, and Prairie Dog coves, while disconnected coves included Indian Hill, Methodist, and Tipover coves. Assuming the sediment berm heights of disconnected coves remained similar since last connected, prior to the beginning of the study, Indian Hill, Methodist, and Tipover coves had been disconnected from the main reservoir (<1 m depth of surface water connection) since September, 1993, February, 2012, and June, 2012 respectively (USBOR 2020; Figure 6).

Study Objectives and Hypothesis

1) Determine differences in water quality (i.e., water temperature, dissolved oxygen, pH, available chlorophyll *a*, turbidity, Secchi depth, dissolved nitrates,

and dissolved phosphates) between the main reservoir, connected coves, and disconnected coves of Harlan County Reservoir.

H1) I hypothesize that water quality of disconnected coves will differ within and between years compared to the main reservoir and to connected coves. More specifically, I hypothesize that disconnected coves will have lower temperatures in the spring and fall and higher temperatures during the summer; lower dissolved oxygen and pH in all seasons; higher available chlorophyll *a* and turbidity in all seasons, with a peak during the summer; lower Secchi depth in all seasons, with the lowest readings during the summer; and higher dissolved nitrates and phosphates in all seasons in comparison to connected coves and the main reservoir. Additionally, I hypothesize that water quality in the main reservoir will be more similar to connected coves than disconnected coves, as connected coves likely have higher amounts of water exchange and thus are more influenced by the main reservoir.

2) Determine differences in zooplankton densities and taxa assemblages between the main reservoir, connected coves, and disconnected coves of Harlan County Reservoir.

H2) I hypothesize that disconnected coves will have higher total densities of zooplankton than the main reservoir or connected coves. Further, disconnected coves will support unique zooplankton taxa not found in the main reservoir or connected coves. Specifically, disconnected coves will have lower densities of

Daphnia spp. and higher densities of rotifers, Bosminidae, Cydoridae, and Ceriodaphnia, compared to the main reservoir and connected coves throughout all seasons, and zooplankton densities in all habitat types will be highest during the summer. Additionally, I hypothesize that densities and assemblages will be more similar between the main reservoir and connected coves than between the main reservoir and disconnected coves, as there is potential for biotic exchange and likely more similar water quality characteristics.

3) Determine if fish communities differ between connected and disconnected coves of Harlan County Reservoir.

H3) I hypothesize that disconnected coves will have fewer species and lower diversity than connected coves. Specifically, fish communities in disconnected coves will include a higher proportion of tolerant, generalist species, such as Common Carp (*Cyprinus carpio*), Orangespotted Sunfish (*Lepomis humilis*) and Black Bullhead (*Ameiurus melas*).

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Figures

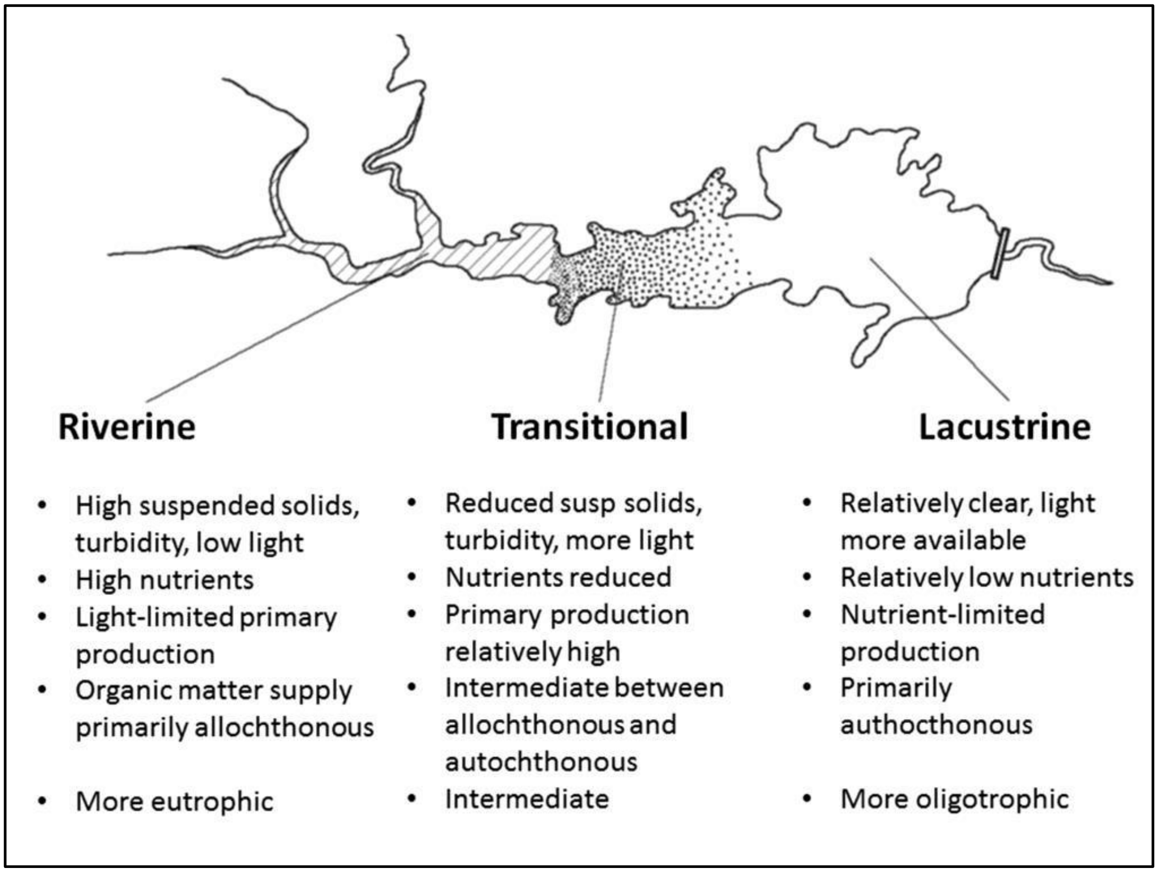

Figure 1. Representation of the riverine, transitional, and lacustrine zones in a typical reservoir (adapted from Kimmel et al. 1990).

Figure 2. Depiction of a disconnected cove (Knee Deep Cove) and a connected cove (Two Dock Cove) within Demopolis Reservoir, Alabama, due to sediment backfill into the cove mouth (Photo Credit: Google Maps 2020).

Figure 3. Disconnection of Yuma Cove within Lake Mohave, Arizona, by the formation of a sediment spit due to shoreline erosion and sediment drift (Mueller 1995; Photo Credit: Google Maps 2019).

Figure 4. Fragmentation within the riverine zone of Lake Texoma, Oklahoma, due to excessive sediment buildup and the formation of natural levees adjacent to the flowing streambed. The solid line indicates the original river. The red arrows indicate areas disconnected from the main reservoir due to the sediment buildup. The yellow arrows indicate areas that are frequently disconnected from the main reservoir. The dashed line indicates the current thalweg of Washita River entering the reservoir, confined by sediment levees on each side (adapted from Miranda 2017).

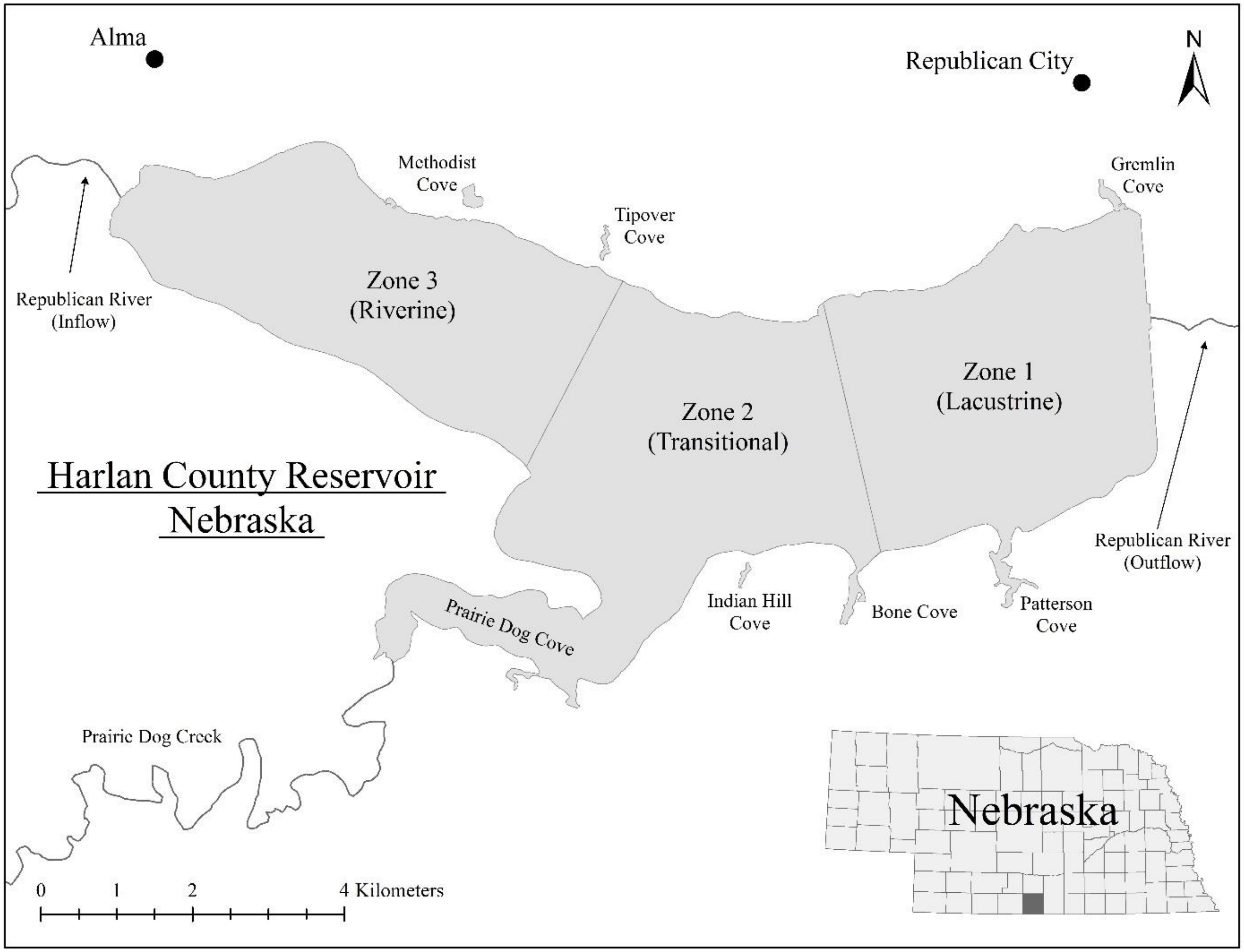
Alma
Republican City
N
Methodist Cove
Tipover Cove
Gremlin Cove
Republican River (Inflow)
Zone 3 (Riverine)
Zone 2 (Transitional)
Zone 1 (Lacustrine)
Republican River (Outflow)
Harlan County Reservoir Nebraska
Prairie Dog Cove
Indian Hill Cove
Bone Cove
Patterson Cove
Prairie Dog Creek
0 1 2 4 Kilometers
Nebraska

Figure 5. Map of Harlan County Reservoir (adapted from aerial imagery taken by the USDA-NRCS July 13, 2016; surface water elevation approximately 591 MASL) and its surrounding cities. Reservoir zones represent the transition seen in reservoirs from being primarily a riverine environment to primarily lacustrine and the zones for Harlan County Reservoir were previously established by Peterson et al. (2005).

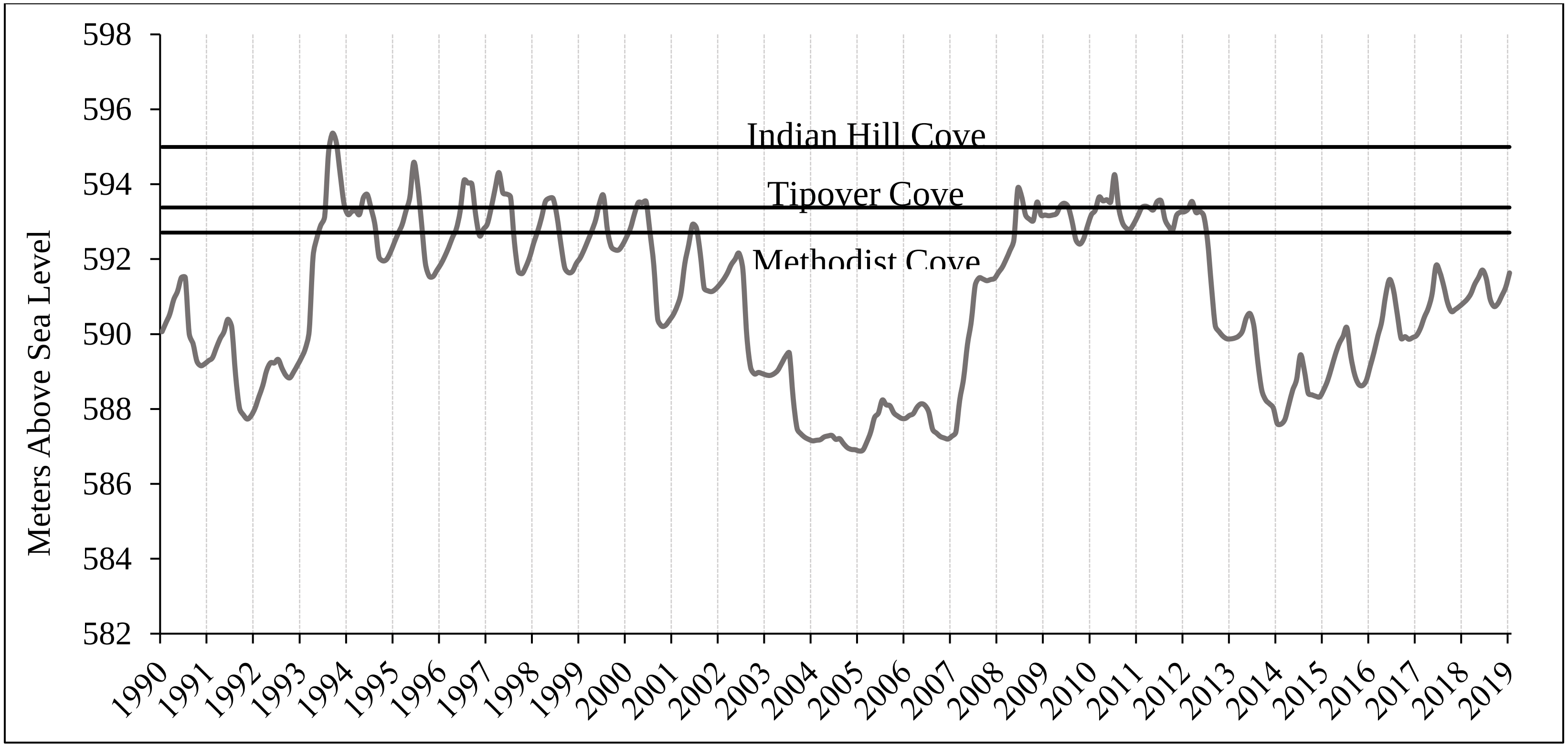

Figure 6. Elevation of the sediment berms disconnecting coves from the main reservoir, compared to end of month water level elevation of Harlan County Reservoir since January 1990. The grey line indicates the water elevation of the main reservoir, recorded at the dam spillway (USBOR 2020). Horizontal black lines indicate minimum water level required for connection (heights of sediment berm plus 1 meter) for each corresponding cove. Vertical dotted lines indicate January 1st of the year labeled below.

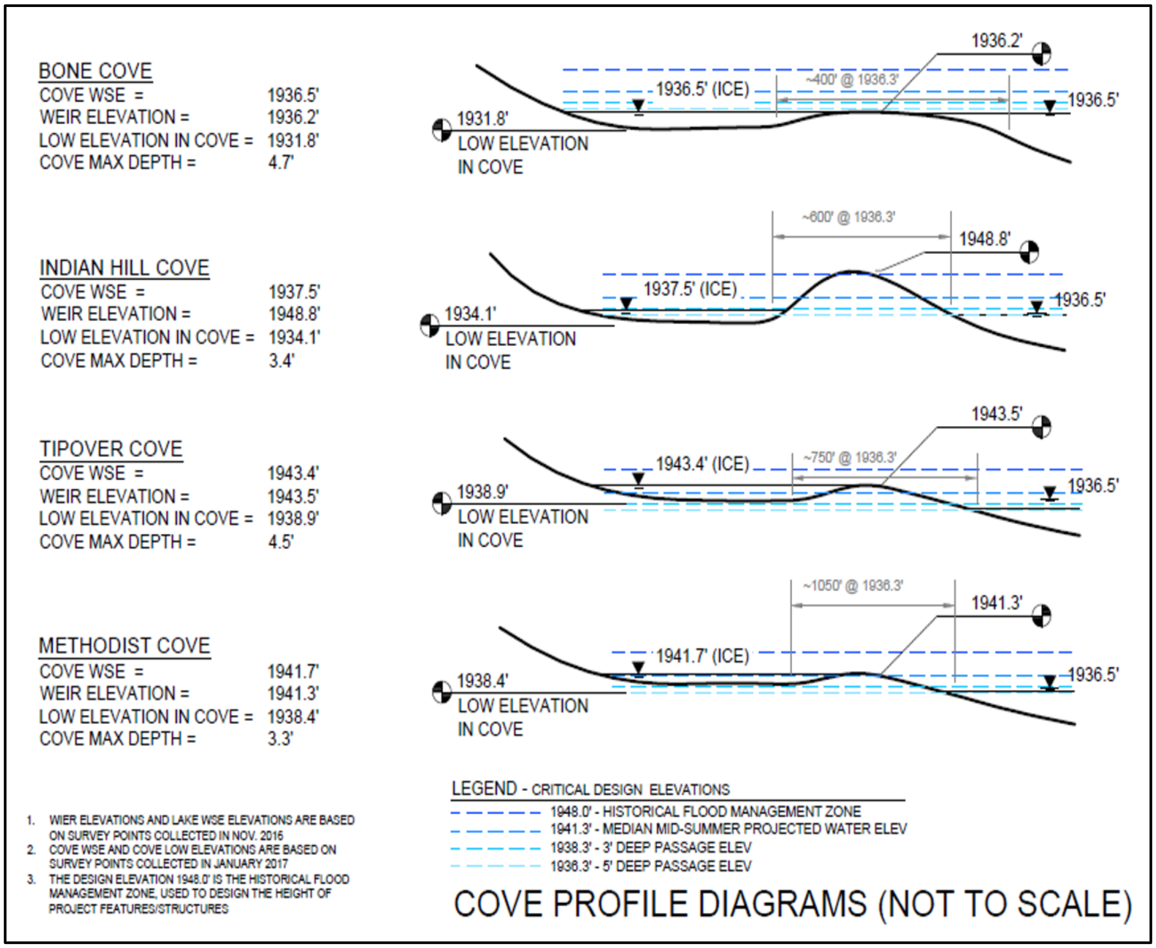

Figure 7. Elevation and water level profiles of several coves and the sediment berm disconnecting them from the main reservoir. Figure does not include Gremlin Cove and Patterson Cove that have been historically dredged to remove berms, or Prairie Dog Cove that has not developed a significant berm due to its large size and protection from erosion. Surveys conducted and image produced by Flatwater Group Inc., 2017.

CHAPTER 2:

COMPARISON OF WATER QUALITY PARAMETERS AND ZOOPLANKTON COMMUNITIES BETWEEN COVES OF VARYING CONNECTION TO HARLAN COUNTY RESERVOIR, NEBRASKA

Brian E. Mason

Introduction

Reservoir shorelines are comprised of many habitat types, including barren beaches, backwaters of parental river valleys, and coves. Reservoir coves typically are formed by the flooding of former tributaries when water levels are elevated following dam construction (Miranda et al. 2014). Coves are common within reservoir systems and often make up much of their shallow littoral area (Miranda et al. 2014). These habitats may be important to reservoir fishery production as they can support more abundant and species rich fish communities compared to the main reservoir (Gido and Matthews 2000). Coves can also act as economic centers of reservoirs, often accommodating marinas, beaches, camping areas, and boating access within their protected waters.

Although coves are part of their larger reservoir, these areas often foster unique conditions regarding water quality, primary productivity, and lower trophic productivity, thus making them ecologically distinct. Cove environments have been found to differ from main reservoirs in a number of ways, including shallower depths and increased variation in temperature (Marsh and Langhorst 1988 Slipke and, Maceina 2005; Slipke et al. 2005); increased nitrogen, phosphorous and chlorophyll *a* concentration (MDNR 2017); increased aquatic vegetation density (Ferrer-Montano and Dibble 2002); and greater abundance of woody debris and detritus (Matthews 1998). Differences in water quality and productivity in coves may subsequently promote more diverse and abundant zooplankton communities compared to main reservoirs (O'Brien and de Noyelles 1974; Blancher1984; Canfield and Jones 1996). Further, higher densities of submergent and emergent vegetation may provide microhabitat for some genera such as *Chydoridae,* *Ceriodaphnia*, and *Bosmina* (Bergström et al. 2000; Geraldes and Boavida 2004). To date, coves are often understudied compared to mainstem reservoirs, despite their documented and hypothesized uniqueness compared to open water habitat (Matthews 1998).

As reservoirs age, coves can become disconnected from the parent reservoir due to sediment accumulation within the cove mouth. Cove isolation can occur by lateral drift of shoreline sediments (Marsh and Langhorst 1988; Mueller 1995) or by backfill of sediments from the main lake (Slipke et al. 2005; Slipke and Maceina 2007). These processes may work independently or in concert to form a sediment berm that separates the cove from the main reservoir, potentially reducing exchange of water between the two water bodies to high water events or groundwater infiltration (Slipke and Maceina. 2005). This disconnection could lead to differences in water quality within the separated habitats, which could influence the biotic community of the waterbody. If water quality is unique within disconnected coves, zooplankton assemblages within could be dominated by taxa more tolerant to the specific water parameters. In addition, disconnection of the coves could limit fish and zooplankton movement between coves and the main reservoir (Slipke and Maceina. 2005; Slipke et al. 2005). Changes in water quality and zooplankton assemblages could alter fish population and community dynamics, as a number of species use reservoir coves for feeding, reproductive, and nursery habitat (Meals and Miranda 1991; Slipke et al. 2005).

Harlan County Reservoir, Nebraska, has numerous coves, several of which have developed sediment berms of various heights over time. Some coves remain continuously connected to the main reservoir due to regular dredging of sediments or natural processes; other coves may be disconnected within or between years, depending on reservoir elevations (Chapter 1, Figures 6 and 7). The length of disconnection time varies by cove, and some coves have been functionally disconnected for several years. Water quality parameters and zooplankton assemblages within the main waterbody of Harlan County Reservoir are well documented (Peterson et al. 2005; Maline et al. 2011; Olds et al. 2011; Olds et al. 2014), however, little attention has been given to cove habitats within the reservoir. Additionally, little research has documented differences in abiotic factors and biotic communities of coves with varying temporal connection to the mainstem reservoir (Miranda 2017). The objective of this study is to compare water quality parameters and zooplankton assemblages between disconnected and connected coves and the mainstem of Harlan County Reservoir. Differences between habitat types were also compared seasonally, during the spring, summer, and fall.

Methods

Harlan County Reservoir (surface area = 5,400 ha at full conservation pool; storage capacity ~ 1 billion m^3) is located on the Republican River near the Nebraska and Kansas border (Figure 1). Dam construction was completed in 1952, and the dam is operated by the U.S. Army Corps of Engineers primarily for flood control and irrigation purposes (USACE 2011). Mean depth of the reservoir at full conservation pool is 4 m and maximum depth is 18 m (Uphoff et al. 2013), however, water elevation can vary up to 3 m on an annual basis (Diffendal et al. 2002).

Sampling took place during spring (May), summer (July), and fall (September/October) in 2017 and 2018. Sampling locations within the mainstem reservoir were standardized and consistent with sites established as part of a long-term effort to capture the spatial and temporal variability of water quality and zooplankton assemblages (see Peterson et al. 2005; Olds et al. 2011; Figure 1). In addition, four locations within each of seven coves (Bone, Gremlin, Indian Hill, Methodist, Patterson, Prairie Dog, and Tipover coves) were randomly selected and standardized across both years (Figure 2). Coves were selected based on their connection status (i.e., connected v. disconnected) and potential candidacy for future renovation projects to improve connectivity to the main reservoir (Keith Koupal, *personal communication*, June 2020). Coves classified as disconnected had <1 m of water connection at the berm saddle point (i.e., the lowest point atop the berm crest; Hanslow et al. 2000) at any time throughout the study, and included Indian Hill, Methodist, and Tipover coves. All other coves were classified as connected.

A suite of water quality metrics were measured at all sampling sites following protocols established by Olds et al. (2011). Water temperature (°C) and dissolved oxygen (mg/L) were measured in 1-m increments throughout the entire water column using a YSI Pro 20-® probe. Because Harlan County Reservoir does not maintain thermal stratification (Olds et al. 2011), temperature and dissolved oxygen readings were averaged among all depths at a given site for each sampling event. Water transparency was indexed using Secchi depth (cm). Remaining water quality parameters were measured from an integrated water column sample collected from a Van Dorn bottle sampler. Water samples were collected at 3-m increments, starting at 1 m below the surface. Relative chlorophyll *a* (relative florescence units; RFU) was measured using a Turner Design AquaFluor Model 8000-010 Fluorometer, and pH was measured using an Oakton-® series 11 pH meter. Turbidity (FAU), dissolved nitrates (mg/L), and dissolved phosphates (mg/L) were measured using a Hach Model DR/870 colorimeter®. Dissolved nitrates (mg/L) and dissolved phosphates (mg/L) were only measured in 2018. Prior to conducting tests for both dissolved nitrates and phosphates using the appropriate Hach® accuvac ampules, subsamples of water were filtered using a 1-µm syringe filter to eliminate suspended particles to improve accuracy.

Zooplankton were collected in tandem with water quality at all cove and main reservoir sites during both years. A circular framed (0.5-m diameter) Wisconsin net (80-µm mesh) was lowered to the bottom of the water column and pulled vertically to the surface. Captured zooplankton were stored in 95% ethanol and transported to the laboratory at the University of Nebraska at Kearney. Samples were diluted with tap water to a known volume and mixed to suspend zooplankton. Four, 1-mL subsamples were drawn using a Hensen-Stempel pipette and placed on a Ward (1955) counting wheel for identification and enumeration by taxa group; taxa groupings were consistent with previously published work for Harlan County Reservoir (Maline et al. 2011; Olds et al. 2014; Table 1). All counts were averaged between the four subsamples. Tow depth and net circumference were used to calculate the volume of water filtered (L) for each sample. Densities of all taxa combined and for each taxonomic group were calculated by dividing the average count per volume of water sampled for each site (number/L).

Data Analysis –

General linear mixed models (SAS Version 9.4) were used to evaluate whether differences existed in water quality parameters and total zooplankton density between connected and disconnected coves and the mainstem reservoir across seasons. Water parameter and total zooplankton density values for the four sample sites within each cove or reservoir zone were averaged to generate a mean value for each study location for every sampling period. Variables were tested for normality using a Shapiro–Wilk test, and those that were non-normally distributed were $\log_{10}$ transformed. Data that failed to normalize following transformation was analyzed using a Kruskal-Wallis test. If differences were noted between connected and disconnected coves and the mainstem reservoir, a follow-up Tukey's test was used to determine which habitat type or types (i.e., connected coves disconnected coves or the main reservoir) differed. The significance level was set at $\alpha = 0.10$.

Non-metric multidimensional scaling (NMDS) using Bray-Curtis distance metrics was used to visualize differences in zooplankton taxa assemblage between connected and disconnected coves and the mainstem reservoir. A stress of < 0.2 is considered suitable for interpreting ecological patterns (Clarke 1993) so the lowest number of axes with the stress of < 0.2 was chosen for the final plot. Spearman's rank correlation coefficients were calculated between taxa abundance and axis values for each site. Taxa groups with relatively strong associations with the axis values ($r_s > 0.40$ or $r_s < -0.40$; Gido et al. 2009) were used to interpret ordinal distribution patterns corresponding to zooplankton assemblage within disconnected coves, connected coves, and the main reservoir.

Results

Mean water temperatures differed between habitat types ($F = 2.79$; $p = 0.07$). Disconnected coves were warmer than the main reservoir, but temperatures were similar between disconnected and connected coves and between connected coves and the main reservoir (Figure 3). Differences between habitat type were not observed within seasons ($F = 58.11$; $p = 0.93$). Average temperatures were cooler in 2017 [19.9 ± 0.9°C (one standard error)] compared to 2018 (21.9 ± 0.7°C; $F = 5.38$; $p = 0.02$) across all habitat types. Unlike temperature, mean dissolved oxygen (DO) did not differ between habitat types ($F = 0.09$; $p = 0.41$) or between habitat types within each season ($F = 0.21$; $p = 0.93$). Mean, minimum, and maximum temperatures and DO across all habitat types, seasons, and years are provided in Appendix 1.

Water quality parameters concerning water clarity and productivity had comparable patterns in their readings. Mean Secchi depth was different between all habitat types ($F = 26.01$; $p < 0.01$). Secchi depths were lowest in disconnected coves and highest in the main reservoir (Figure 4). Secchi depth did not differ between habitat types within each season ($F = 1.07$; $p = 0.38$). Mean relative chlorophyll *a* differed between all habitat types ($F = 42.47$; $p < 0.01$). Disconnected coves had the highest concentrations of chlorophyll *a*, and the main reservoir had the lowest (Figure 5). Chlorophyll *a* concentrations did not differ between habitat types within each season ($F = 1.69$; $p = 0.17$). Mean turbidity differed between all habitat types ($F = 75.98$; $p < 0.01$). Disconnected coves were the most turbid and the main reservoir was the least turbid of the three habitat types (Figure 6). Turbidity between habitat types did differ within each season ($F = 2.95$; $p = 0.03$). The same pattern of turbidity, however, was observed for each season as the habitat type main effect, indicating differences likely occurred between seasons within each habitat type (Figure 7). Mean, minimum, and maximum Secchi depths, relative chlorophyll *a*, and turbidity readings across all habitat types, seasons, and years are provided in Appendix 1.

Other water chemistry parameters had inconsistent patterns of significance. Mean pH was similar between all habitat types ($F = 1.94$; $p = 0.16$) and between habitat types within each season ($F = 1.55$; $p = 0.20$). Mean dissolved nitrate concentrations were similar between all habitat types ($F = 0.96$; $p = 0.40$). Mean dissolved nitrates also did not differ between habitat types within each season ($F = 0.90$; $p = 0.99$). No differences in mean dissolved phosphate were noted across habitat types ($F = 0.56$; $p = 0.58$). However, differences were noted between habitat types within a given season ($F = 3.03$; $p = 0.04$), particularly within spring and summer. In the spring, dissolved phosphate was highest in disconnected coves. In contrast, dissolved phosphate was higher in connected compared to disconnected coves, while the main reservoir was similar to both cove types during the summer (Figure 8). Dissolved phosphate was similar across all three habitat types during the fall (Figure 8). Mean, minimum, and maximum pH, dissolved nitrate, dissolved phosphate readings across all habitat types, seasons, and years are provided in Appendix 1.

Mean density of total zooplankton differed by habitat type ($F = 36.45$; $p < 0.01$). Densities of zooplankton were highest in disconnected coves and lowest within the main reservoir (Figure 9). Zooplankton densities did not differ between habitat types within each season ($F = 0.47$; $p = 0.76$). Mean densities of zooplankton were lower in 2017 (225 ± 69 count/L) compared to 2018 (1048 ± 396 count/L; $F = 15.97$; $p < 0.01$). Mean, minimum, and maximum total zooplankton across all habitat types, seasons, and years are provided in Appendix 1.

Zooplankton communities did appear to differ between habitat types (Figures 10 and 11). *Daphnia lumholtzi*, *Leptodora*, Harpacticoida were absent from disconnected coves and no Cydoridae or *Leptodora* were found in the main reservoir throughout this study. All other taxa were found in all three habitat types. All taxa collected were found in connected coves. Percent composition of taxa were more similar between the main reservoir and connected coves than between the main reservoir and disconnected coves (Figure 10). Calanoida and nauplii composed a smaller portion of the zooplankton community in disconnected coves compared to the main reservoir or connected coves, and *Bosmina* composed a larger proportion of the community in disconnected coves compared to the other two habitat types (Figure 10). *Daphnia* composed a smaller proportion of the community in connected and disconnected coves compared to the main reservoir (Figure 10). Rotifers composed a higher proportion of the community in both cove types compared to the main reservoir and consisted of over half of all zooplankton sampled within disconnected coves (Figure 10).

When examining zooplankton assemblage using NMDS (stress = 0.16), several patterns were observed between disconnected coves, connected coves, and the main reservoir. Sites within the main reservoir were grouped relatively closely in ordinal space, indicating consistency in their zooplankton assemblage (Figure 11). Disconnected coves, conversely, had larger distribution in ordinal space compared to the main reservoir and connected coves, indicating more variable zooplankton assemblages (Figure 11). Connected coves also show larger variability in their zooplankton assemblage compared to the main reservoir, however not to the degree of disconnected coves (Figure 11). No overlap of convex hulls in ordinal space occurred between the main reservoir and disconnected coves, indicating distinct assemblages within each habitat (Figure 11). The convex hull of connected coves overlapped entirely with that of the main reservoir and partially with disconnected coves, suggesting some degree of similarity of assemblage to both other habitat types (Figure 11). When examining correlations between taxa abundance per liter and axis scores, axis 1 was negatively correlated to nauplii, Cyclopoida, Rotifera, and *Ceriodaphnia*, and axis 2 was negatively correlated to Calanoida (Table 2). No zooplankton taxa showed a strong positive correlation for either axis (Table 2). Although distribution was highly variable, nearly all disconnected cove values and several connected cove values tended to orient at lower values of axis 1 compared to the main reservoir (Figure 11), indicating the noticeably higher proportions of nauplii, Cyclopoida, Rotifera, and *Ceriodaphnia* within the zooplankton assemblages of connected and disconnected coves compared to the main reservoir.

Discussion

Overall, the results from this study provide evidence that reservoir coves may differ abiotically and biotically from the main reservoir and the pattern of differences may reflect the level of connectivity between coves and the main reservoir. Disconnected cove habitats may have unique water quality compared to connected coves and the main reservoir. For instance, water temperature differed between disconnected coves and the main reservoir. This difference in temperature may be due to differences in water body size, as smaller and shallower water bodies, such as coves, require less energy to increase temperature (Wetzel 2001a). Disconnected coves in Lake Mohave, Arizona, and Demopolis Reservoir, Alabama, have been found to have independent water temperature regimes (i.e., warmer during the spring and summer and colder in winter) compared to the main reservoir (Marsh and Langhorst 1988; Maceina and Slipke 2003; Slipke et al. 2005). The warmer temperatures found in disconnected coves of Harlan County Reservoir throughout our study suggests that groundwater exchange is not adequate to synchronize temperatures, allowing for independent temperature regimes. Seasonal variability in temperature within disconnected coves could have ecological impacts affecting larval fish growth (Claramunt and Wahl 2000), fish and zooplankton biomass and abundance (Winemiller et al. 2000), and available DO (Cole and Hannah 1990).

DO is another water quality parameter that is critical for nearly all aquatic organisms. Reduced concentrations of DO can stress fish, potentially leading to mortality (Rottmann et al. 1992). Low oxygen levels have also been shown to influence zooplankton communities, with rotifers frequently become more abundant due to their tolerance to low DO concentrations relative to crustacean zooplankton (Karpowicz et al. 2020). All habitats in this study had somewhat similar DO levels across the three seasons sampled in this study, most measurements of DO were within values acceptable for most warm and cool water species (Doudoroff and Shumway 1970). However, no sampling was completed during the winter, and ice cover could lead to lower concentrations and, subsequently, localized winterkill events (Stefan and Fang 1997). Furthermore, because disconnected coves are isolated waterbodies, fish may have limited opportunities to seek refugia from these conditions. This limiting factor could impact the biotic communities of disconnected coves, even if occurring for only a short time frame, promoting species assemblages that are tolerant of low DO concentrations.

Water clarity parameters differed between disconnected coves, connected coves, and the main reservoir. Secchi depth was shallowest within disconnected coves, while the main reservoir had the deepest recorded depth, likely corresponding to the significantly higher and lower turbidity readings (Bachmann et al. 2017) within disconnected coves and the main reservoir, respectively. Differences in water clarity between the three habitats could be related to water depth, as shallow habitats are often more vulnerable to wind driven sediment resuspension (Miranda 2005; Knight et al. 2008). Within Harlan County Reservoir, disconnected coves varied little in depth, all being approximately 1 m deep. In contrast, depths of connected coves and main reservoir sites varied between 1 – 5 m and 3 – 12 m, respectively. Shallow depths could contribute to increased turbidity and lower Secchi depth readings, even with little wind with mild turbulence.

Primary productivity within each habitat may also influence water clarity. Like turbidity, available chlorophyll *a* was highest in disconnected coves and lowest in the main reservoir. Available chlorophyll *a* was also noted to be higher in disconnected compared to connected coves in Demopolis Reservoir, Alabama (Maceina and Slipke 2003; Slipke et al. 2005). Higher primary productivity can result in increased turbidity, as algae are suspended particles in the water column (Wetzel 2001b). The relationship between turbidity and chlorophyll *a* within Harlan County Reservoir could indicate high amounts of biogenic turbidity. Differences in chlorophyll *a* could also be related to water depth, similar to water clarity parameters, however, due to different processes. Because chlorophyll *a* samples were taken via an integrated water sample of the water column, samples from sites with deeper depths could be diluted as most of the primary production occurs in the first few meters (Kimmel et al. 1990).

Interestingly, all other water quality parameters (pH, dissolved nitrates, dissolved phosphate) did not differ significantly by habitat type alone. Dissolved phosphate, however, did differ significantly when compared on a seasonal basis. These differences could be due to differences in water depth and resuspension of phosphate locked in the sediment (Koski-Vähälä and Hartikainen 2001), differences in utilization and uptake from aquatic vegetation and algae (Knight et al. 2008), and difference in landscape runoff vulnerability between the different cove types and the main reservoir (Kennedy and Walker 1990). Additionally, it was difficult to get accurate readings, specifically for dissolved phosphate at times, and error could have significant impact on results for this parameter for this study. The similarities of pH and dissolved nitrates between disconnected coves, connected coves, and the main reservoir may suggest that those variables are being regulated by similar processes, regardless of habitat types.

In addition to water quality parameters, in this study, densities of total zooplankton were distinct between the different habitat types. Marsh and Langhorst (1988) observed similar findings within Lake Mohave, AZ, with disconnected coves higher densities of zooplankton compared to the main reservoir. Differences in abundance of zooplankton (higher in disconnected coves and lower in the main reservoir) could be related to primary production within each habitat, indicated by the levels of available chlorophyll *a* (Canfield and Watkins 1984; Dodson 1992; Canfield and Jones 1996; Shuter and Ing 1997). Higher productivity within disconnected coves, as indexed by available chlorophyll-*a* may indicate more abundant food for zooplankton in these habitats compared to connected coves and the main reservoir. Large differences in zooplankton densities between habitats could also be related to water depth. Disconnected coves often had approximately 1 m of water depth to sample zooplankton, while connected coves and the main reservoir had areas of deeper water. If the majority of zooplankton are near the surface grazing where most photosynthetic activity occurs (Adams et al. 1974; Maline et al. 2011), samples from the main reservoir and deeper connected coves may be diluted by many meters of water with negligible zooplankton abundances. Maline et al. (2011) did find that zooplankton taxa could have distinct vertical distribution within Harlan County Reservoir during the month of May, however, communities were homogeneous during most of the year. Future studies could examine zooplankton densities at various depths between habitats to determine if the same homogenization occurs within cove habitats. However, limited depths within disconnected coves may not allow for comprehensive comparisons of deeper habitat.

Differences in water quality parameters could be a factor explaining the differences in abundance of and the lack of certain taxa within different habitats. For example, *Daphnia* spp. were found in higher abundance in the main reservoir and have been shown to have limited tolerance to suspended solids and high water turbidity

(Arruda 1983; McCabe and O'Brien 1983). Rotifers, however, are more tolerant of turbid waters (Bernot et al. 2004) and were found in higher abundance within both cove habitats of Harlan County Reservoir. Similar findings were by observed by Marsh and Langhorst (1988) within lake Mohave, AZ, with rotifers being 50% more abundant within disconnected coves compared to the main reservoir. The lack of *Daphnia* spp. can influence densities of other taxa such as rotifers and *Bosmina* due to decreases in competition (Wolfinbarger 1999; DeMott and Kerfoot 1982). These competitive interactions could help explain the abundance of rotifer and *Bosmina* populations within disconnected coves. Additionally, community similarities between connected coves and the main reservoir likely indicates that these habitats are affected from their connection and made more similar due to interchange of water and organisms. Disconnected coves, conversely, are secluded and have little direct influence from connection affecting their zooplankton communities, likely producing the unique assemblages within.

This study examines water quality parameters and zooplankton assemblages within various reservoir habitats. As more reservoirs reach the end of their original life expectancy, cove disconnection is likely to become a reoccurring issue for managers. Little research to date on the ecological change that may occur within coves over time due to disconnection has been completed, but research on other disconnected aquatic habitats may provide some insights. For example, oxbow lakes have been found to experience increases in turbidity and temperature and reductions in DO the longer that they are disconnected from rivers (Miranda 2005). The results regarding oxbow lakes could indicate a progression of poor water quality over time within disconnected coves unless assuaged. Disconnected coves, however, may contribute to the biodiversity of reservoir ecosystems as they may harbor distinct zooplankton and possibly fish communities. If reduced water quality is persistent, however, cove reconnection may be necessary as poor water quality for too long could lead to reductions of some taxa and species in aquatic communities.

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Tables and Figures

Table 1. List of zooplankton taxa identified within Harlan County Reservoir. Bold classifications represent the lowest taxonomic grouping identified using taxonomic keys.

Kingdom	Phylum	Class	Order	Family	Genus	Species
Animalia	Arthropoda	Branchiopoda	Cladocera	Daphniidae	*Daphnia*	***pulcaria***
"	"	"	"	"	"	***retrocurva***
"	"	"	"	"	"	***lumpultsi***
"	"	"	"	"	***Ceriodaphnia***	
"	"	"	"	Bosminidae	***Bosmina***	
"	"	"	"	Sididae	***Diaphanosoma***	
"	"	"	"	Leptodoridae	***Leptodora***	
"	"	"	"	**Chydoridae**		
"	"	Copepoda	**Calanoida**			
"	"	"	**Cyclopoida**			
"	"	"	**Harpacticoida**			
"	"	**Ostracoda**				
"	**Rotifera**					

Table 2. Spearman rank correlation coefficients between density (number/L) of various zooplankton taxa groups by site and the corresponding NMDS axis score. Higher correlation coefficients ($r_s > 0.40$ or $r_s < -0.40$; Gido et al. 2009) are represented in bold and were used to interpret NMDS ordination plot.

Spearman's Rank Correlation (r_s)		
Taxa	Axis 1	Axis 2
Daphnia pulicaria	0.01	0.24
Daphnia retrocurva	0.19	-0.23
Daphnia lumholtzi	0.02	-0.34
Immature *Daphnia*	0.27	0.10
Bosmina	-0.28	0.34
Ceriodaphnia	**-0.40**	0.09
Diaphansoma	0.23	-0.35
Leptodora	-0.14	-0.19
Cydoridae	-0.14	0.21
Calanoida	-0.28	**-0.51**
Cyclopoida	**-0.81**	0.26
Harpacticoida	0.16	0.04
nauplii	**-0.90**	-0.10
Ostracoda	-0.06	0.10
Rotifera	**-0.74**	0.08

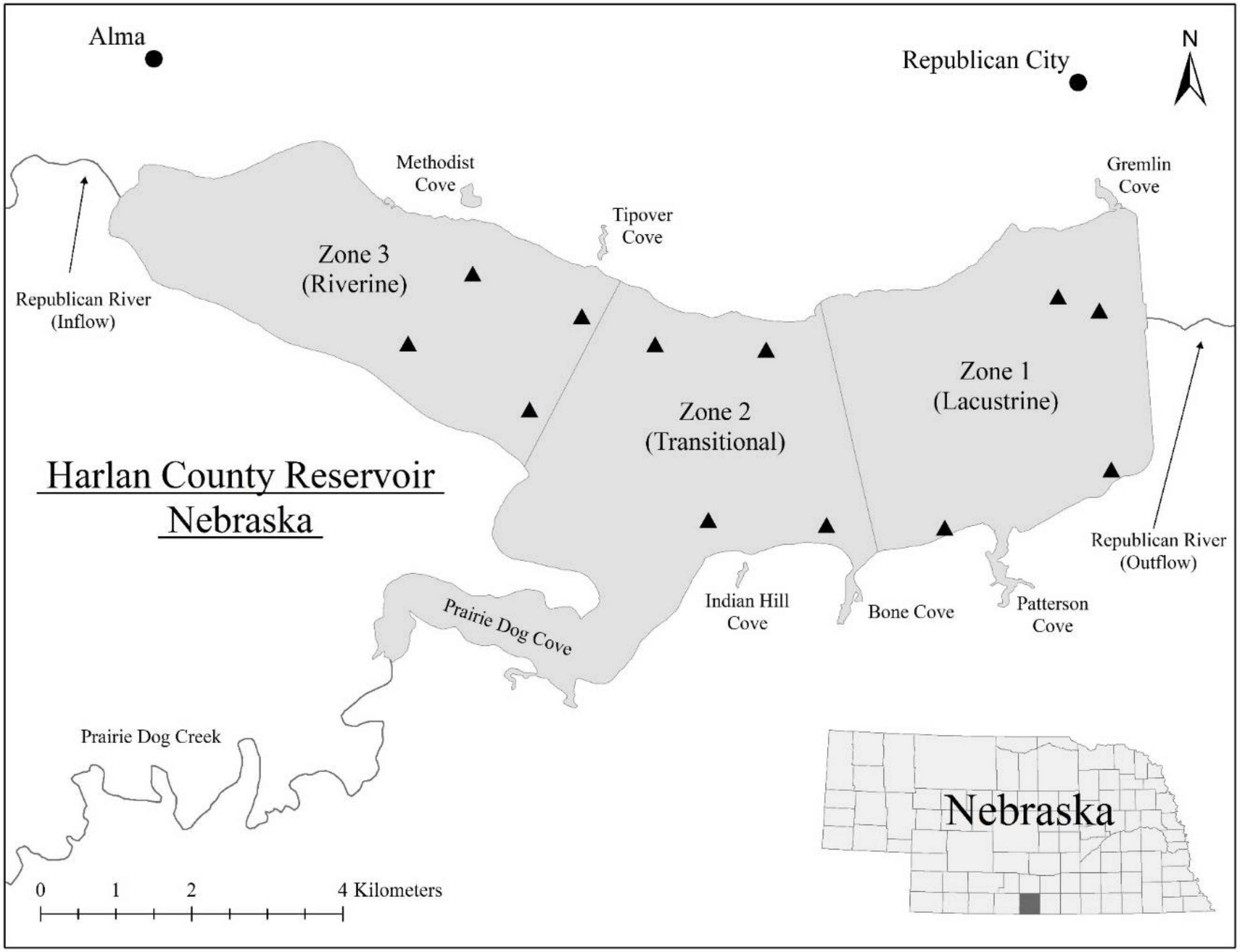
Alma
Republican City
N
Methodist Cove
Gremlin Cove
Tipover Cove
Zone 3 (Riverine)
Republican River (Inflow)
Zone 2 (Transitional)
Zone 1 (Lacustrine)
Harlan County Reservoir
Nebraska
Republican River (Outflow)
Indian Hill Cove
Bone Cove
Patterson Cove
Prairie Dog Cove
Prairie Dog Creek
Nebraska
0
1
2
4 Kilometers

Figure 1. Map of Harlan County Reservoir (adapted from aerial imagery taken by the USDA-NRCS July 13, 2016; surface water elevation approximately 591 MASL) and its surrounding cities. Reservoir zones were previously established by Peterson et al. (2005). Black triangles represent sampling stations within each of the main reservoir zones.

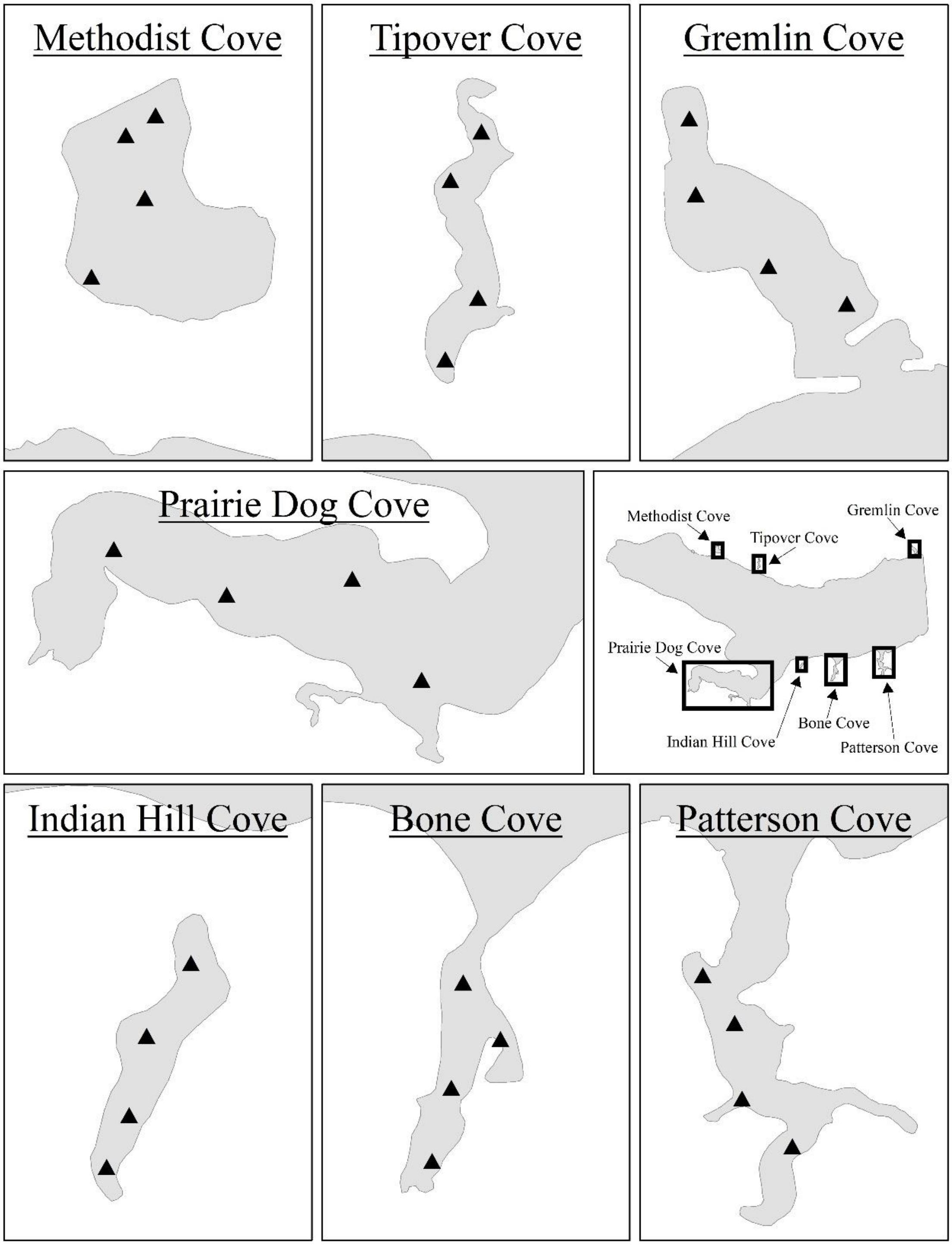
Methodist Cove
Tipover Cove
Gremlin Cove
Prairie Dog Cove
Methodist Cove
Gremlin Cove
Tipover Cove
Prairie Dog Cove
Bone Cove
Indian Hill Cove
Patterson Cove
Indian Hill Cove
Bone Cove
Patterson Cove

Figure 2. Map of coves within Harlan County Reservoir (adapted from aerial imagery taken by the USDA-NRCS July 13, 2016; surface water elevation approximately 591 MASL). Black triangles represent sampling stations within each study coves.

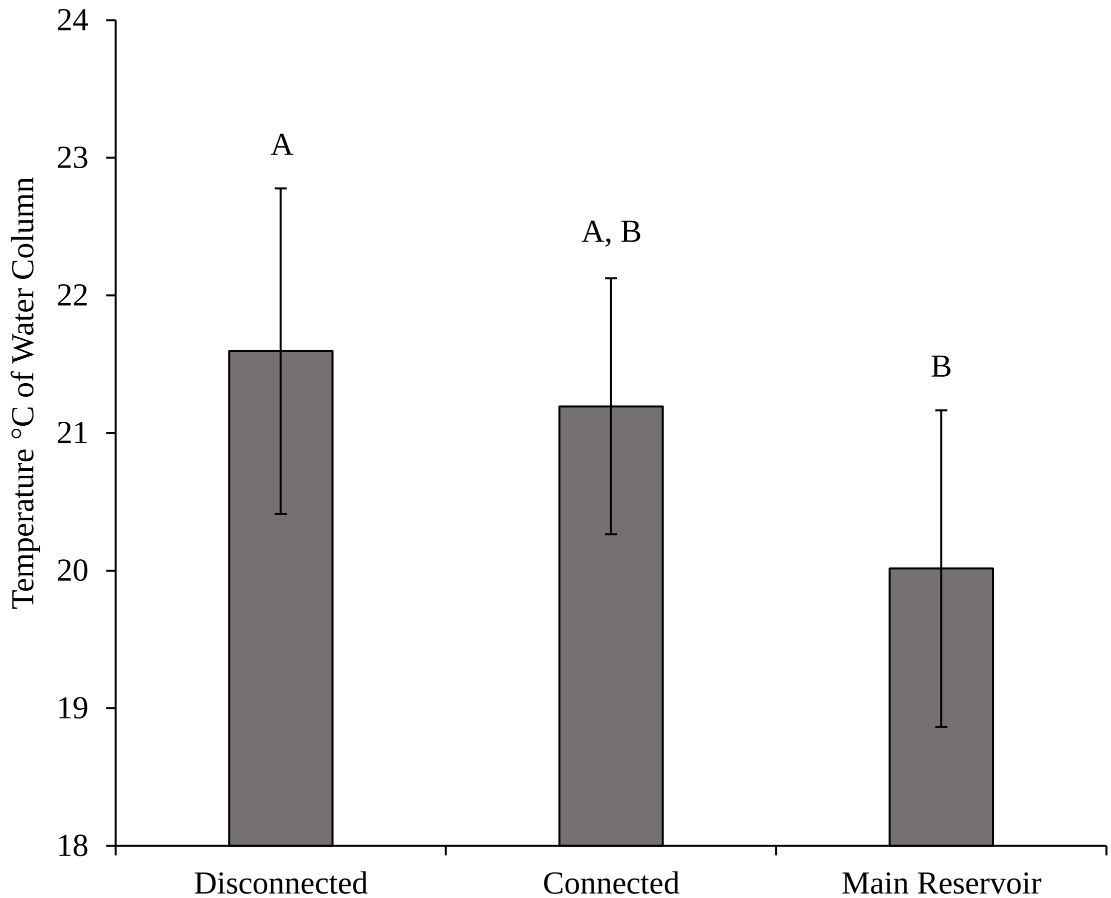

Figure 3. Mean water column temperatures (°C) within different habitat types in Harlan County Reservoir. Error bars denote one standard error. Letters above the bars denote significant differences based on a Tukey test (α = 0.10).

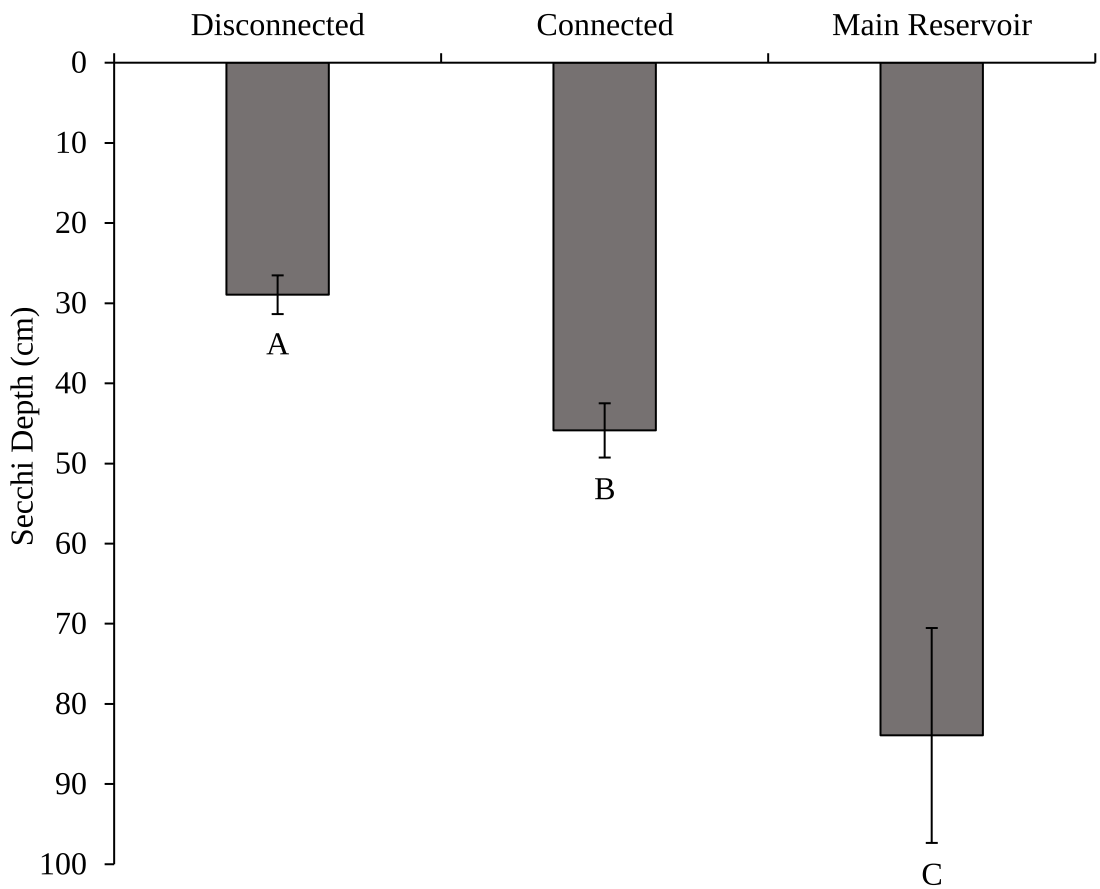

Figure 4. Mean Secchi depth within different habitat types in Harlan County Reservoir. Numerical axis has been inverted to better represent depth within the water column from the water surface. Error bars denote one standard error. Letters above the bars denote significant differences based on a Tukey test (α = 0.10).

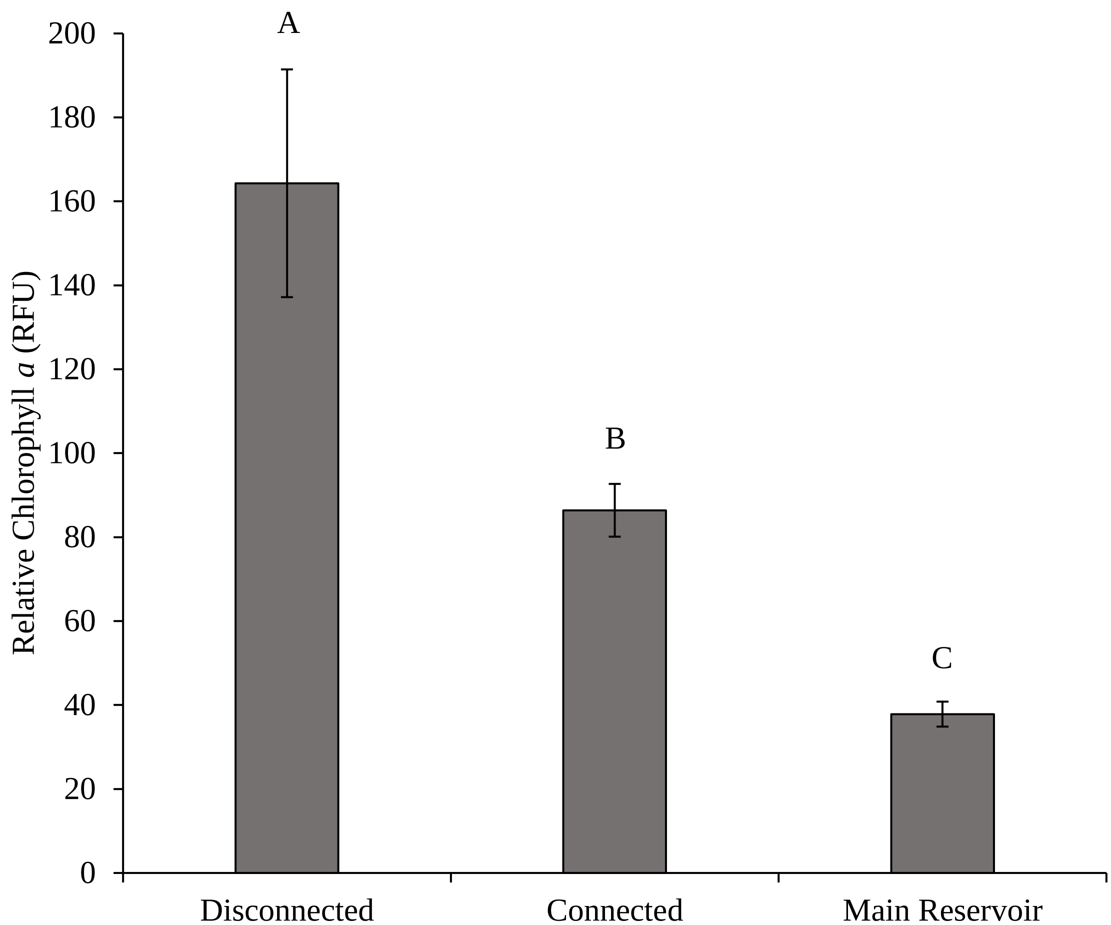

Figure 5. Mean relative chlorophyll *a* within different habitat types in Harlan County Reservoir. Error bars denote one standard error. Letters above the bars denote significant differences based on a Tukey test ($\alpha = 0.10$).

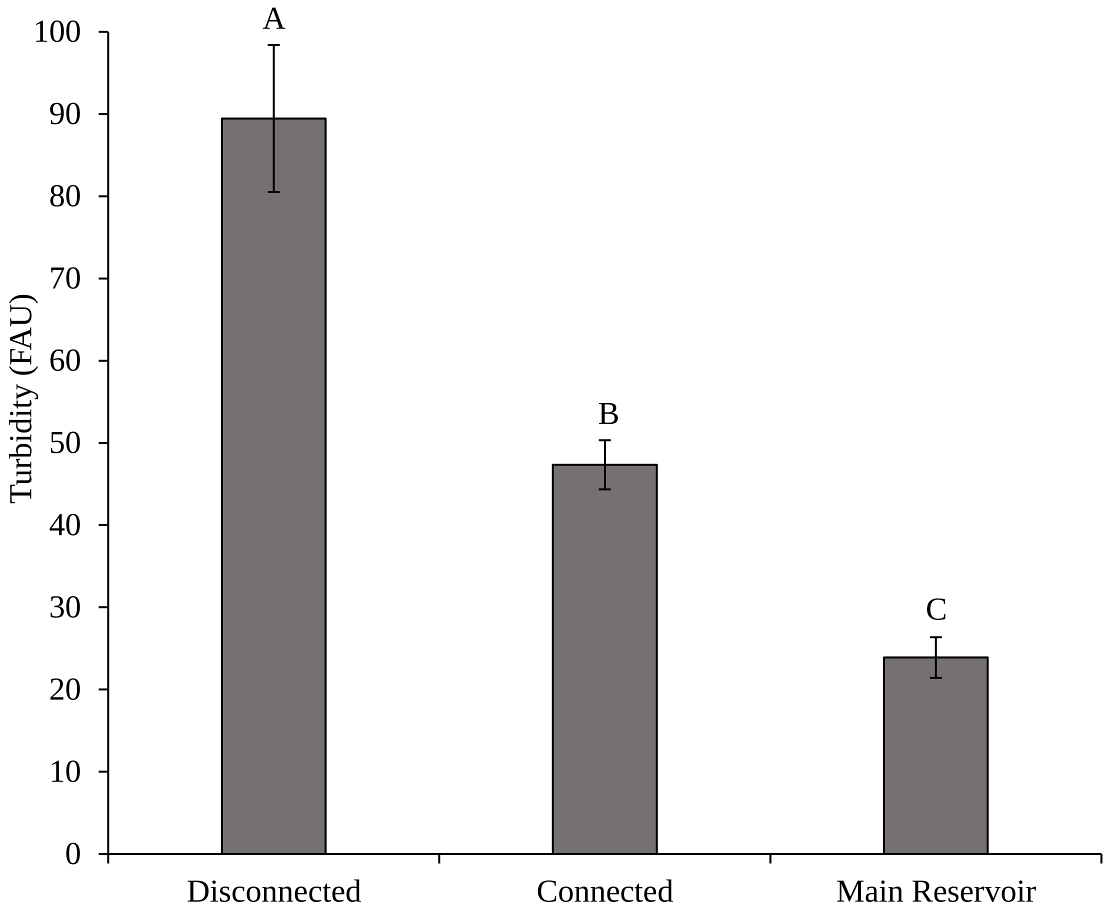

Figure 6. Mean turbidity (FAU) within different habitat types in Harlan County Reservoir. Error bars denote one standard error. Letters above the bars denote significant differences based on a Tukey test ($\alpha = 0.10$).

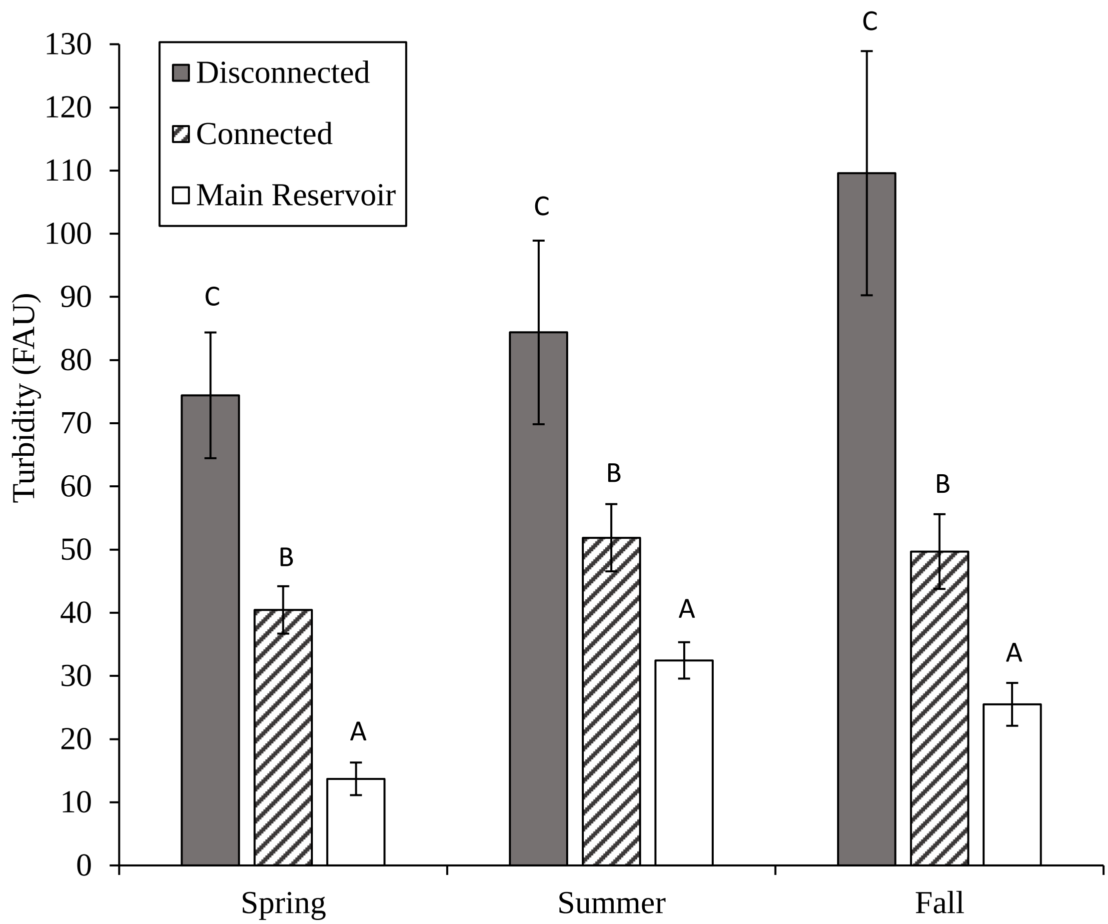

Figure 7. Mean turbidity (FAU) within disconnected coves, connected coves and the main reservoir for each sampling season. Error bars denote one standard error. Letters above the bars denote significant differences based on a Tukey test conducted within the seasons (α = 0.10).

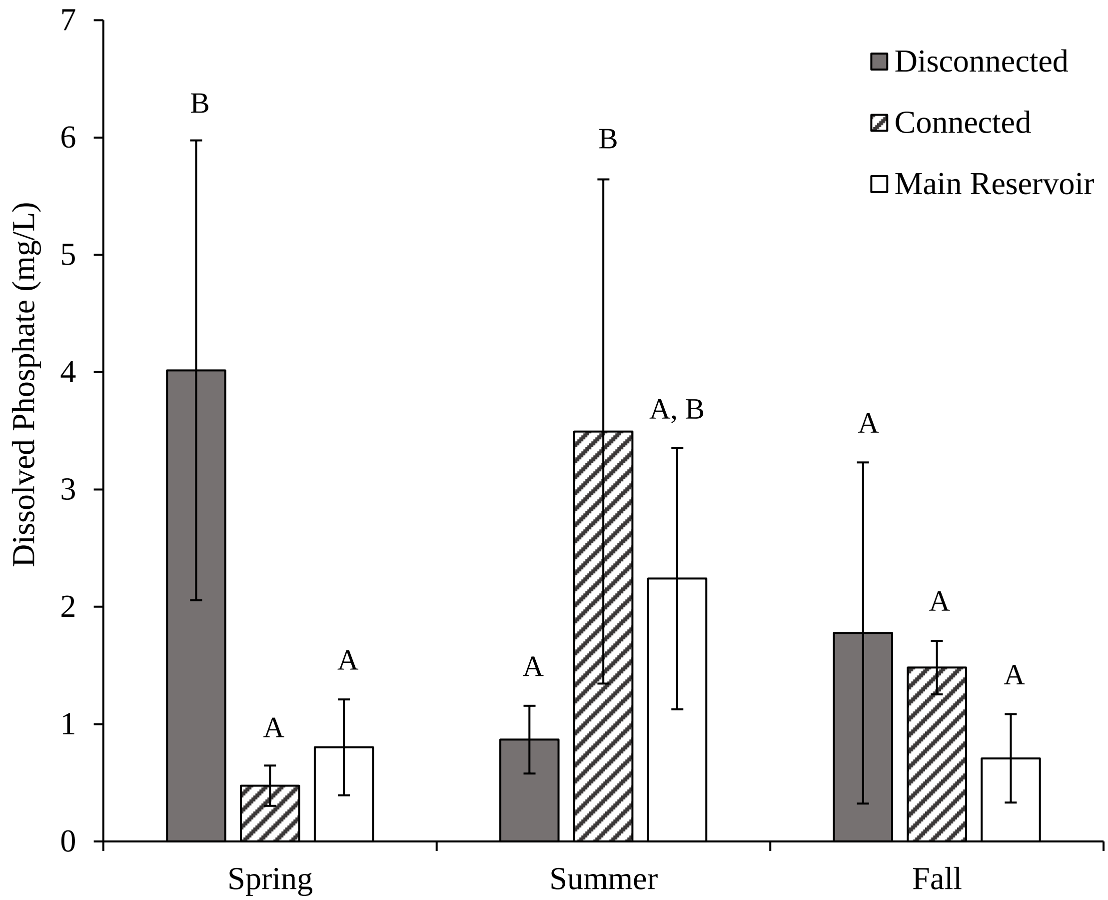

Figure 8. Mean dissolved phosphate (mg/L) within disconnected coves, connected coves and the main reservoir for each sampling season. Error bars denote one standard error. Letters above the bars denote significant differences based on a Tukey test ($\alpha = 0.10$).

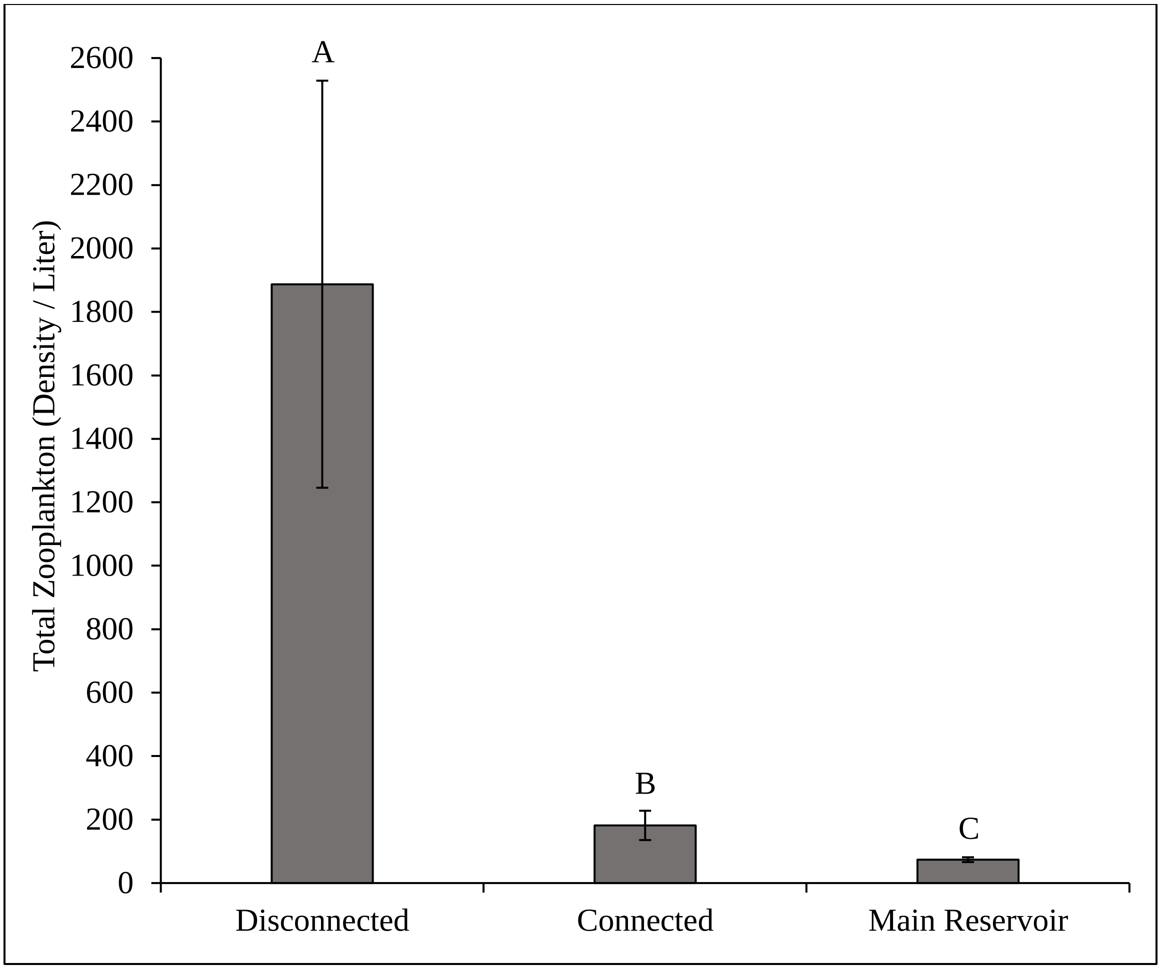

Figure 9. Mean total zooplankton density within different habitat types in Harlan County Reservoir. Error bars denote one standard error. Letters above the bars denote significant differences based on a Tukey test ($\alpha = 0.10$).

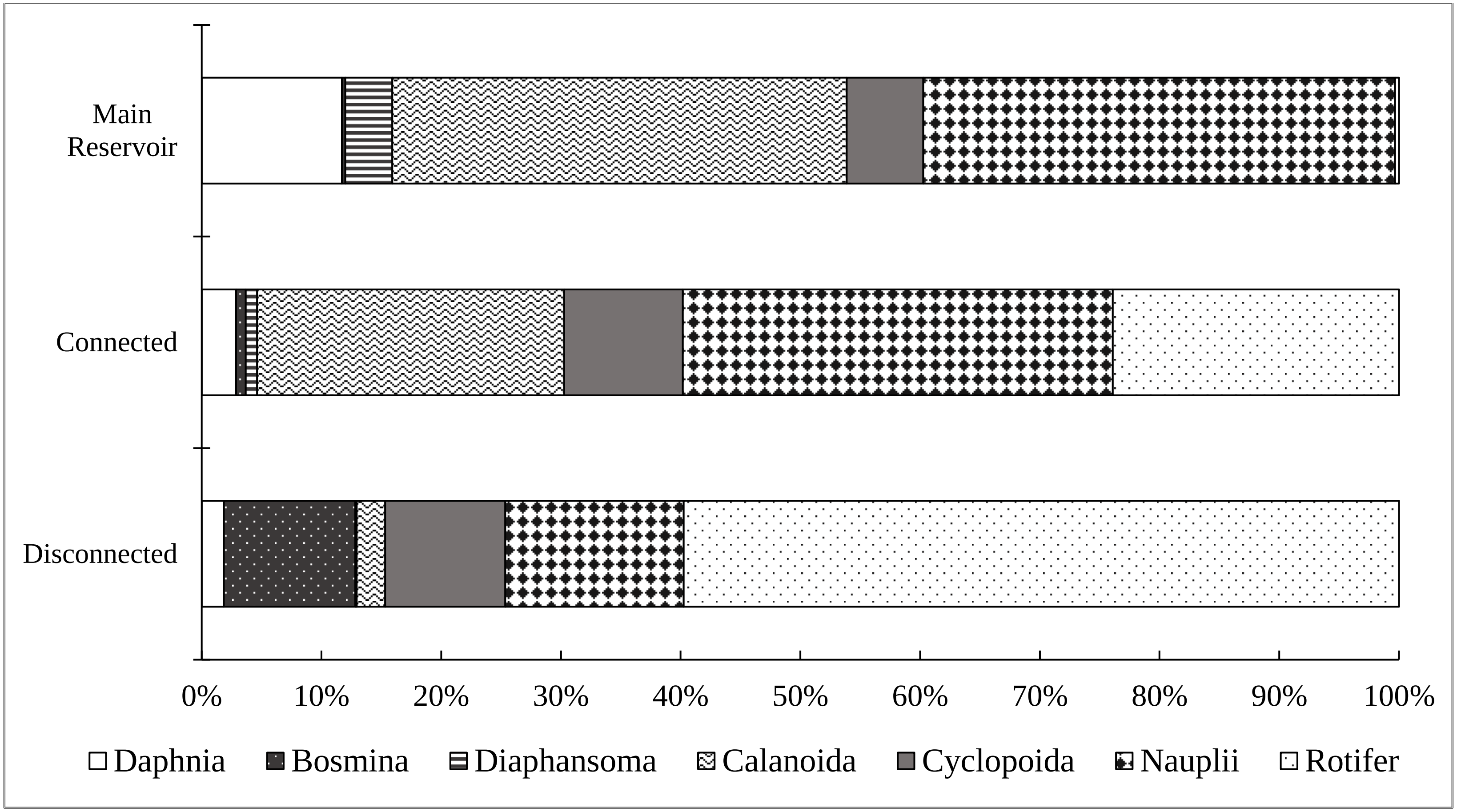

Figure 10. Comparison of the mean percent abundance of zooplankton taxa groups within different habitat types in Harlan County Reservoir. Zooplankton taxa groups included in this figure consist of those with ≥ 1% of the total density of zooplankton for at least one of the three habitat types.

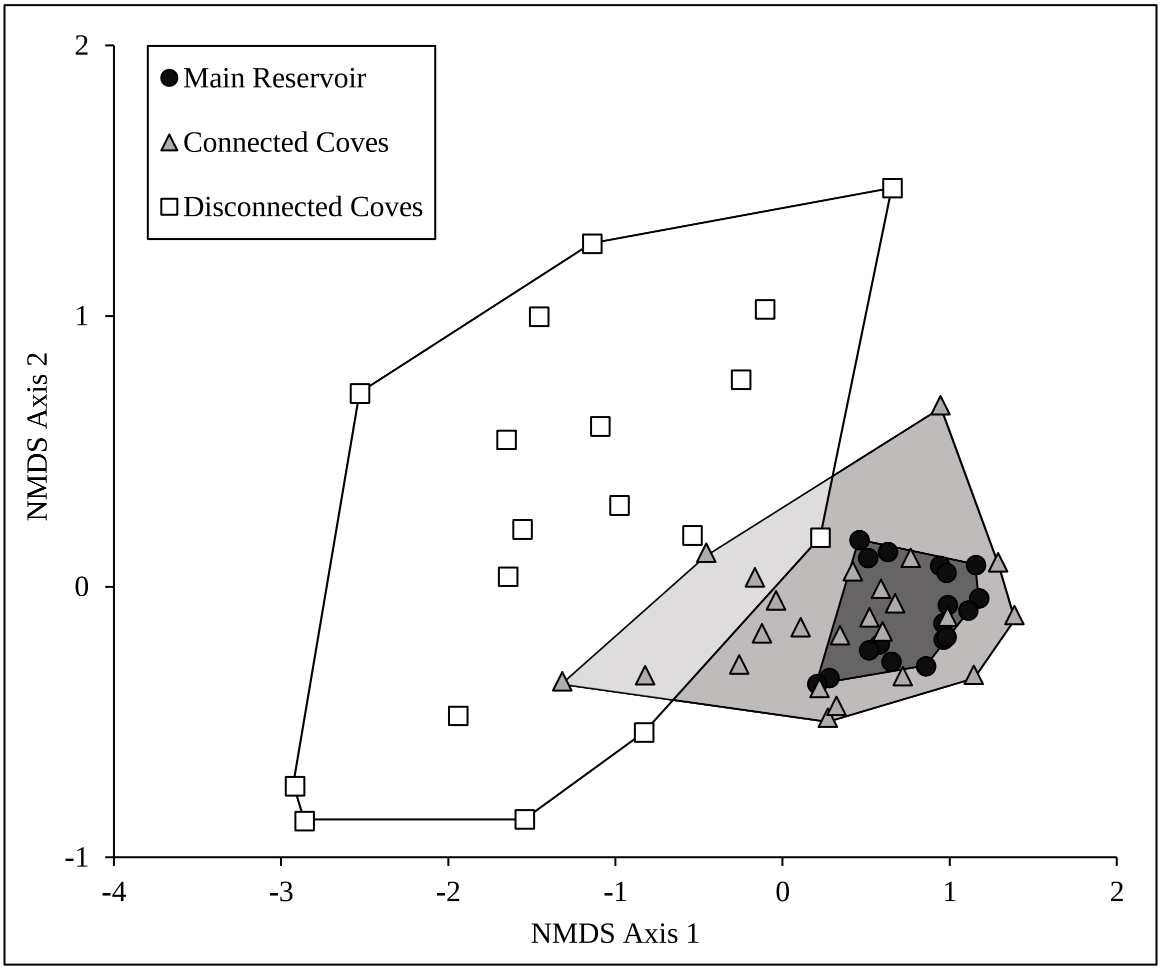

Figure 11. Plot of non-metric multidimensional scaling (NMDS) of zooplankton taxa densities per liter at sampling location within Harlan County Reservoir. Polygons represent convex hulls around each habitat type in ordinal space; the white polygon represents disconnected coves, the light gray represents connected coves, and dark gray represents the main reservoir.

CHAPTER 3:

COMPARISON OF FISH COMMUNITIES BETWEEN COVES OF VARYING CONNECTION TO HARLAN COUNTY RESERVOIR, NEBRASKA

Brian E. Mason

Introduction

Coves are common features within reservoir systems and are typically formed by the flooding of former tributaries when water levels are elevated following dam construction (Miranda et al. 2014). Cove habitats often encompass a notable amount of the limited shallow littoral area within reservoir systems (Miranda et al. 2014). Physical, chemical, and biological attributes of coves are often different compared to those within the main reservoir (Meals and Miranda 1991). Aquatic macrophytes more readily established in coves compared to other areas of the reservoir due to water level stability and protection from wave and wind action (Marsh and Langhorst 1988; Slipke et al. 2005; Dagel and Miranda 2012).

The unique physical, chemical and biological attributes of cove habitats within a reservoir can be important for biological communities, affecting both aquatic invertebrates and fish (Ferrer-Montaño and Dibble 2002; Geraldes and Boavida 2004). Coves often have higher zooplankton abundances and unique taxa compared to the main reservoir (See Chapter 2). These community differences could be important for larval fish recruitment, as small zooplankton such as rotifers and copepod nauplii are important diet items for many larval fish species (Ludwig 1999). Many recreationally important centrarchids such as Bluegill (*Lepomis macrochirus*), Crappie (*Pomoxis spp*.) and Largemouth Bass (*Micropterus salmoides*), often utilize coves for spawning and nursery habitat (Meals and Miranda 1991; Warren 2009). Although other shallow areas may be available for nest spawning fish, higher spawning success and recruitment has been reported for centrarchids within cove habitats compared to main reservoir shoreline, due to the protection they offer from wind and wave action (Meals and Miranda 1991). Cove fish communities may also have higher abundance and species richness compared to the main reservoir (Gido and Matthews 2000). Some species, such as Brook Silverside (*Labidesthes sicculus*) are typically only found within coves (Matthews 1998). Because numerous fish species use reservoir coves for feeding, reproductive, and nursery habitat (Meals and Miranda 1991; Slipke et al. 2005), disconnection could alter the dynamics of the sportfish population within isolated coves and reduce overall recruitment to the reservoir.

As reservoirs age, coves can become disconnected from their parent reservoir due to sediment accumulation at the cove mouth. Cove isolation can occur by lateral drift of shoreline sediments (Marsh and Langhorst 1988; Mueller 1995) or by backfill of sediments from the main lake (Slipke et al. 2005; Slipke and Maceina 2007). The development of sediment berms restricts or eliminates exchange of surface water between coves and the main lake (Marsh and Langhorst 1988; Slipke and Maceina. 2005). As a result, fish within isolated coves may be confined to this habitat alone, depending on water depth (Slipke and Maceina. 2005; Slipke et al. 2005). This isolation could restrict access to critical resources for fish either located within coves or the main reservoir. Examples of this would be the restriction of available spawning habitat within coves for species that utilize these areas (Meals and Miranda 1991), or fish in disconnected coves having a lack of access to high-quality food resources found within the main reservoir, such as large annual year classes of bait fish including Gizzard Shad (*Dorosoma* *cepedianum*)(Sullivan et al. 2011). Furthermore, isolation may lead to changes in water quality of coves, potentially favoring fish species tolerant of poor water condition.

Harlan County Reservoir, Nebraska, has numerous coves (Chapter 2, Figure 2). Several coves have developed sediment berms at their mouths that have isolated them from the main reservoir for numerous consecutive years, while other coves have remained continuously connected to the main reservoir due to limited berm development or by dredging of sediments. The objective of this study is to describe fish assemblages within coves of Harlan County Reservoir, Nebraska, and to compare those assemblages between coves connected to the main reservoir and those that have experienced isolation (i.e., > 4 years) due to berm formation.

Methods

Harlan County Reservoir is operated by the U.S. Army Corps of Engineers and is located in southcentral Nebraska near the Kansas border (Chapter 1, Figure 5). Initiation of dam construction began in 1946, and the dam currently operates for the primary purposes of flood control on the Republican River and to withhold water for irrigation (USACE 2011). At full conservation pool, Harlan County Reservoir has a surface area of approximately 5,400 ha and a storage capacity of approximately 1 billion m^3 (USACE 2011). Mean depth of the reservoir at full conservation pool is 4 m and maximum depth is 18 m (Uphoff et al. 2013) Water elevation, however, can vary up to 3 m on an annual basis due to seasonal changes in precipitation and irrigation demands (Diffendal et al. 2002). Thermal stratification rarely occurs due to the reservoir’s westward orientation and its long fetch distance (Olds et al. 2011; Uphoff et al. 2013).

For the purpose of this study, definitions of disconnected and connected coves remain consistent with those described in Chapter 1. All connected coves within Harlan County Reservoir (Bone, Gremlin, Patterson and Prairie Dog Coves) were included in the study. Disconnected coves (Indian Hill, Methodist, and Tipover Coves) were selected based on their candidacy for future renovation projects to reconnect coves to the main reservoir (Keith Koupal, *personal communication*, June 2020). Indian Hill, Methodist, and Tipover coves had been disconnected from the main reservoir (≥1 m depth of surface water connection) since September 1993, February 2012, and June 2012, respectively, assuming sediment berm heights have remained similar since the last connection event (USBOR 2020; Figure 1).

Fish were sampled during the spring (May), summer (July), and fall (September – October) of 2017 and 2018. During fall and spring, four 137 x 93 cm frame, single-throat modified fyke nets [two 16-mm mesh and two 19-mm mesh] with a maximum lead length of 15 m, were deployed overnight within each cove. During summer, a 15.25 x 1.22-m purse seine with 0.63-cm bar mesh netting were pulled in quarter-arcs at four locations within each cove (Pope et al. 2009; Miller et al. 2018). The location of the four fyke net sets and seine hauls were equally spaced throughout each cove and were maintained for the entirety of the project or moved to the nearest possible location if water level fluctuations and gear restrictions prevented adequate sampling in a specified location. All fish collected during this study were identified to species and released.

Species data was combined across cove type and years. Species data was then used to calculate species richness and Shannon's diversity index (Shannon 1948) for each cove type for comparison. Species data was also used to calculate Jaccard's similarity index (Jaccard 1901) and Renkonen percent similarity index (Renkonen 1938) between cove types. Both measures of similarity were calculated as Renkonen percent similarity index is based on the relative abundance of fish instead of presence data as used in the Jaccard's index.

To examine species associations to connected or disconnected coves, we used a metric multidimensional scaling (MMS) ordination, using Bray-Curtis distance metrics, calculated from species abundances from each cove. The MMS ordination procedure was used in preference to other ordination methods as it does not require datasets to be normally distributed and retains reliability when examining datasets with a large number of zero values (Miranda et al. 2014). Spearman's rank correlation coefficients were calculated between fish species abundance and axis values for each cove. Species with relatively strong abundance associations to the axis values ($r_s > 0.40$ or $r_s < -0.40$; Gido et al. 2009) were used to interpret ordinal distribution patterns corresponding to fish species assemblages within disconnected and connected coves. Indicator species analysis (Dufrene and Legendre 1997) was conducted using the same species abundance data as the MMS ordination analysis to further identify species that are commonly associated with either connected or disconnected coves. The resulting indicator values are a function of a species relative abundance and relative frequency within either connected or disconnected coves. Indicator species analysis was conducted using the "indval" function from the "labdsv" package of program R (R Core Team 2020). In order to test for statistical significance, the Monte Carlo method was used to randomly reassign sample units and recalculate indicator values for each species. After 10,000 iterations, the p-value was calculated using the proportion of the recalculations for each species that was greater than or equal to the observed maximum indicator value. Species were considered indicators if their indicator values were >25 (Miranda et al. 2014) and $p \leq 0.1$.

Results

A total of 5,964 fish (3,426 within four connected coves; 2,538 within three disconnected coves), representing 24 species, were collected between all study coves throughout this study. Eight species of fish were collected only within connected coves while disconnected coves had no unique species (Table 1). Species richness was lower in disconnected versus connected coves (16 versus 24, respectively). Diversity was higher in connected (H' = 1.96) than disconnected coves (H' = 1.45). Species assemblages between the two cove types were highly similar (J = 66.67) based on presence/absence data. However, similarity via abundances of like species (Renkonen percent similarity index) were not similar between cove types (PS = 0.32). Fish species with the largest difference in the percent of assemblage, lowering the Renkonen similarity value, included Gizzard Shad, having higher percent abundance within connected coves; and Black Bullhead (*Ameiurus melas*) and Orangespotted Sunfish (*Lepomis humilis*), having higher percent abundance within disconnected coves (Table 1).

The MMS ordinations showed distinct separation between fish assemblages within connected and disconnected coves (Figure 2). Connected coves typically had high axis 1 values with lower axis 2 values, and all four coves appeared to cluster together

(Figure 2). Variation of axis 2 was higher among disconnected coves, and individual coves appear to separate into two distinct ordinal patterns. One such pattern involves a single cove, Indian Hill Cove, which was observed with the lowest axis 1 and 2 values. Conversely, the other disconnected coves in this study had considerably higher axis 2 scores compared to connected coves. Spearman correlations between individual species abundance and ordination axes indicate that Walleye (*Sander vitreus*), Northern Pike (*Esox lucius*), Red Shiner (*Cyprinella lutrensis*), White Bass (*Morone chrysops*), Hybrid Striped Bass (*Morone chrysops x saxatilis*), Channel Catfish (*Ictalurus punctatus*), Emerald Shiner (*Notropis atherinoides*), Gizzard Shad, River Carp Sucker (*Carpiodes carpio*), Largemouth Bass, Western Mosquitofish (*Gambusia affinis*), and Freshwater Drum (*Aplodinotus grunniens*) were positively correlated to Axis 1, and connected coves respectively (Figure 2). Longnose Gar (*Lepisosteus osseus*), Shortnose Gar (*Lepisosteus platostomus*), and Freshwater Drum were negatively correlated with axis 2, and thus were also associated with connected coves (Figure 2). Black Crappie (*Pomoxis nigromaculatus*), White Crappie (*Pomoxis annularis*), Green Sunfish (*Lepomis cyanellus*), Black Bullhead, Bluegill, Orangespotted Sunfish (*Lepomis humilis*), Brook Silverside, and Hybrid Sunfish (*Lepomis macrochirus x cyanellus*) were species positively correlated with axis 2 and having higher affinity toward disconnected coves, with the exception of Indian Hill Cove (Figure 2). Common Carp were negatively correlated to both axes, suggesting a presence in both cove types (Figure 2).

Similar to the MMS ordination, the indicator species analysis suggested differences in the fish assemblages between disconnected and connected coves. White

Bass, Freshwater Drum, Channel Catfish, Gizzard Shad, River Carp Sucker, Walleye, Shortnose Gar, Northern Pike, Western Mosquitofish, Largemouth Bass, and Red Shiner appeared to be indicator species of connected coves (Figure 3). All of the species identified as indicators for connected coves also coincide with several species having strong positive correlation of axis 1 or strong negative correlations to axis 2 of the MMS ordination, signifying affinity toward connected cove habitat. Although several species had indicator values for disconnected coves higher than 25, no species had a $p \leq 0.1$, henceforth, no species were indicators for disconnected coves (Figure 3).

Discussion

Lateral connectivity of isolated waterbodies to their parent source can play an important role in determining fish assemblages within these habitats (Tockner et al. 1999; Miranda et al. 2014; Gilbert and Pease 2019). Connected waterbodies likely have species assemblages with high degrees of similarity, as fish can freely navigate and be exchanged between the two habitats (Patton and Lyday 2008). In contrast, waterbodies in isolated systems lose the external contribution of fish, often for long stretches of time (Patton and Lyday 2008). The loss of cove habitats due to disconnection could further impact the reservoir fish community, as coves are a vital component of the mosaic of habitats.

Previous studies examining fish communities of disconnected and connected coves and backwater habitats both support and conflict with the results of this study. Slipke and Maceina (2005) found that diversity was similar between connected and disconnected backwater habitats of Demopolis Reservoir, AL. However, disconnected backwaters had slightly reduced species richness (Slipke and Maceina 2005). Patton and Lyday (2008) found similar species richness and diversity between disconnected and connected habitats in Lake Texoma, OK. In contrast, this study of Harlan County Reservoir found connected coves had higher species richness and higher diversity compared to disconnected coves. The different results between studies may be due to reginal differences, as well as differences in reservoir dynamics and functions.

Species composition of fish communities within disconnected and connected habitats have also shown documented differences. In Demopolis Reservoir, AL, Slipke and Maceina (2005), found high abundances of Gizzard Shad and centrarchid species within both connected and disconnected habitats, however, disconnected backwaters had significantly higher abundances of Bluegill, Redear Sunfish (*Lepomis microlophus*), and Spotted Gar (*Lepisosteus oculatus*). Connected coves, on the other hand, had significantly higher abundances of Largemouth Bass and White Crappie, indicating connection may play an important role for these species within backwater habitats (Slipke and Maceina 2005). Disconnected habitats within Lake Texoma had more variable fish assemblages but generally separated into two groups; those with assemblages more strongly influenced by White Crappie, Channel Catfish, and River Carpsucker, and those influenced more heavily by Freshwater Drum and Blue Catfish (*Ictalurus furcatus*) (Patton and Lyday 2008). Furthermore, White Bass and Striped Bass had a consistently higher affinity to connected habitats (Patton and Lyday 2008). Species within the family Moronidae have been documented as absent or found in greatly reduced numbers within disconnected habitats compared to those with connection, perhaps indicating reductions in long-term survival and/or recruitment within small, isolated systems (Slipke and Maceina 2005; Patton and Lyday 2008). Gilbert and Pease (2019) found higher abundance of larval *Pomoxis*, and *Morone* species within disconnected habitats of Lake Texoma, OK, during a year of high water and connectivity of these habitats, compared to the previous year which included a period of isolation. Within disconnected coves of Harlan County Reservoir, no fish from the Moronidae were recorded. Walleye, Northern Pike, and Emerald Shiner were also absent from disconnected coves, but were consistently found in connected coves. Additionally, River Carpsuckers were also absent from disconnected coves but frequently found in connected coves in Harlan County Reservoir. This is in contradiction to the findings by Patton and Lyday (2008), as River Carpsucker were more associated with fragmented habitats in Lake Texoma, OK. The lack of consistency in abundance of River Carpsucker within disconnected habitats may indicate that factors other than isolation could be impacting this species.

Species assemblages within disconnected habitats may be linked to the length of time of disconnection. Patton and Lyday (2008) indicated that the distinct assemblage groupings noted within fragmented habitats were likely related to the duration of disconnection, as the site containing abundant Blue Catfish and Freshwater Drum was more recently connected to the main reservoir than the other fragmented habitats in their study. Disconnected coves of Harlan County Reservoir also exhibit two distinct patterns of assemblage: two coves had numerous Centrarchid species and Black Bullhead, and the other cove (Indian Hill) was composed entirely of Common Carp (*Cyprinus carpio*). The latter cove had been disconnected from the main reservoir for approximately 24 years at the beginning of this study, whereas the other two had only been disconnected for approximately 5 years. This relationship between species assemblage and timeframe of isolation within disconnected coves of Harlan County Reservoir could indicate that Indian Hill Cove is more advanced in its species assemblage succession than the younger coves, supporting only a single generalist species, extremely tolerant of disconnected conditions. Additionally, several droughts had occurred over Indian Hill Cove's period of isolation, causing low water levels within the main reservoir (figure 6, chapter 1) and possibly exacerbating the extreme conditions within disconnected coves. These periods may have accelerated assemblage succession within Indian Hill Cove, while coves with less developed sediment berms experienced reconnection after low periods and were likely replenished with species from the main reservoir. Moreover, the lack of identification of indicator species for disconnected coves within this study may be due to the lack of consistency of species assemblage within said habitats.

Water quality and zooplankton differ between connected and disconnected coves (see Chapter 2), and these differences likely influence the fish assemblages. Disconnected coves within Harlan County Reservoir had higher water turbidity, lower Secchi depth, and potentially limited dissolved oxygen compared to connected coves (see Chapter 2). Common Carp, Black Bullhead, and Orangespotted Sunfish are highly tolerant to these conditions (Zambrano et al. 2001; Novomeská and Kováč 2009; Warren 2009), thus, their increased abundance within these habitats may be expected. Additionally, benthivorous fish, such as Common Carp, may contribute to the increased turbidity observed within disconnected coves by resuspending solids with their feeding behavior

(Zambrano et al. 2001). Because of this phenomenon, a negative feedback may become established within disconnected coves, as increased turbidity caused by benthivorous fish behavior leads to reduced vegetation growth, additional de-stabilizing of the substrate, and subsequent increased turbidity (Scheffer 1997). High turbidity may also help tolerant species avoid predation, thus, increasing their density and supporting additional increases in turbidity (Miranda 2005). The relationship between fish assemblage and water quality may, in turn, have an impact on the zooplankton within disconnected coves of Harlan County Reservoir. Taxa with a high tolerance to turbidity, such as rotifers, were found in more abundance within these habitats, and other species such as Daphnia and Calanoida were reduced (See chapter 2). The difference in zooplankton assemblage could lead to other changes in fish assemblages. For example, Daphnia and Calanoida are the preferred food source for larval Walleye and White Bass (Beck 1998; Miller et al. 2019; Uphoff et al. 2019); the absence of food for these individuals may lead to avoidance of or lower survival within disconnected coves for these species. Alternatively, larval *Pomoxis* spp. have been shown to select for smaller zooplankton (Dubuc and DeVries 2002) such as those taxa found in disconnected coves (see Chapter 2); thus, individuals of this genera may have an abundant food source in these habitats early in their development.

Overall, this study of Harlan County Reservoir provides information regarding differences in fish assemblages between cove types that may be useful for fisheries managers to consider for other reservoirs across the U.S. with similar habitats. Assemblages within disconnected coves may be impacted by the extent of isolation from the parent reservoir, particularly the longer disconnection is maintained. These assemblage changes likely impact the water quality and zooplankton communities within disconnected coves. Reservoir managers may find value in reconnecting disconnected cove habitats, as connected coves had higher diversity and species richness and higher association with sportfish species, such as White Bass, Walleye, Norther Pike, and Largemouth Bass. Alternatively, disconnected coves may offer unique ecological and recreational potential by harboring distinct fish assemblages compared to connected coves and the main reservoir, containing species such as Orangespotted Sunfish, Green Sunfish, Black Bullhead, and Brook Silversides. If disconnected coves remain isolated, they may require additional ongoing maintenance to help alleviate potentially low DO, improve water clarity, and control measures of potentially nuisance benthivorous fish. These measures could extend the longevity of disconnected coves and prevent assemblages of only the most tolerant species. Managers can use this information when planning cove renovations by weighing the costs and benefits of either maintaining ecologically distinct disconnected coves, versus connecting coves and improving habitat for reservoir fisheries.

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Tables and Figures

Table 1: Percent composition of the total assemblage of fish species collected within connected and disconnected coves of Harlan County Reservoir. Collection took place during the spring summer and fall of 2017 and 2018. Counts are indicated within parentheses. Dash marks indicate the non-detection of the corresponding species.

Family	Species	Common Name	Connected	Disconnected
Atherinidae	*Labidesthes sicculus*	Brook Silverside	0.06 (2)	0.12 (3)
Catostomidae	*Carpiodes carpio*	River Carpsucker	0.64 (22)	-
Centrarchidae	*Lepomis cyanellus*	Green Sunfish	0.35 (12)	2.68 (68)
	Lepomis humilis	Orangespotted Sunfish	0.53 (18)	11.70 (297)
	Lepomis macrochirus	Bluegill	6.80 (233)	7.45 (186)
	Lepomis macrochirus x cyanellus	Hybrid Sunfish	0.09 (3)	0.16 (4)
	Micropterus salmoides	Largemouth Bass	3.56 (122)	0.28 (7)
	Pomoxis annularis	White Crappie	2.89 (99)	2.80 (71)
	Pomoxis nigromaculatus	Black Crappie	12.70 (435)	17.34 (440)
Clupeidae	*Dorosoma cepedianum*	Gizzard Shad	47.61 (1631)	4.37 (111)
Cyprinidae	*Cyprinella lutrensis*	Red Shiner	3.74 (128)	0.47 (12)
	Cyprinus carpio	Common Carp	3.42 (117)	15.45 (392)
	Notropis atherinoides	Emerald Shiner	0.44 (15)	-
Esocidae	*Esox lucius*	Northern Pike	0.38 (13)	-
Ictaluridae	*Ameiurus melas*	Black Bullhead	0.09 (3)	36.84 (935)
	Ictalurus punctatus	Channel Catfish	0.61 (21)	0.04 (1)
	Pylodictis olivaris	Flathead Catfish	0.03 (1)	-
Lepisosteidae	*Lepisosteus osseus*	Longnose Gar	0.03 (1)	-
	Lepisosteus platostomus	Shortnose Gar	5.60 (192)	0.24 (6)
Moronidae	*Morone chrysops*	White Bass	5.28 (181)	-
	Morone chrysops x saxatilis	Hybrid Striped Bass	0.58 (20)	-
Percidae	*Sander vitreus*	Walleye	1.17 (40)	-
Poeciliidae	*Gambusia affinis*	Western Mosquitofish	0.73 (25)	0.04 (1)
Sciaenidae	*Aplodinotus grunniens*	Freshwater Drum	12.69 (92)	0.04 (1)

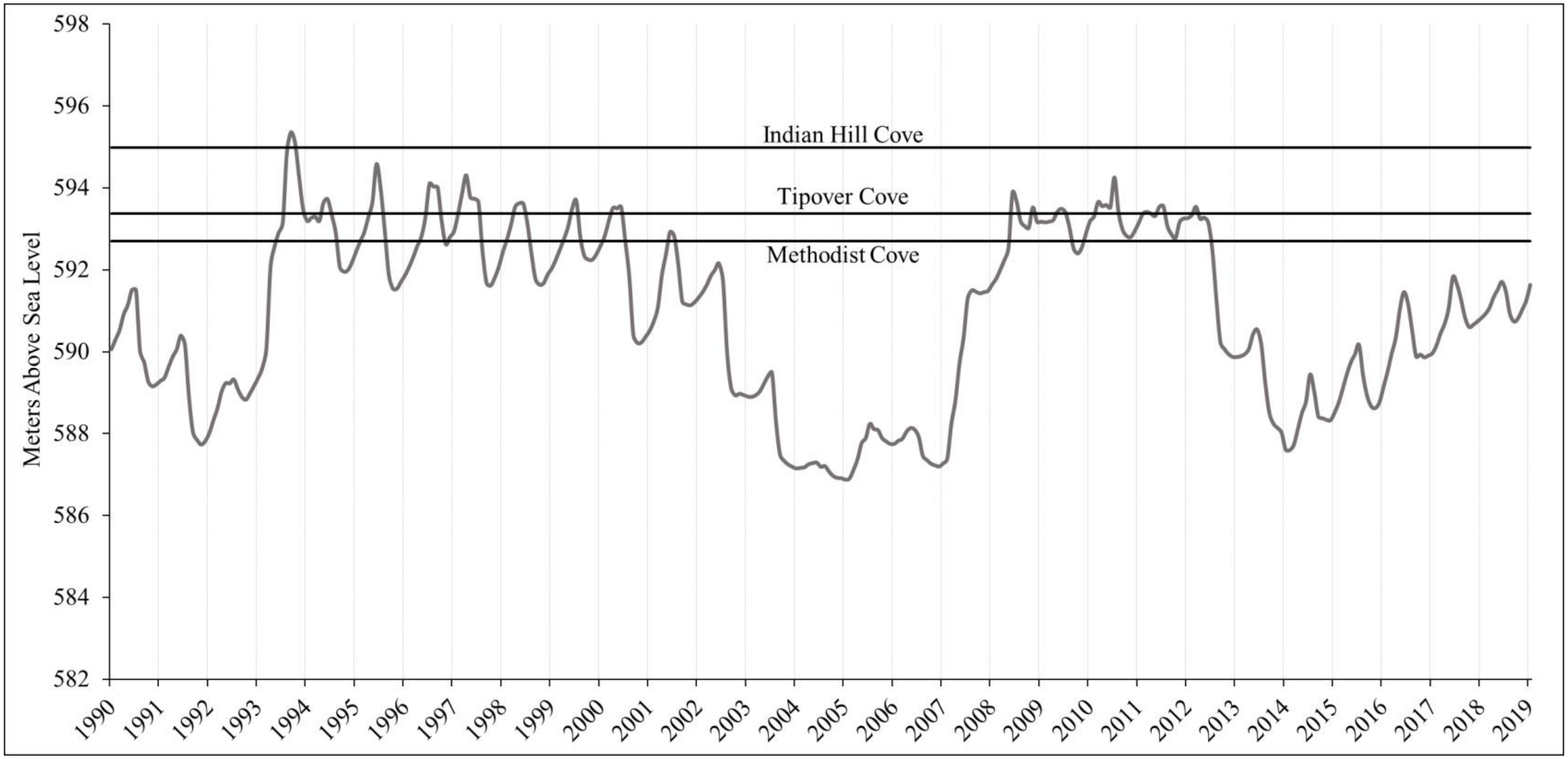

Figure 1. Elevation of the sediment berms disconnecting coves from the main reservoir, compared to end of month water level elevation of Harlan County Reservoir since January1990. The grey line indicates the water elevation of the main reservoir, recorded at the dam spillway (USBOR 2020). Horizontal black lines indicate minimum water level required for connection (heights of sediment berm plus 1 meter) for each corresponding cove. Vertical dotted lines indicate January 1st of the year labeled below.

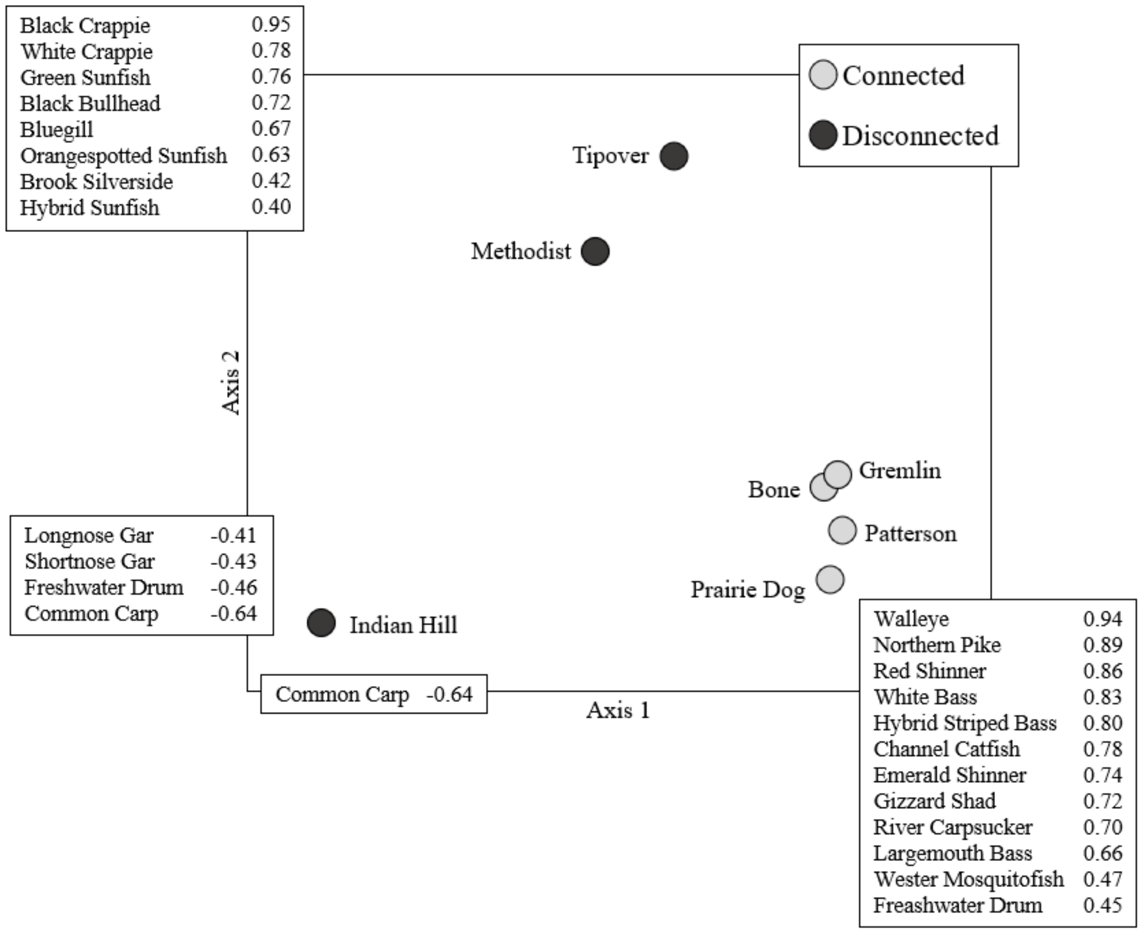

Figure 2. Metric multidimensional scale ordination of species abundance in connected and disconnected coves of Harlan County Reservoir. Insets tables on the axes show strong Spearman correlations ($r_s > 0.40$ or $r_s < -0.40$; Gido et al. 2009) between species abundance and ordination axes scores.

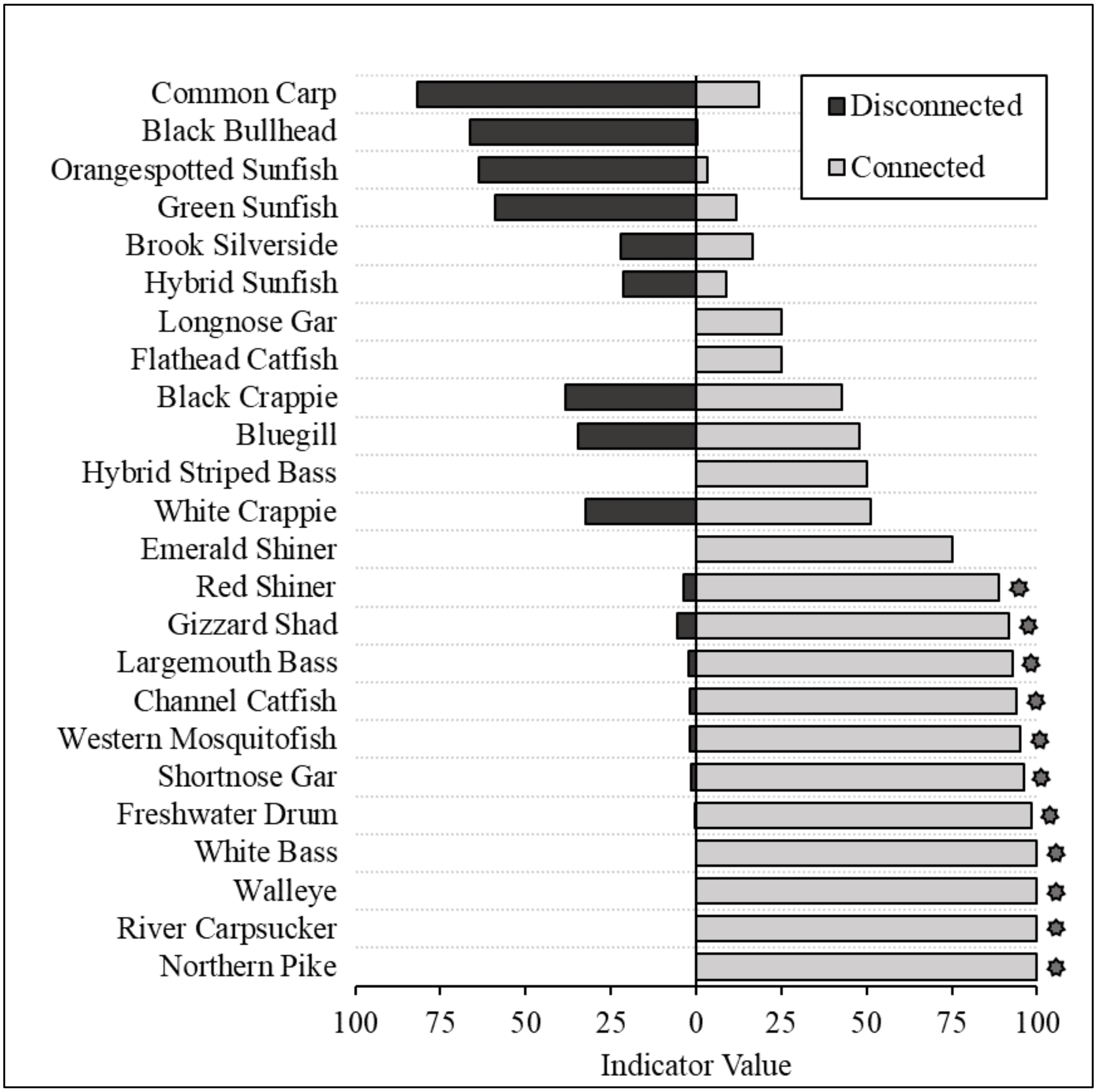

Figure 3. Results of indicator species analysis for connected and disconnected coves of Harlan County Reservoir. Indicator species for either habitat are denoted by an asterisk. Species were considered indicators if their indicator value for a particular habitat was ≥ 25 and had $p \leq 0.1$.

CHAPTER 4:

RESEARCH IMPLICATIONS AND MANAGEMENT RECOMMENDATIONS

Brian E. Mason

Introduction

Coves are part of the mosaic of habitats in reservoir systems, but surprisingly few studies have focused on understanding cove habitats that have become disconnected. As reservoirs age and water levels change within and between years, understanding how water quality, zooplankton, and fish communities change between coves that remain connected to the main reservoir and those that become disconnected can be informative for management of these systems. Within Harlan County Reservoir, disconnected coves had distinct water quality parameters and zooplankton assemblages compared to connected coves and the main reservoir (see Chapter 2). Disconnected coves also had unique fish communities compared to those connected to the main reservoir (see chapter 3). Furthermore, biotic communities (zooplankton and fish) were more variable in terms of biodiversity and relative abundances within disconnected coves, while connected coves had little inconsistency in these communities (See Chapters 2 and 3).

Cove Reconnection

Disconnected coves appear to have reduced water quality, which may lead to reduced biodiversity and population stability, suggesting efforts to reconnect or maintain connection of these cove habitats should be investigated. Reconnection of disconnected waterbodies is a relatively new approach to increase connectivity of backwater habitats and has been conducted primarily on river-floodplain systems, with little to no documented application on reservoir systems (Miranda 2017). Renovation projects to reconnect side channels of rivers that have been disconnected due to the buildup of natural levees allows for increased water, sediment, and nutrient flow between the two previously disconnected water bodies (Pess et al. 2005; Roni et al. 2005). Reconnecting side channels and oxbow lakes also can increase the total amount and diversity of available habitat, supporting additional fishes that utilize backwater habitats (Pess et al. 2005; Roni et al. 2005; Jackson and Pringle 2010). Similarly, reconnection of disconnected coves could allow reservoir sport fish populations access to these habitats and allow exchange of diverse fish assemblages between cove habitats and the main reservoir (Slipke and Maceina 2005; Miranda 2017). In addition to potentially enhancing the water quality and fish populations of the coves, renovation projects to maintain connection and reconnect disconnected coves would likely improve the access for boat anglers, potentially increasing fishing opportunities (Slipke and Maceina 2005).

Periodic higher water elevations may reconnect isolated coves, depending on the size of the sediment berm that isolates them and the extent of water level rise. Similar to other flood control and irrigation reservoirs, Harlan County Reservoir regularly experiences large variations in reservoir elevation due to inflow and irrigation demands throughout the year (Diffendal et al. 2002). These natural connection events could then lead to water quality changes in disconnected coves and allow for biotic exchange with the main reservoir, such that they more closely resemble connected coves from an ecological standpoint. Similar patterns were noted in Harlan County Reservoir in 2019 following higher-than-average precipitation throughout the Republican River watershed. During the spring and early summer, water elevations within the reservoir were approximately 597 msl, connecting all previously disconnected coves (Figure 1; USGS 2020). Although no statistical analyses have yet been conducted, an anecdotal review of collected data has shown decreased turbidity, decreased relative chlorophyll *a*, increased Secchi depths, reduced zooplankton densities, and increased fish species richness in these reconnected coves (Koupal et al. 2020). Interestingly, some fish species that were common within disconnected coves (e.g., Orangespotted Sunfish and Green Sunfish) were either absent or reduced in abundance, while other species that had not previously been found in disconnected coves (e.g., Northern Pike, White Bass, and Walleye), were sampled relatively frequently. Similar findings were noted in Lake Texoma, OK, when previously disconnected habitats had higher abundance of larval *Morone* spp. after reconnection (Gilbert and Pease 2019). These results indicate potential shifts in fish assemblages post-reconnection, with predatory pelagic species potentially taking advantage of a previously unavailable habitat and food sources. Such changes in water quality, zooplankton communities, and fish assemblages within disconnected coves of Harlan County Reservoir could be similar to those seen if artificial reconnection occurred.

Disconnected Cove Management

Reconnection of cove habitats may appear as a desirable option, however, there may be negative effects associated with this management strategy. Although disconnected coves have potentially lower densities of sport fish such as Largemouth Bass, White Bass, Walleye, and Northern Pike (Slipke and Maceina. 2005; Patton and Lyday 2008; see Chapter 3), these habitats may offer unique opportunities both ecologically and recreationally. Within Harlan County Reservoir, some native fish species (e.g., Orangespotted Sunfish, Green Sunfish, and Black Bullhead; Sowa et al.

2006) that have limited presence within the main reservoir are found in higher abundance in disconnected coves (see Chapter 3). Thus, disconnected coves could be potential sources of diversity for the main reservoir during years of high water, as connection would allow these species to move to other coves and recolonize other areas of the reservoir where local extirpation may have occurred.

Because disconnected coves may provide opportunities to catch fish not found in the main reservoir, these coves could be managed as separate and smaller waterbodies. Some issues such as habitat limitations or degraded water quality during at least some times of the year, however, may need to be addressed in order to sustain these fisheries. This study identified that disconnected coves were susceptible to poor water quality including high turbidity, warmer temperatures, and possible limitations of dissolved oxygen (See Chapter 2). Additionally, fish assemblages within disconnected coves tended to have higher proportions of generalist fish species tolerant of poor water quality; in fact, one cove (Indian Hill) only had Common Carp (see Chapter 3). Isolation time has been shown to potentially impact fish assemblages within disconnected coves (Patton and Lyday 2008; see Chapter 3), likely due to the ongoing reductions in water quality within these habitats over time. Benthivorous fish such as Common Carp and Black Bullhead can suspend sediments during feeding and can reduce submergent vegetation (Zambrano et al. 2001). Droughts and low water conditions within the main lake could also exaggerate water quality problems within disconnected coves (see Chapter 3), potentially accelerating reductions in species richness and diversity within disconnected coves if not reconnected or managed. Management strategies to resolve these issues could include aeration (Abdelrahman and Boyd 2018), removals of nuisance species (Zambrano et al. 2001), management for increased aquatic vegetation (Scheffer 1997), chemical treatments to reduce turbidity such as alum (Boyd 1979), and dredging to provide potential cooler temperature refugia (Miranda 2017).

Management of disconnected coves could be used to help achieve other management and conservation goals not possible solely within main reservoir. The conservation of Razorback Sucker within Lake Mohave, AZ, (Mueller 1995; Mueller 2006) is a good example of an innovative use of these habitats. To compensate for poor recruitment of Razorback sucker due to non-native predation, a disconnected cove (Yuma Cove) of Lake Mohave, AZ, has been used to propagate and rear this endangered species since the early 1990’s (Mueller 1995; Mueller 2006). Furthermore, the sediment berm isolating the habitat has been raised to maintain disconnection because of the value of Yuma Cove to the persistence of this species (Mueller 1995). Disconnected coves could also be used in propagation programs for sport fish that may be desirable in the main reservoir. Beyerle and Williams (1973) used marshes adjacent to Long Lake (Barry County, MI) as rearing and nursery habitat for the production of Norther Pike that were later stocked into the lake. A similar technique could be implemented within Harlan County Reservoir by either rearing a desired species within disconnected coves for later stocking into the main reservoir, or by simply transferring sport fish species in relatively high abundance within some disconnected coves (e.g., Black and White Crappie, and Bluegill) to the main reservoir. These strategies could make cove disconnection potentially beneficial with little effort or cost.

Disconnected coves could also provide diverse fishing opportunities. Because many centrarchid species, such as Bluegill and Crappie, are abundant in disconnected coves, these habitats could provide a resource for shoreline angling if reasonably accessible to public use. Infrastructure such as roads, trails or paths, restroom facilities, and fishing piers, would encourage this type of use, potentially making this a popular venue for the local community. Additionally, disconnected coves could serve those interested in micro-fishing, an untraditional angling practice where the goal is to angle and document as many species as possible, often including small or underutilized species (Cooke 2020). Disconnected coves could become a valuable resource for this emerging group of anglers, especially when these habitats have unique or uncommon species or fish communities, having potential to be managed and promoted as a form of ecotourism (Zwirn et al. 2005).

Future Directions and Research

In total, data collected in this study could be used to establish a baseline of the cove conditions within Harlan County Reservoir, for studying the abiotic and biotic changes that may occur due to the natural reconnection event of 2019, or for helping to plan potential renovation projects that re-establish connection in the future. Future studies should examine the succession of fish assemblages within disconnected coves over time, as this could be valuable information in determining longevity of these habitats. Long-term studies should include examining the effects of droughts and floods on disconnected cove water quality, zooplankton communities, and fish assemblages. Further, examining these parameters during winter could be beneficial in further understanding seasonal dynamics of disconnected coves. Additionally, sampling fish using additional gear types and extending sampling to include the main reservoir would allow for further comparison of these habitats. The information collected in this study can assist state and federal agencies working on aging reservoir systems to manage cove habitats with varying degrees of disconnection. When considering cove reconnection, managers should weigh the costs and benefits of available management strategies, as they likely vary widely between regions and specific objectives. As relatively little research has been done regarding cove disconnection and/or reconnection, additional research from other reservoirs in the U.S. and around the world is required to gain further understanding of these habitats and how they may change under various management strategies.

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Tables and Figures

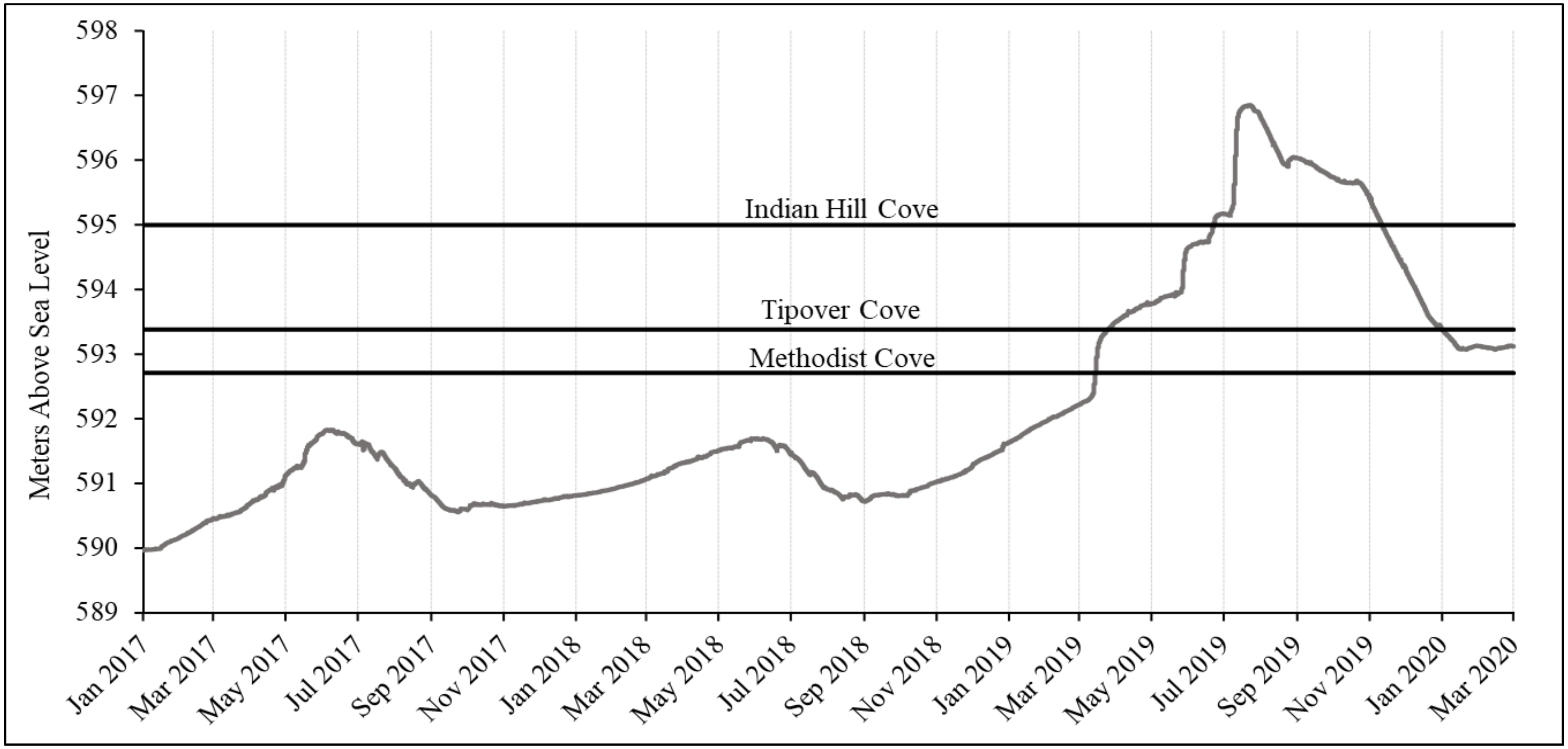

Figure 1. Elevation of the sediment berms disconnecting coves from the main reservoir, compared to hourly water level elevation of Harlan County Reservoir during the period of this study and the high-water event the following year. The grey line indicates the water elevation of the main reservoir (USGS 2020). Horizontal black lines indicate minimum water level required for connection (heights of sediment berm plus 1 meter) for each corresponding cove. Vertical dotted lines indicate 1st of the month labeled below.

APPENDIX 1:

WATER QUALITY PARAMETERS AND ZOOPLANKTON DENSITIES WITHIN DISCONNECTED COVES, CONNECTED COVES AND THE MAIN RESERVOIR OF HARLAN COUNTY RESERVOIR, NEBRASKA

Brian E. Mason

Tables

Table A1-1. Mean, minimum, and maximum readings for all water quality parameters tested and total zooplankton densities within disconnected coves, connected coves and the main reservoir for each season (spring, summer, and fall) in 2017. Numbers in parentheses represent one standard error.

		Spring			Summer			Fall		
		Disconnected	Connected	Main Reservoir	Disconnected	Connected	Main Reservoir	Disconnected	Connected	Main Reservoir
Temperature C°	Mean	18.2 (1.82)	17.9 (1.06)	17.4 (0.55)	26.6 (0.07)	25.5 (0.78)	25.6 (0.45)	19.5 (2.71)	17.2 (2.57)	13.4 (0.09)
	Minimum	15.7	16.3	16.7	26.5	23.8	24.9	14.1	12.0	13.3
	Maximum	21.8	21.0	18.5	26.7	27.5	26.4	22.8	21.7	13.6
Dissolved oxygen (mg/L)	Mean	7.5 (0.91)	7.9 (1.02)	9.1 (0.21)	2.7 (1.21)	5.6 (0.52)	5.4 (0.62)	10.3 (3.11)	10.0 (0.75)	9.3 (0.02)
	Minimum	6.1	5.2	8.8	1.5	4.5	4.2	7.2	8.6	9.3
	Maximum	9.2	9.8	9.5	3.9	6.8	6.4	16.5	11.7	9.3
Chlorophyll *a* (RFU)	Mean	74.7 (21.06)	112.7 (16.39)	34.2 (11.82)	164.1 (56.70)	69.0 (17.75)	26.1 (3.17)	259.3 (119.44)	112.1 (9.89)	45.1 (1.88)
	Minimum	40.1	68.1	16.9	74.1	28.7	22.5	72.5	92.6	42.8
	Maximum	112.8	138.4	56.8	268.8	111.5	32.4	481.6	137.1	48.8
Turbidity (FAU)	Mean	57 (10.02)	48 (5.11)	18 (3.96)	76 (5.85)	54 (5.19)	35 (5.15)	95 (23.83)	48 (7.72)	19 (2.67)
	Minimum	42	33	13	65	43	26	60	34	16
	Maximum	76	55	26	84	68	44	141	65	24
Secchi depth (cm)	Mean	37 (3.96)	47 (4.67)	58 (17.57)	26 (0.58)	24 (6.36)	82 (16.72)	30 (3.44)	50 (6.30)	70 (1.95)
	Minimum	29	40	35	25	13	58	26	39	67
	Maximum	42	61	93	27	42	114	37	67	74
pH	Mean	7.9 (0.21)	8.2 (0.17)	8.5 (0.10)	8.4 (0.59)	7.9 (0.25)	7.9 (0.37)	7.7 (0.70)	8.3 (0.18)	8.7 (0.05)
	Minimum	7.6	7.7	8.4	7.6	7.4	7.3	6.8	7.8	8.6
	Maximum	8.3	8.5	8.7	9.6	8.6	8.6	9.0	8.6	8.7
Total zooplankton (number / L)	Mean	293 (135.17)	64 (17.87)	44 (4.69)	821 (450.52)	122 (14.75)	86 (25.65)	532 (393.00)	133 (79.86)	49 (5.22)
	Minimum	81	27	34	330	79	54	117	29	39
	Maximum	544	105	49	1720	141	137	1318	370	56

Table A1-2. Mean, minimum, and maximum readings for all water quality parameters tested and total zooplankton densities within disconnected coves, connected coves and the main reservoir for each season (spring, summer, and fall) in 2018. Numbers in parentheses represent one standard error.

		Spring			Summer			Fall		
		Disconnected	Connected	Main Reservoir	Disconnected	Connected	Main Reservoir	Disconnected	Connected	Main Reservoir
Temperature C°	Mean	20.9 (0.42)	18.7 (0.85)	16.6 (0.05)	28.0 (0.05)	27.2 (0.13)	26.3 (0.22)	18.1 (3.08)	20.8 (1.19)	20.8 (0.18)
	Minimum	20.1	16.7	16.5	27.9	26.8	26.0	12.5	18.9	20.6
	Maximum	21.4	20.1	16.6	28.1	27.4	26.7	23.0	24.3	21.2
Dissolved oxygen (mg/L)	Mean	8.0 (0.36)	9.8 (0.68)	8.4 (0.10)	5.3 (1.75)	6.0 (0.79)	6.9 (0.18)	7.8 (1.25)	7.3 (0.08)	7.1 (0.15)
	Minimum	7.3	8.3	8.2	3.4	3.9	6.7	5.6	7.0	6.9
	Maximum	8.5	11.2	8.5	8.8	7.7	7.3	10.0	7.4	7.4
Chlorophyll *a* (RFU)	Mean	148.3 (41.43)	83.7 (14.80)	32.5 (7.52)	134.1 (15.04)	65.1 (10.35)	37.1 (4.68)	205.39 (87.82)	75.7 (8.12)	52.1 (1.59)
	Minimum	79.1	66.5	19.9	106.4	36.6	30.7	103.6	59.6	48.9
	Maximum	222.4	128.0	45.9	158.1	85.8	46.2	380.3	94.2	54.1
Turbidity (FAU)	Mean	92 (9.10)	33 (1.79)	10 (1.59)	93 (30.88)	49 (10.05)	30 (2.75)	124 (33.15)	52 (10.08)	32 (1.94)
	Minimum	75	29	7	52	33	27	71	36	29
	Maximum	107	37	13	153	78	35	185	79	35
Secchi depth (cm)	Mean	28 (1.45)	63 (6.84)	187 (42.74)	32 (11.16)	46 (6.91)	52 (4.02)	19 (4.76)	46 (8.00)	55 (1.76)
	Minimum	26	51	122	18	26	44	14	27	53
	Maximum	31	81	268	54	55	56	29	60	58
pH	Mean	7.7 (0.11)	8.2 (0.04)	8.3 (0.02)	8.2 (0.48)	8.1 (0.14)	8.2 (0.02)	8.1 (0.54)	8.3 (0.05)	8.3 (0.03)
	Minimum	7.5	8.2	8.3	7.5	7.8	8.2	7.5	8.2	8.3
	Maximum	7.9	8.3	8.3	9.1	8.4	8.3	9.2	8.4	8.4
Nitrate (mg/L)	Mean	0.3 (0.11)	0.3 (0.01)	0.4 (0.02)	0.2 (0.14)	0.3 (0.08)	0.3 (0.08)	0.2 (0.09)	0.2 (0.07)	0.3 (0.03)
	Minimum	0.2	0.3	0.4	0.0	0.1	0.2	0.1	0.1	0.3
	Maximum	0.6	0.3	0.4	0.5	0.4	0.4	0.3	0.4	0.4
Phosphate (mg/L)	Mean	4.0 (1.96)	0.5 (0.17)	0.8 (0.41)	0.9 (0.29)	3.5 (2.15)	2.2 (1.11)	1.8 (1.45)	1.5 (0.23)	0.7 (0.38)
	Minimum	1.4	0.2	0.2	0.3	1.3	0.3	0.2	0.9	0.3
	Maximum	7.9	0.7	1.6	1.3	8.5	4.2	4.7	2.0	1.5
Total zooplankton (number / L)	Mean	1266 (171.02)	144 (36.48)	86 (2.10)	4391 (2101.97)	370 (245.97)	111 (20.25)	3528 (2599.21)	256 (93.94)	66 (13.41)
	Minimum	966	67	84	1145	49	84	398	87	47
	Maximum	1558	238	90	8328	1090	151	8688	525	92

Table A1-3. Comparisons of annual means of all water quality parameters total zooplankton densities across habitat types (connected and disconnected coves and main reservoir) and seasons (spring, summer, and fall). Numbers in parentheses represent one standard error. General linear mixed models (SAS Version 9.4) were used to compare means between years ($\alpha = 0.10$).

Water parameter	2017	2018	*P*
Temperature C°	19.9 (0.91)	22.0 (0.79)	0.03*
Dissolved oxygen (mg/L)	7.7 (0.53)	7.4 (0.32)	0.62
Chlorophyll *a* (RFU)	99.5 (16.97)	90.9 (12.92)	0.96
Turbidity (FAU)	50 (4.93)	56 (7.55)	0.89
Secchi depth (cm)	46 (4.42)	58 (9.26)	0.29
pH	8.2 (0.11)	8.2 (0.07)	0.93
Nitrate (mg/L)	-	0.3 (0.02)	-
Phosphate (mg/L)	-	1.8 (0.38)	-
Total zooplankton (number / L)	225 (68.91)	1048 (395.55)	< 0.01*

Table A1-4. Comparisons of seasonal (spring, summer, and fall) means of all water quality parameters total zooplankton densities across habitat types (connected and disconnected coves and main reservoir) and years (2017 and 2018). Numbers in parentheses represent one standard error. General linear mixed models (SAS Version 9.4) were used to compare means between seasons, and letters denote significant differences between seasons based on a Tukey test (α = 0.10)

Water parameter	Spring	Summer	Fall	*P*
Temperature C°	18.3 (0.45) ^{a}	26.5 (0.27) ^{b}	18.4 (0.92) ^{a}	< 0.01*
Dissolved oxygen (mg/L)	8.5 (0.31)^{b}	5.5 (0.41)^{a}	8.6 (0.53)^{b}	< 0.01*
Chlorophyll *a* (RFU)	82.7 (11.62) ^{a}	81.0 (13.72) ^{a}	122 (25.79) ^{b}	0.02*
Turbidity (FAU)	43 (6.30) ^{a}	56 (6.58) ^{b}	60 (9.77) ^{b}	< 0.01*
Secchi depth (cm)	69 (13.21) ^{b}	44 (5.60) ^{a}	45 (4.11) ^{a}	0.01*
pH	8.1 (0.07)	8.1 (0.12)	8.2 (0.13)	0.77
Nitrate (mg/L)	0.3 (0.03)	0.3 (0.05)	0.3 (0.04)	0.37
Phosphate (mg/L)	1.6 (0.74)	2.3 (0.78)	1.3 (0.42)	0.34
Total zooplankton (number / L)	295 (99.42)^{a}	910 (436.64)^{b}	704 (430.05)^{a,b}	0.08*

Table A1-5. Comparisons of means of all water quality parameters total zooplankton densities between habitat types (connected and disconnected coves and main reservoir) and across seasons (spring, summer, and fall) and years (2017 and 2018). Numbers in parentheses represent one standard error. General linear mixed models (SAS Version 9.4) were used to compare means between habitat types (α = 0.10), and letters denote significant differences between habitat types based on a Tukey test (α = 0.10)

Water parameter	Disconnected	Connected	Main Reservoir	*P*
Temperature C°	21.6 (1.18) [b]	21.2 (0.93) [a,b]	20.0 (1.15) [a]	0.07*
Dissolved oxygen (mg/L)	7.2 (0.81)	7.8 (0.43)	7.7 (0.35)	0.41
Chlorophyll *a* (RFU)	164.3 (27.13) [c]	86.4 (6.29) [b]	37.8 (2.98) [a]	< 0.01*
Turbidity (FAU)	90 (8.94) [c]	47 (2.99) [b]	24 (2.47) [a]	< 0.01*
Secchi depth (cm)	29 (2.42) [a]	46 (3.38) [b]	84 (13.41) [c]	< 0.01*
pH	8.0 (0.17)	8.2 (0.06)	8.3 (0.08)	0.16
Nitrate (mg/L)	0.3 (0.06)	0.3 (0.03)	0.3 (0.03)	0.40
Phosphate (mg/L)	2.5 (0.90)	1.8 (0.64)	1.3 (0.44)	0.58
Total zooplankton (number / L)	1805 (610.24)[c]	182 (46.23)[b]	74 (7.59)[a]	< 0.01*

Table A1-6. Comparisons of means of all water quality parameters total zooplankton densities between habitat types (connected and disconnected coves and main reservoir) within each season (spring, summer, and fall). Data was combined across years (2017 and 2018) when available. Numbers in parentheses represent one standard error. General linear mixed models (SAS Version 9.4) were used to evaluate whether an interaction effect between season and habitat type occurred, and letters denote significant differences between habitat types within each season based on a Tukey test ($\alpha = 0.10$).

Season	Habitat type	Temperature C°	Dissolved oxygen (mg/L)	Chlorophyll *a* (RFU)	Turbidity (FAU)	Secchi depth (cm)
Spring	Disconnected	19.0 (1.03)	7.5 (0.45)	75.8 (26.52)	69 (9.95) [c]	34 (2.69)
	Connected	17.9 (0.65)	7.9 (0.67)	112.7 (11.60)	48 (3.75) [b]	47 (4.83)
	Main Reservoir	17.2 (0.31)	9.0 (0.20)	30.6 (6.73)	15 (2.58) [a]	111 (35.47)
Summer	Disconnected	27.1 (0.35)	4.7 (1.22)	162.6 (27.08)	75 (14.54) [c]	26 (6.32)
	Connected	25.5 (0.48)	5.6 (0.44)	69.0 (9.54)	54 (5.32) [b]	24 (6.06)
	Main Reservoir	25.7 (0.28)	5.7 (0.45)	28.2 (3.53)	33 (2.88) [a]	75 (10.22)
Fall	Disconnected	20.3 (1.86)	10.2 (1.60)	227.5 (67.38)	100 (19.33) [c]	26 (3.59)
	Connected	17.2 (1.48)	10.0 (0.62)	112.2 (9.09)	48 (5.92) [b]	50 (4.78)
	Main Reservoir	15.4 (1.66)	8.7 (0.49)	46.0 (1.91)	23 (3.39) [a]	67 (3.71)
	P	0.93	0.93	0.17	0.03*	0.38

Table A1-6 Continued.

Season	Habitat type	pH	Nitrate (mg/L)	Phosphate (mg/L)	Total zooplankton (number/L)
Spring	Disconnected	7.8 (0.11)	0.3 (0.11)	4.0 (1.96) [b]	609 (238.37)
	Connected	8.2 (0.08)	0.3 (0.01)	0.5 (0.17) [a]	64 (24.11)
	Main Reservoir	8.5 (0.07)	0.4 (0.02)	0.8 (0.41) [a]	54 (9.77)
Summer	Disconnected	8.6 (0.34)	0.2 (0.14)	0.9 (0.29) [a]	2697 (1249.71)
	Connected	7.9 (0.14)	0.3 (0.08)	3.5 (2.15) [b]	122 (123.33)
	Main Reservoir	8.0 (0.18)	0.3 (0.08)	2.2 (1.11) [a,b]	89 (15.69)
Fall	Disconnected	8.0 (0.41)	0.2 (0.09)	1.8 (1.45) [a]	2571 (1353.12)
	Connected	8.3 (0.09)	0.2 (0.07)	1.5 (0.23) [a]	133 (61.67)
	Main Reservoir	8.6 (0.08)	0.3 (0.03)	0.7 (0.38) [a]	52 (7.47)
	P	0.20	0.99	0.04*	0.76

Table A1-7. Comparisons of means of all water quality parameters total zooplankton densities between season (spring, summer, and fall) within each habitat types (connected and disconnected coves and main reservoir). Data was combined across years (2017 and 2018) when available. Numbers in parentheses represent one standard error. General linear mixed models (SAS Version 9.4) were used to evaluate whether an interaction effect between season and habitat type occurred, and letters denote significant differences between seasons within each habitat type based on a Tukey test ($\alpha = 0.10$).

Habitat type	Season	Temperature C°	Dissolved oxygen (mg/L)	Chlorophyll *a* (RFU)	Turbidity (FAU)	Secchi depth (cm)
Disconnected	Spring	19.0 (1.03)	7.5 (0.45)	75.8 (26.52)	69.2 (9.95) [a]	34 (2.69)
	Summer	27.1 (0.35)	4.7 (1.22)	162.6 (27.08)	75.2 (14.54) [a,b]	26 (6.32)
	Fall	20.3 (1.86)	10.2 (1.60)	227.5 (67.38)	100.5 (19.33) [b]	26 (3.59)
Connected	Spring	17.9 (0.65)	7.9 (0.67)	112.7 (11.60)	47.8 (3.75) [a]	47 (4.83)
	Summer	25.5 (0.48)	5.6 (0.44)	69.0 (9.54)	54.4 (5.32) [a]	24 (6.06)
	Fall	17.2 (1.48)	10.0 (0.62)	112.2 (9.09)	47.9 (5.92) [a]	50 (4.78)
Main Reservoir	Spring	17.2 (0.31)	9.0 (0.20)	30.6 (6.73)	14.9 (2.58) [a]	111 (35.47)
	Summer	25.7 (0.28)	5.7 (0.45)	28.2 (3.53)	33.2 (2.88) [b]	75 (10.22)
	Fall	15.4 (1.66)	8.7 (0.49)	46.0 (1.91)	22.7 (3.39) [b]	67 (3.71)
	P	0.93	0.93	0.17	0.03*	0.38

Table A1-7 Continued.

Habitat type	Season	pH	Nitrate (mg/L)	Phosphate (mg/L)	Total zooplankton (number/L)
Disconnected	Spring	7.8 (0.11)	0.3 (0.11)	4.0 (1.96) ^{b}	609 (238.37)
	Summer	8.6 (0.34)	0.2 (0.14)	0.9 (0.29) ^{a}	2697 (1249.71)
	Fall	8.0 (0.41)	0.2 (0.09)	1.8 (1.45) ^{a}	2571 (1353.12)
Connected	Spring	8.2 (0.08)	0.3 (0.01)	0.5 (0.17) ^{a}	64 (24.11)
	Summer	7.9 (0.14)	0.3 (0.08)	3.5 (2.15) ^{b}	122 (123.33)
	Fall	8.3 (0.09)	0.2 (0.07)	1.5 (0.23) ^{b}	133 (61.67)
Main Reservoir	Spring	8.5 (0.07)	0.4 (0.02)	0.8 (0.41) ^{a}	54 (9.77)
	Summer	8.0 (0.18)	0.3 (0.08)	2.2 (1.11) ^{a}	89 (15.69)
	Fall	8.6 (0.08)	0.3 (0.03)	0.7 (0.38) ^{a}	52 (7.47)
	P	0.20	0.99	0.04*	0.76

APPENDIX 2:

ASSESSMENT OF BLACK CRAPPIE GROWTH AND CONDITION WITHIN DISCONNECTED AND CONNECTED COVES OF HARLAN COUNTY RESERVOIR, NEBRASKA

Brian E. Mason

Objectives

Black and White Crappie (*Pomoxis nigromaculatus* and *P. annularis*) are popular sport fish commonly found within reservoir littoral areas and often utilize cove environments for spawning and nursery habitat (Meals and Miranda 1991; Warren 2009). As reservoirs age, however, sedimentation and erosion may create disconnections between reservoirs and their coves. Population demographics and dynamics of fish species inhabiting both disconnected and connected cove habitats could, thus, differ between cove types. For example, McInerny and Cross (2008) found that small impoundments produced shorter Black Crappie (*Pomoxis nigromaculatus*) compared to larger natural lakes and reservoirs throughout Minnesota, likely due to differences in food availability and predator densities. Because of their similarities to small impoundments, growth patterns and population demographics of the fish species within disconnected coves may behave in a similar manner. The objective of this appendix is to compare mean length and relative weight of age 1, 2, and 3 year-old Black and White Crappie between connected and disconnected coves in Harlan County Reservoir.

Methods

Sampling

Fish were collected for this study in tandem other fish as described in Chapter 3. Ten Black Crappie and ten White Crappie from each 1-cm length bin; fish were euthanized in accordance to approved procedures (UNK Institutional Animal Care and Use Committee Approval #032918). Euthanized fish were then transported back to the laboratory where their weight (g) and total length (TL; mm) were recorded, their gonads were examined to determine sex, and otoliths were extracted to determine age. Both Otoliths were individually set in clear acrylic epoxy, sectioned using a rotary saw, and examined under a Motic model BA410 microscope to determine age.

Data analysis

Data analysis was only conducted on crappies collected during spring sampling. Mean total length and relative weight (W_r; Neumann and Murphy 1991) at each age was compared by cove type using a two-way analysis of variance (ANOVA), followed by a Tukey test ($\alpha < 0.05$). Due to the overall low abundance of White Crappie sampled in this study, only Black Crappie were included in this analysis. Annual mean length, mean weights, and fish counts for both Black Crappie and White Crappie at all ages and cove types are provided in tables A2-1 through A2-6.

Results

Mean length at age of Black Crappie increased significantly for each older year class ($F = 480.46$; $p < 0.01$; Figure A2-1). Conversely, the mean W_r at age significantly decreased for each older year class ($F = 45.12$; $p < 0.01$; Figure A2-2). Black Crappie sampled in disconnected coves were significantly shorter ($F = 94.43$; $p < 0.01$; Figure A2-3) and had significantly lower W_r ($F = 10.45$; $p < 0.01$; Figure A2-4) than those from connected coves. Differences were also noted between disconnected and connected coves within age groups in both length ($F = 29.24$; $p < 0.01$) and W_r ($F = 5.49$; $p < 0.01$). Black Crappie at ages 2 and 3 were significantly shorter in disconnected versus connected coves

(Figures A2-5), and fish at age 1 and 2 were in poorer condition in disconnected coves compared to connected coves (Figure A2-16).

Conclusions

Differences in diet and food availability could influence the length and W_r at age of both habitats. Although disconnected coves contained higher densities of zooplankton compared to connected coves and the main reservoir, zooplankton community structure differed between the different habitats; connected coves and the main reservoir having higher relative densities of crustacean zooplankton such as Calanoida and *Daphnia*, and disconnected coves were dominated by Rotifers (see Chapter 2). While Rotifers are an important food source for numerous larval fish species, they are often negligibly utilized by fish at later life stages (Lubzens et al. 1989; DeVries et al. 1998). The high concentration of rotifers within disconnected coves may result in limited alternative zooplankton foods sources for growing fish. Additionally, disconnected coves may have reduced access to larval Gizzard Shad that are prevalent within the main reservoir during the summer and are a major food resources for reservoir fishes (Sullivan et al. 2011; Miller et al. 2019). This could limit a potentially beneficial food source for Black Crappie within disconnected coves, however, further research is needed to determine the extent that larval Gizzard Shad are utilized by crappie.

The differences in mean length and W_r at age *Pomoxis spp.* between disconnected and connected coves of Harlan County Reservoir may be a result of other factors as well. Disconnected coves had observed higher water temperature and potentially decreased dissolved oxygen during parts of the year compared to connected coves (see Chapter 2). These parameters within disconnected coves could subject fish within these habitats to seasonal stress, thus affecting growth (Abdel-Tawwab et al. 2019). Additionally, fish in disconnected coves appeared to have higher parasite loads than those in connected coves, revealed at the time of sex determination. Parasites could potentially further stress afflicted fish, depriving them of nutrients, limiting growth, and reducing condition (Hugghins 1959). Furthermore, while fish densities within the different habitats were not directly measured within this study, higher densities in small, confined habitats may limit available resources, potentially affecting growth.

Results from this study may be used to evaluate the influence of connection to reservoirs on fish community characteristics within coves. In this study specifically focuses on growth and body condition of Black Crappie as an indicator species because this species is known to affiliate more with coves rather than the main reservoir. Cove reconnection could lead to increased growth rates and improved body condition of Black Crappie as reconnection may lead to improvements in water quality and food availability. Future studies could examine growth and condition of other species found commonly within both habitats (such as White Crappie and Bluegill) in order to predict how other species may respond to reconnection of cove habitats to the main reservoir, whether by natural or artificial means.

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Tables and Figures

Table A2-1. Mean total length (mm), mean weight (g), and fish counts by age for Black Crappie collected within connected and disconnected coves during 2017. Fish form only the spring sampling period were taken for aging during 2017. Numbers in parentheses represent one standard error. Dash marks indicate that no fish were sampled with the indicated age.

	Connected			Disconnected		
Age	Count	Total length (mm)	Weight (g)	Count	Total length (mm)	Weight (g)
0	0	-	-	0	-	-
1	5	114 (6.22)	24 (3.81)	23	112 (1.18)	21 (0.62)
2	35	173 (1.63)	88 (2.07)	11	135 (5.59)	39 (3.51)
3	5	238 (15.95)	237 (39.17)	1	138 (N/A)	34.5 (N/A)
4	0	-	-	15	166 (5.44)	73 (9.28)
5	1	280 (N/A)	328 (N/A)	0	-	-
6	1	290 (N/A)	456 (N/A)	0	-	-
7	0	-	-	0	-	-
8	1	296 (N/A)	384 (N/A)	0	-	-
9	0	-	-	0	-	-
10	0	-	-	0	-	-
11	0	-	-	0	-	-

Table A2-2. Mean total length (mm), mean weight (g), and fish counts by age for White Crappie collected within connected and disconnected coves during 2017. Fish form only the spring sampling period were taken for aging during 2017. Numbers in parentheses represent one standard error. Dash marks indicate that no fish were sampled with the indicated age.

	Connected			Disconnected		
Age	Count	Total length (mm)	Weight (g)	Count	Total length (mm)	Weight (g)
0	0	-	-	0	-	-
1	0	-	-	3	119 (5.90)	23 (4.07)
2	9	204 (6.80)	126 (10.96)	10	162 (3.55)	53 (4.31)
3	1	261 (N/A)	290 (N/A)	0	-	-
4	1	296 (N/A)	384 (N/A)	1	288 (N/A)	411 (N/A)
5	2	318 (7.50)	454 (10.00)	0	-	-
6	2	344 (7.50)	641 (11.25)	0	-	-
7	0	-	-	0	-	-
8	0	-	-	0	-	-
9	0	-	-	0	-	-
10	0	-	-	0	-	-
11	0	-	-	0	-	-

Table A2-3. Mean total length (mm), mean weight (g), and fish counts by age for Black Crappie collected within connected and disconnected coves during 2018. Fish form the spring, summer, and fall sampling periods were taken for aging during 2018. Numbers in parentheses represent one standard error. Dash marks indicate that no fish were sampled with the indicated age.

	Connected			Disconnected		
Age	Count	Total length (mm)	Weight (g)	Count	Total length (mm)	Weight (g)
0	84	63 (2.31)	4 (0.50)	13	80 (8.79)	10 (2.15)
1	39	146 (3.57)	47 (3.23)	69	129 (2.80)	32 (1.92)
2	55	196 (2.76)	122 (4.98)	76	168 (2.20)	71 (2.99)
3	40	222 (2.24)	177 (6.71)	21	180 (4.31)	87 (7.41)
4	3	266 (8.08)	325 (36.83)	0	-	-
5	1	230 (N/A)	236 (N/A)	35	196 (4.46)	116 (10.94)
6	0	-	-	0	-	-
7	1	290 (N/A)	444 (N/A)	0	-	-
8	2	309.5 (0.50)	492 (36.00)	0	-	-
9	1	310 (N/A)	478 (N/A)	0	-	-
10	0	-	-	0	-	-
11	0	-	-	0	-	-

Table A2-4. Mean total length (mm), mean weight (g), and fish counts by age for White Crappie collected within connected and disconnected coves during 2018. Fish form the spring, summer, and fall sampling periods were taken for aging during 2018. Numbers in parentheses represent one standard error. Dash marks indicate that no fish were sampled with the indicated age.

	Connected			Disconnected		
Age	Count	Total length (mm)	Weight (g)	Count	Total length (mm)	Weight (g)
0	8	83 (9.57)	8 (2.16)	5	129 (2.96)	24 (1.60)
1	13	201 (5.94)	97 (13.53)	16	187 (3.85)	82 (5.15)
2	6	248 (8.74)	205 (28.61)	3	202 (1.67)	119 (11.10)
3	3	277 (28.48)	409 (146.95)	11	250 (5.95)	238 (21.38)
4	0	-	-	0	-	-
5	0	-	-	1	230 (N/A)	168 (N/A)
6	1	340 (N/A)	660	0	-	-
7	1	340 (N/A)	636	0	-	-
8	0	-	-	0	-	-
9	0	-	-	0	-	-
10	0	-	-	0	-	-
11	2	348 (2.00)	623 (63)	0	-	-

Table A2-5. Mean total length (mm), mean weight (g), and fish counts by age for Black Crappie collected within connected and disconnected coves during 2019. Fish form the spring, summer, and fall sampling periods were taken for aging during 2019. Numbers in parentheses represent one standard error. Dash marks indicate that no fish were sampled with the indicated age.

	Connected			Disconnected		
Age	Count	Total length (mm)	Weight (g)	Count	Total length (mm)	Weight (g)
0	133	97 (2.29)	15 (0.71)	83	96 (2.98)	16 (1.05)
1	53	186 (1.98)	108 (3.17)	21	190 (2.49)	118 (5.62)
2	11	202 (6.77)	149 (17.23)	16	194 (7.08)	126 (13.37)
3	39	224 (4.52)	200 (15.10)	8	240 (7.88)	250 (28.21)
4	1	223 (N/A)	162 (N/A)	6	225 (14.42)	196 (49.03)
5	0	-	-	0	-	-
6	1	233 (N/A)	188 (N/A)	5	221 (12.97)	166 (26.45)
7	0	-	-	0	-	-
8	0	-	-	0	-	-
9	1	303 (N/A)	410 (N/A)	0	-	-
10	0	-	-	0	-	-
11	0	-	-	0	-	-

Table A2-6. Mean total length (mm), mean weight (g), and fish counts by age for White Crappie collected within connected and disconnected coves during 2019. Fish form the spring, summer, and fall sampling periods were taken for aging during 2019. Numbers in parentheses represent one standard error. Dash marks indicate that no fish were sampled with the indicated age.

	Connected			Disconnected		
Age	Count	Total length (mm)	Weight (g)	Count	Total length (mm)	Weight (g)
0	0	-	-	0	-	-
1	1	122 (N/A)	24 (N/A)	0	-	-
2	1	282 (N/A)	325 (N/A)	3	227 (3.48)	154 (8.72)
3	2	268 (0.50)	255 (13.00)	2	273 (1.50)	269 (15.00)
4	0	-	-	1	310 (N/A)	444 (N/A)
5	0	-	-	0	-	-
6	0	-	-	0	-	-
7	0	-	-	0	-	-
8	0	-	-	0	-	-
9	0	-	-	0	-	-
10	0	-	-	0	-	-
11	0	-	-	0	-	-

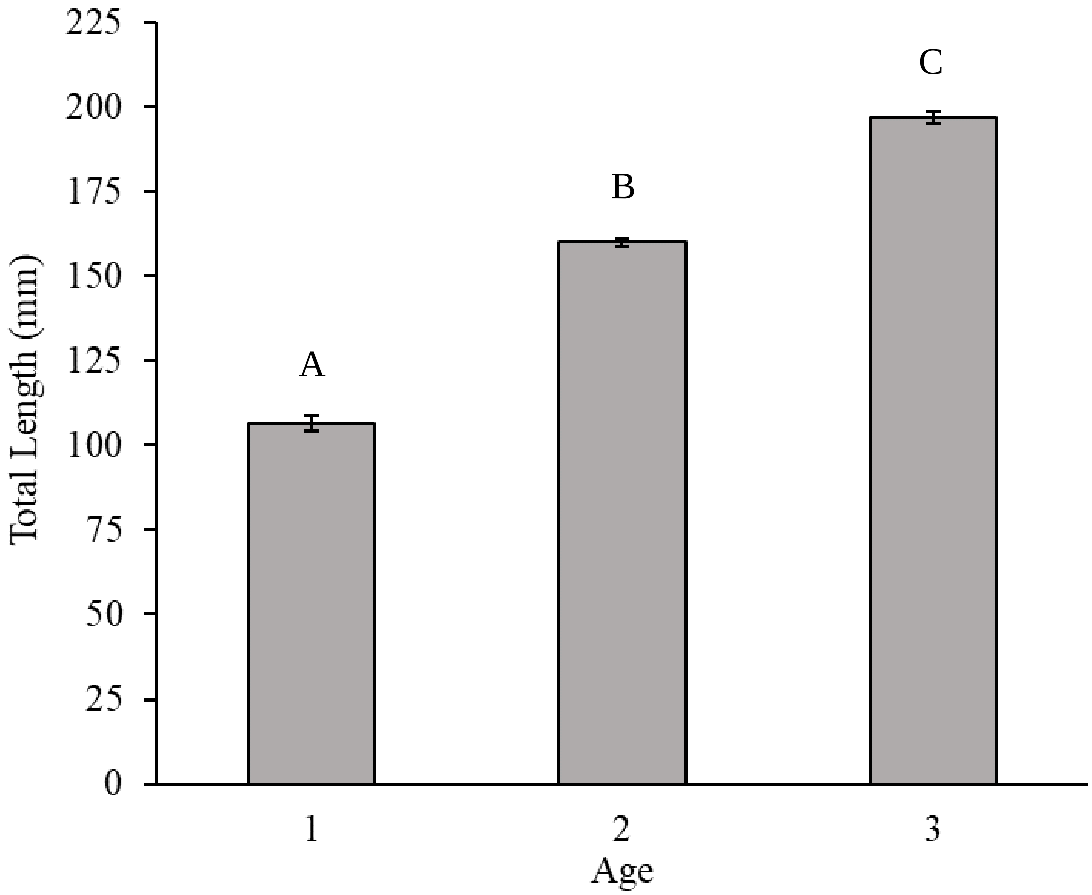

Figure A2-1. Mean total length of Black Crappie at ages 1, 2, and 3 across both disconnected and connected coves of Harlan County Reservoir. Error bars denote one standard error. Letters above the bars denote significant differences based on a Tukey test ($\alpha = 0.05$).

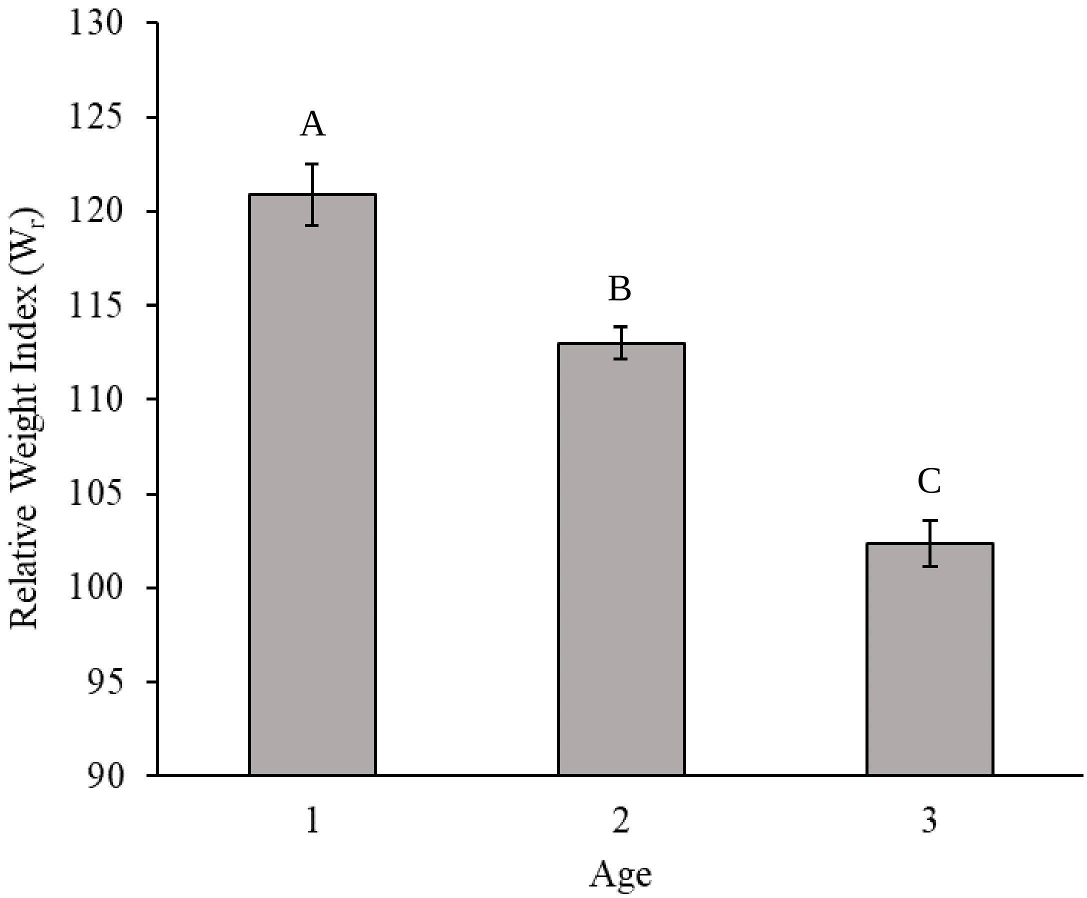

Figure A2-2. Mean W_r of Black Crappie at ages 1, 2, and 3 across both disconnected and connected coves of Harlan County Reservoir. Error bars denote one standard error. Letters above the bars denote significant differences based on a Tukey test ($\alpha = 0.05$).

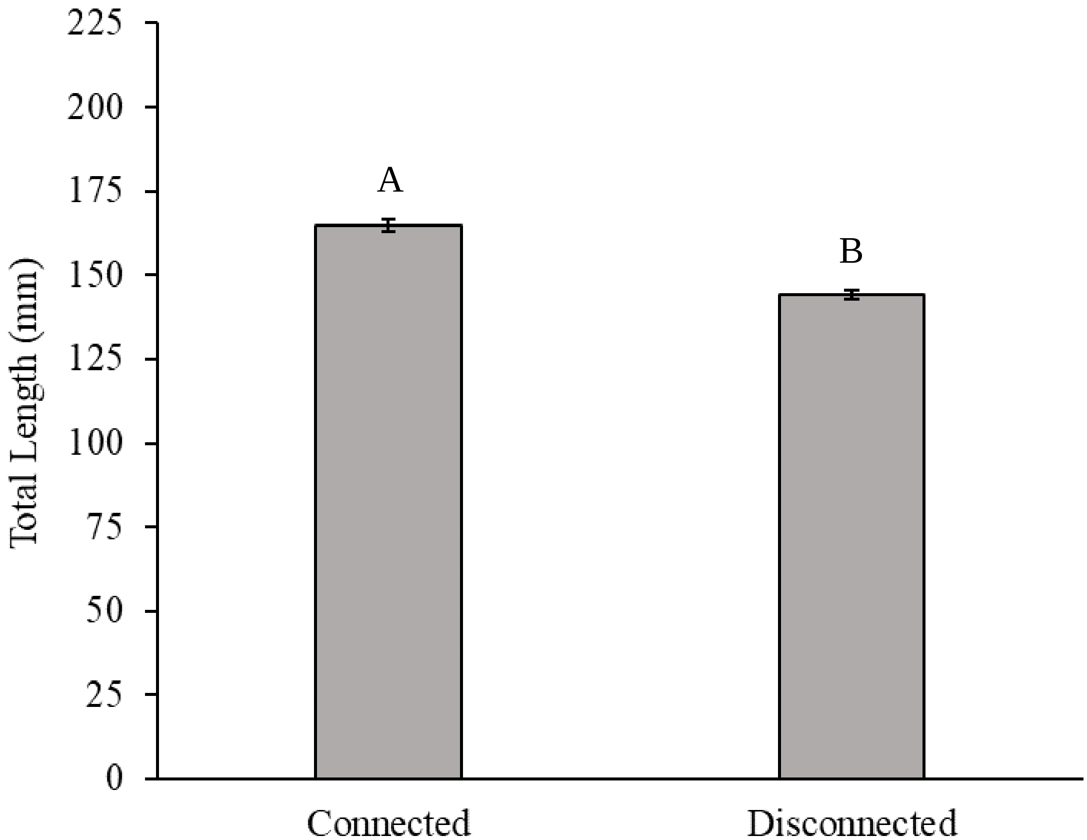

Figure A2-3. Mean total length of Black Crappie at within both disconnected and connected coves of Harlan County Reservoir across ages 1, 2, and 3. Error bars denote one standard error. Letters above the bars denote significant differences based on a Tukey test ($\alpha = 0.05$).

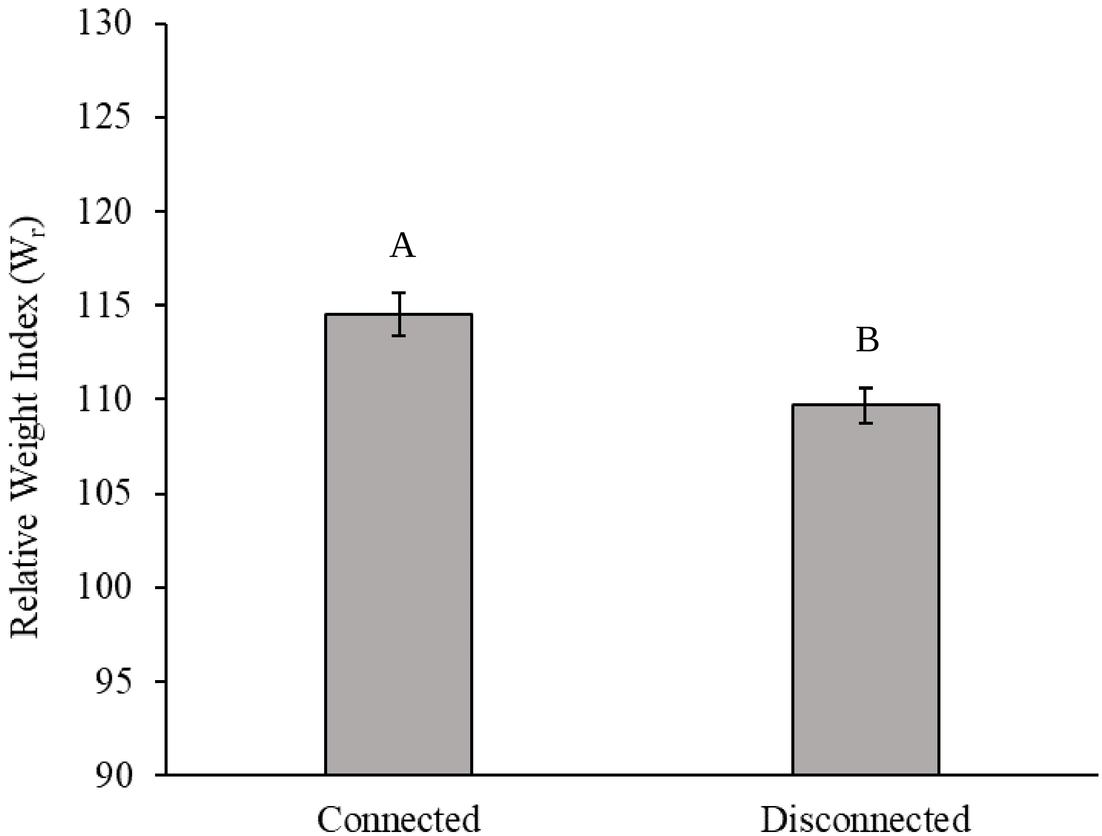

Figure A2-4. Mean W_r of Black Crappie at within both disconnected and connected coves of Harlan County Reservoir across ages 1, 2, and 3. Error bars denote one standard error. Letters above the bars denote significant differences based on a Tukey test ($\alpha = 0.05$).

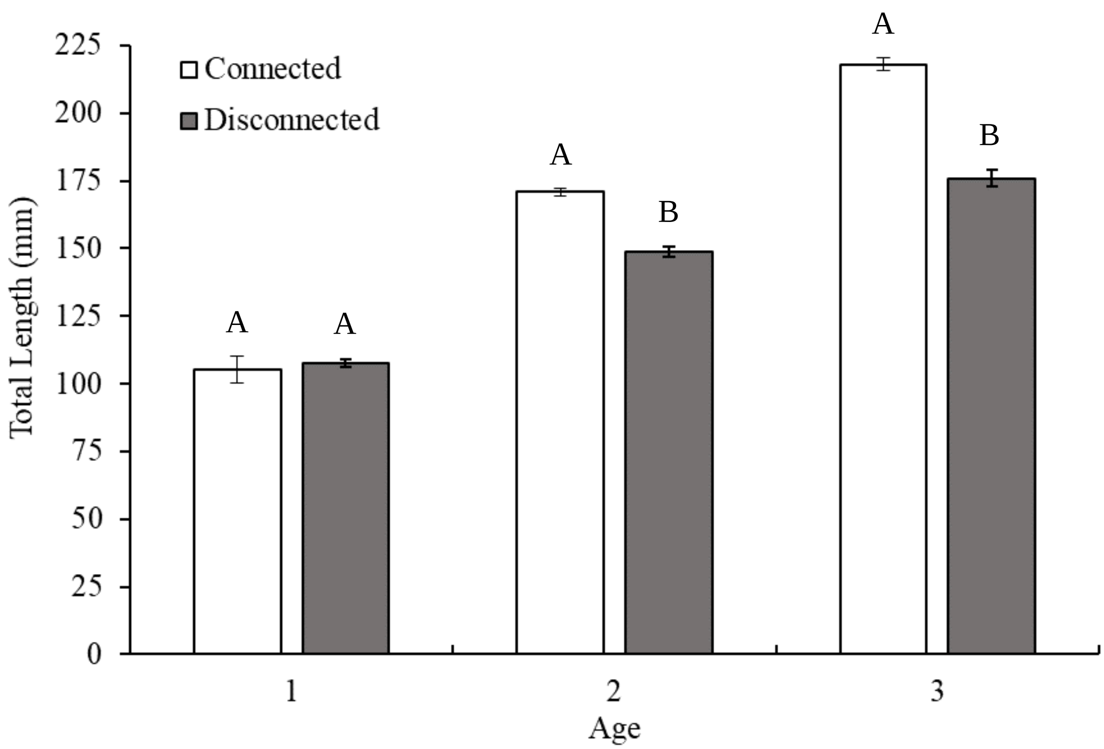

Figure A2-5. Mean total length of Black Crappie at ages 1, 2, and 3 within both disconnected and connected coves of Harlan County Reservoir. Error bars denote one standard error. Letters above the bars denote significant differences between cove types within age groups based on a Tukey test (α = 0.05).

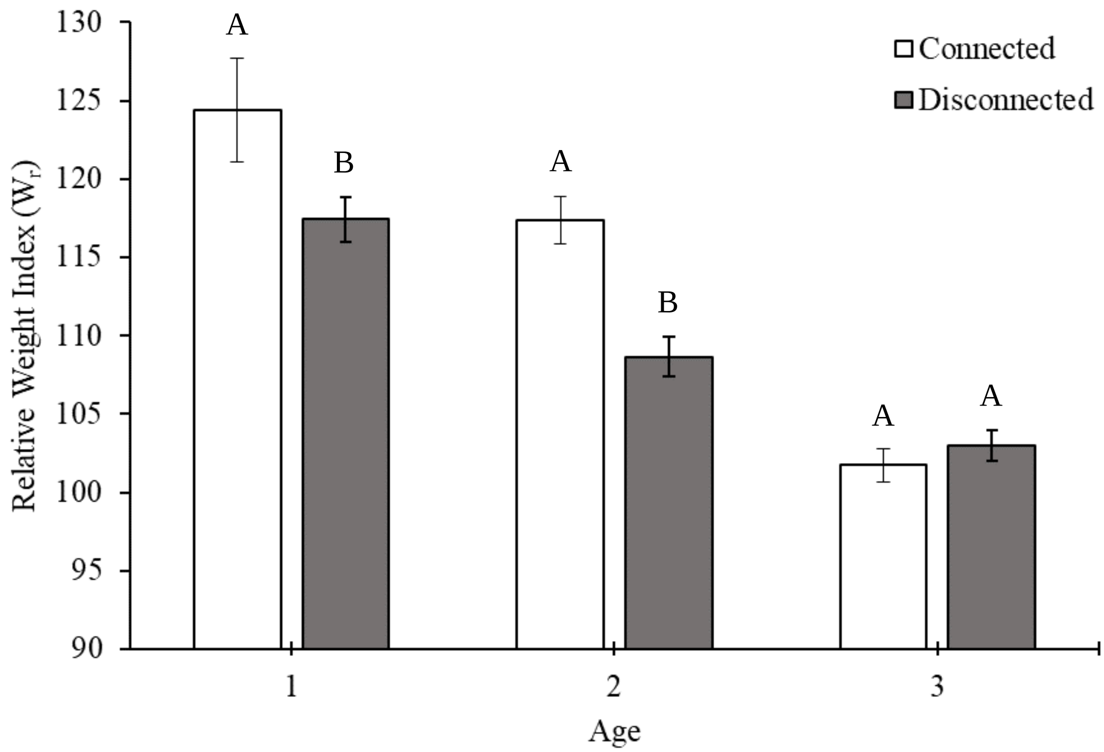

Figure A2-6. Mean W_r of Black Crappie at ages 1, 2, and 3 within both disconnected and connected coves of Harlan County Reservoir. Error bars denote one standard error. Letters above the bars denote significant differences between cove types within age groups based on a Tukey test ($\alpha = 0.05$).