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Response of Large River Fishes to a Prolonged High Water Event in the Missouri River, Nebraska

Nicholas Paul Hogberg

University of Nebraska-Lincoln, nick.hogberg@gmail.com

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RESPONSE OF LARGE RIVER FISHES TO A PROLONGED HIGH
WATER EVENT IN THE MISSOURI RIVER, NEBRASKA

By

Nicholas Paul Hogberg

A THESIS

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RESPONSE OF LARGE RIVER FISHES TO A PROLONGED HIGH WATER EVENT IN THE MISSOURI RIVER, NEBRASKA

Nicholas P. Hogberg, M.S.

University of Nebraska, 2014

Adviser: Mark A. Pegg

Regulation and modification of large rivers to accommodate human uses have been a root cause of freshwater biodiversity declines. The Missouri River is among the most drastically-altered large river systems in North America, with a series of mainstem impoundments in the upper watershed altering flow characteristics downstream, and channelization throughout the lower river homogenizing instream habitat and reducing off-channel habitat. Precipitation events during the winter and spring 2010-2011 caused flooding of the greatest magnitude and duration since reservoir completion. The large magnitude and long duration of this flood made it unlike any flood in recent history and provided a unique opportunity to investigate fish response to floodplain connectivity in a regulated river system. Therefore, the objectives of my research were to 1.) compare fish community characteristics across five floodplain sites along the Missouri River, Nebraska, 2.) compare mean stomach fullness, frequency of empty stomachs, condition, size-at-shift to piscivory, and specific diet items consumed by flathead catfish during the flood year in 2011 and non-flood year in 2012, and 3.) use a long-term data set to relate age-0 channel catfish growth rates to environmental conditions in the channelized Missouri River bordering Nebraska. Differences in community structure and composition existed between the uppermost and lowermost sites, and between a middle site and the remaining four sites; however, associations between the fish community and habitat

attributes were weak. Flathead catfish had higher mean stomach fullness and condition, lower occurrence of empty stomachs, and began consuming fish and crayfish at about a 150 mm smaller size during the flood year in 2011. Specific diet items differed between years only for smaller flathead catfish with invertebrate-dominated diets. Among the five environmental variables used in the modeling process, growing season duration and low discharge duration were most important in predicting juvenile catfish growth rate. This research provides insight to Missouri River fish community structure, trophic response to hydrological events, and aspects of hydrology that affect first-year fish growth, and should add to the ecological components of future water management in the Missouri River.

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CHAPTER 1

GENERAL INTRODUCTION AND STUDY OBJECTIVES

Regulation and modification of large rivers (i.e., rivers with drainage basin > 250,000 km²; Pracheil et al. 2013) to accommodate human uses has been the root of freshwater biodiversity declines (Allan and Flecker 1993; Dudgeon et al. 2006; Vorosmarty et al. 2010). Anthropogenic alterations to aquatic habitat have significantly influenced fish communities at several scales (Ward and Stanford 1983; White et al. 2009; Pierce et al. 2013) and future human uses will continue to press on these already stressed ecosystems (Martinuzzi et al. 2013) with potentially dire consequences for already imperiled biota (Pracheil et al. 2013).

Regular floodplain inundation is a natural phenomenon in unaltered lotic systems, but regulation of these events is among the human alterations to large river ecosystems (Nilsson et al. 2005). In the USA alone, approximately 2.5 million dams and other water control structures are in place, and only about two percent of rivers are unmodified (Lytle and Poff 2004). Regulated flow regimes are characterized by homogeneous discharge through much of the year, resulting in reduced spring flows and increased autumn flows (Lytle and Poff 2004). Long term seasonal and annual flow homogenization reduces regional differences in environmental conditions within a river system, and may cause native biota, having evolved to survive in the naturally dynamic flow conditions, to have reduced fitness due to changes in environment (Poff et al. 2007) and connectivity (Bunn and Arthington 2002).

The Missouri River is among the most drastically-altered large river systems in North America. A series of six mainstem impoundments was constructed in the upper and

middle Missouri River between 1937 and 1963 to secure water for navigation, irrigation, municipal use, hydropower, and flood control (U.S. Army Corps of Engineers 2006). Operations of these dams have severely altered flows throughout the Missouri River, and particularly affected spring flows in portions of the middle and lower river (Pegg et al. 2003). The post-impoundment Missouri River flow regime has replaced seasonal flow pulses with relatively stable discharge across seasons (Hesse and Mestl 1993; Pegg et al. 2003). Among the changes caused by flow regulation in the Missouri River has been the disconnection between the river and its floodplain. Population declines have been documented for several native fish species in the Missouri River since the onset of flow regulation (e.g., Hesse et al. 1993; Galat et al. 2005), and river-floodplain disconnection was identified as a contributing factor (Hesse et al. 1993).

Record quantities of water from snowmelt and precipitation events in the upper Missouri River watershed during spring 2011 added an estimated $7.5 \times 10^{10} \text{ m}^3$ of water to the basin above Sioux City, Iowa; $1.5 \times 10^{10} \text{ m}^3$ greater than the previous record set in 1997 and more than $3.3 \times 10^{10} \text{ m}^3$ greater than the 112 year median (U.S. Army Corps of Engineers, unpublished data). Portions of Wyoming and Montana experienced the second and third wettest May on record, and precipitation runoff during May 2011 ranged from 100% to over 500% of the average May runoff in the middle and upper Missouri River (National Oceanic and Atmospheric Administration, unpublished data).

Storage capacity in the mainstem reservoirs was quickly exhausted, and releases were increased to record levels by mid-summer. By June 18, 2011, the Missouri River was 1.2 m above flood stage at Sioux City, Iowa, and remained above flood stage until August 25, 2011. The discharge at Sioux City was nearly 5,400 cubic meters per second

($\text{m}^3 \cdot \text{s}^{-1}$); $4,530 \text{ m}^3 \cdot \text{s}^{-1}$ greater than the 59 year median June 18 discharge. Similarly, the Missouri River at Nebraska City, Nebraska was at or above flood stage from mid-April through late-September, 2011, with a maximum gage height of over three meters above flood stage reached on June 26 (United States Geological Survey, unpublished data). The maximum discharge at Nebraska City was recorded on July 7 at $6,484 \text{ m}^3 \cdot \text{s}^{-1}$ and stayed above $4,200 \text{ m}^3 \cdot \text{s}^{-1}$ through the end of August.

Record-setting precipitation and subsequent water releases from the Missouri River reservoirs resulted in substantial and prolonged river-floodplain connectivity for the first time in the post-impoundment era. Therefore, the 2011 Missouri River flood presented a rare opportunity for fisheries scientists to evaluate interactions between fish and the inundated floodplain. Particularly, my objectives included assessing longitudinal differences in fish community structure and composition at five locations along flooded the Missouri River, and to determine the features of floodplain habitat that may be useful in explaining habitat use (Chapter 2). Second, I quantitatively compared flathead catfish diet between the floodplain-connected year during 2011 and floodplain-not-connected year in 2012 to determine whether floodplain access influences the trophic dynamics of a native Missouri River predator (Chapter 3). Finally, I assessed first year growth in juvenile channel catfish in the Missouri River to investigate relations between environmental conditions and first-year growth (Chapter 4).

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CHAPTER 2

FISH COMMUNITY COMPARISONS AND FLOODPLAIN HABITAT USE AT FIVE SITES IN THE FLOODED MISSOURI RIVER, NEBRASKA

ABSTRACT

Anthropogenic alterations to large rivers have caused widespread changes in river function, and are likely responsible for freshwater biodiversity declines. Human modifications to rivers in the United States alone have resulted in over two million water control structures, and subsequently altered the flow dynamics of many large North American rivers. The Missouri River and its fish community have been strongly influenced by decades of homogenized flow and channelization, and these changes have contributed to declines in several native species. We collected fish and habitat data at five locations on the Missouri River, Nebraska floodplain during the largest flood since the construction of the mainstem reservoirs. We found differences in fish community structure (ANOSIM $R = 0.255$, $P = 0.001$) and composition ($R = 0.253$, $P = 0.001$) between the two most longitudinally-separated sites, Ponca State Park and Indian Cave State Park, as well as between a middle site at Tieville Bend and all other sites ($R = 0.270$ - 0.480 , $P = 0.001$). Differences in the fish community between Tieville Bend and the other four sites were largely caused by the high abundance of black bullhead *Ameiurus melas* at Tieville Bend. Attempts to relate species abundance to habitat measurements resulted in weak correlations, and suggest that factors influencing floodplain habitat use are complex, and possibly driven by habitat components not measured in this study. Future work is needed to investigate relations between the fish community and its use of floodplain resources.

INTRODUCTION

Native aquatic organisms in large rivers have evolved their life history, as well as behavioral and morphological characteristics, to subsist during extreme flow conditions they experience as part of the natural flow regime (Poff et al. 1997; Lytle and Poff 2004). Lateral connectivity between the main channel and floodplain has been identified as a critical component of large river function by increasing primary and secondary production in riverine ecosystems and providing beneficial off-channel habitat for aquatic organisms (Junk et al. 1989). Floodplain habitat use is believed to be important for several aspects of large river fish ecology, including foraging, spawning, and offspring rearing (Poff et al. 1997). Alterations to large river natural flow regimes and aquatic habitat, however, have been implemented to limit flooding and manipulate the river for human uses. Modifications to flow and habitat have been identified as a contributing factor to worldwide declines in freshwater biodiversity (Allan and Flecker 1993; Strayer and Dudgeon 2010; Vorosmarty et al. 2010).

Many large rivers in North America were altered and regulated beginning in the 19th and 20th centuries to accommodate human needs (Bravard and Petts 1996). In the United States, nearly 2.5 million water control structures are in place on rivers (Lytle and Poff 2004), and only 42 rivers > 200 km long are not influenced by dams (Poff et al. 2007). Regulated flow regimes are characterized by homogeneous discharge through much of the year, resulting in reduced spring flows, increased autumn flows, and reduced flooding (Lytle and Poff 2004). Long-term flow homogenization is likely to influence fish community structure and diversity, and may have deleterious effects on species relying on elements of natural flow (Bunn and Arthington 2002; Lytle and Poff 2004).

Channelization projects on the Missouri River shortened the river by over 200 km (Hesse et al. 1993) and reduced the wetted area of the river and floodplain wetlands downstream of Sioux City, Iowa by 50%. Additionally, six impoundments in the upper watershed permanently inundated over 500,000 ha of floodplain habitat (Whitley and Campbell 1974). Flow regulation by the six mainstem impoundments has resulted in a significant reduction in flow variability and prevented or reduced the magnitude of floods in the middle Missouri River since the 1950s (Hesse and Mestl 1993a; Pegg et al. 2003). Population declines have been documented for several native large river species in the Missouri River (Hesse 1993; Hesse and Mestl 1993b; Galat et al. 2005), and habitat modification has been identified as a contributing factor.

During winter and spring 2010-2011, record-setting snowpack and rainfall caused the mainstem Missouri River to connect with its floodplain for the greatest duration and greatest discharge magnitude since the completion of Gavins Point Dam (RKM 1305) in 1957. Few studies to date have assessed fish community use of floodplain habitats along the Missouri River, and no such study has been done on the Missouri River during a flood of this magnitude. The objectives of this study were to determine if floodplain sites along the Missouri River, Nebraska were different in terms of fish community composition and structure. We also evaluated floodplain habitat use by fish for associations of fish to specific habitat attributes.

METHODS

Study area

Floodplain sampling was conducted at five locations along the Missouri River, Nebraska (Figure 2-1). Sampling sites and river kilometer (RKM) locations were Ponca State Park (Ponca SP; RKM 1212), Tieville Bend (RKM 1116), Boyer Chute National Wildlife Refuge (Boyer Chute NWR; RKM 1025), Schilling Wildlife Management Area (Schilling WMA; RKM 952), and Indian Cave State Park (Indian Cave SP; RKM 834). Each site was sampled for two consecutive days every two weeks between late June and mid-August.

Data collection

We collected fish by daytime boat electrofishing with a Smith-Root 5.0 GPP control box at 15 Hz, 5 amp and 60 Hz, 8 amp settings. Habitats were visually assessed prior to each electrofishing run and placed within one of six habitat classifications (Table 2-1). We attempted to expend equal sampling effort of both electrofishing settings in each available habitat type at each site. Electrofishing runs were five minutes, or until the habitat patch had been sampled in its entirety. Water depth (m) was recorded at the beginning, midpoint, and end of each electrofishing run, and other metrics of physical and chemical habitat conditions including velocity (m/s), conductivity ($\mu\text{s}/\text{cm}$), and temperature ($^{\circ}\text{C}$) were recorded after each electrofishing run. Species-specific catch-per-unit-effort (CPUE) for each electrofishing run was calculated as number of fish collected per minute.

Data analysis

Fish community structure and composition were compared across sites with a matrix of species-specific catch rates calculated for each electrofishing run. The species pool used for community comparisons was truncated to include species represented by at least 10 individuals, as rare species may not have been susceptible to the sampling gear and have little influence on community analyses (Arscott et al. 2006). Catch data used for community structure analysis were fourth-root transformed to down weight extreme abundance values; whereas community composition analyses used presence/absence. A Bray-Curtis resemblance matrix was used to generate pairwise similarity values for each pair of samples. Similarity values range from 0 to 100, with higher values indicating more similar samples (Bray and Curtis 1957). One-way analysis of similarity (ANOSIM; Clarke 1993) tested for differences in community composition and structure among sites and provided pairwise comparisons of dissimilarity. Results of the ANOSIM were interpreted based on a combination of Global and pairwise R and P statistics. Values of the R statistic typically range from 0 to 1, with values closer to 0 suggesting weak distinction between groups, and values closer to 1 indicating strong differences between groups. The P statistic also ranges from 0 to 1, and is interpreted similarly to other frequentist statistical tests, such that values less than a pre-determined critical alpha (e.g., 0.05) are assumed significant. Clarke and Gorley (2006) report that Global and pairwise R statistics are less sensitive than P statistics to sample size evenness across treatments, and therefore suggest reliance on R statistics to infer significance when sample sizes across treatments are uneven. A similarity test (SIMPER; Clarke 1993) based on was used to describe species-specific contributions to overall dissimilarity in community

structure between sites. Samples where no fish from any species were collected were excluded from ANOSIM and SIMPER tests due to the inability to construct resemblance matrices for samples with no fish. Therefore, abundances reported in the results of the SIMPER analysis are reported as “adjusted abundances,” and are higher than the true abundances when all zero catches are incorporated. Two dimensional non-metric multidimensional scaling ordinations of catch data were used to visually inspect separation between sites. Bubble plots were used to depict differences in species abundance between sites, where larger bubbles indicate higher abundances and the locations of bubbles in the plot indicate the sites where species were collected. Ordination plots include a stress value to represent how accurately the data are represented in the given number of dimensions. Lower stress (generally < 0.05) is desirable, and stress far exceeding 0.1 should be mitigated by adding additional dimensions (Manly 2005).

Distinctness of visual habitat assignments, determined prior to sampling, was assessed using a linear discriminant function analysis (DFA; SAS 2009) to determine whether habitat designations assigned in the field could be differentiated based on environmental and biological data collected within each habitat type. A DFA predicts the group (in this case habitat designation) in which a given observation belongs based on a suite of continuous predictor variables (Manly 2005). The DFA of environmental data used a matrix of depth, velocity, conductivity, and temperature measurements at each electrofishing location. We established a species pool of 25 species that were each represented by at least 10 individuals across sites, and used a matrix of species-specific catch rates for each electrofishing run for the DFA based on biological data. The result of this analysis was a matrix of percentages indicating the proportion of observations within

each habitat type that would be placed into the correct category and into each incorrect category based solely on the combination of predictor variable measurements taken from an observation with no knowledge of the categorical assignment.

We also used a Mantel correlation (Primer-E version 6 “BEST” procedure; Clarke 1993; Clarke and Gorley 2006) to relate habitat variables to catch data by fourth-root transforming catch data and generating a Bray-Curtis resemblance matrix.

Environmental variables were normalized (mean = 0, standard deviation = 1) to remove the effect of differing scale across metrics.

RESULTS

Floodplain sampling consisted of 400 electrofishing runs, and ranged from 57 electrofishing runs at Indian Cave State Park to 92 electrofishing runs at Tieville Bend (Table 2-2). Across sites, most (average 35%) electrofishing runs were conducted in bankline habitats; whereas the smallest percentage of samples was collected in Open Water – Ag habitats (average 4%; Table 2-2). A total of 1,831 fish representing 45 species was collected across sites, and 25 species were represented by at least 10 individuals (Appendix 1).

Differences in fish community structure between sites (Global $R = 0.263$, $P = 0.001$; Figure 2-2) were most notable between the upper most site (Ponca SP) and lower most site (Indian Cave SP; $R = 0.255$, $P = 0.001$; Table 2-3), and between Tieville Bend and all other sites ($R = 0.270-0.480$, $P = 0.001$; Table 2-3). Total dissimilarity in fish community structure between Ponca SP and Indian Cave SP was most influenced by the adjusted abundance of flathead catfish *Pylodictus olivaris* (18%), common carp *Cyprinus*

carpio (11%), and river carpsucker *Carpionodes carpio* (6%; Table 2-4; Figure 2-3). Black bullhead *Ameiurus melas* adjusted abundances contributed the greatest percent contribution to dissimilarity comparisons between Tieville Bend and the other four sites, representing between 22% and 26% of the difference in fish community structure (Table 2-5; Figure 2-3). Other species contributing to dissimilarity between Tieville Bend and the other four sites were common carp (8%-13%) and bigmouth buffalo *Ictiobus cyprinellus* (9%-12%; Table 2-5; Figure 2-3). Flathead catfish contributed to differences in fish community structure between Tieville Bend and Boyer Chute National Wildlife Refuge (Boyer Chute NWR; 6%), Schilling Wildlife Management Area (Schilling WMA; 7%), and Indian Cave SP (17%; Table 2-5). Gizzard shad *Dorosoma cepedianum* contributed to differences in fish community structure between Tieville Bend and Boyer Chute NWR (7%), Schilling WMA (6%), and Indian Cave SP (6%; Table 2-5).

Differences in community composition across sites (Global $R = 0.259$, $P = 0.001$; Figure 2-4) were also most notable between Ponca SP and Indian Cave SP ($R = 0.253$, $P = 0.001$; Table 2-6), and between Tieville Bend and all other sites ($R = 0.262$ - 0.469 , $P = 0.001$; Table 2-6). Most of the dissimilarity between Ponca SP and Indian Cave SP was contributed by the occurrence of flathead catfish (67% of samples at Indian Cave SP; 9% of samples at Ponca SP) and common carp (44% of samples at Indian Cave SP; 9% of samples at Ponca SP). Black bullhead occurrence contributed the most dissimilarity in community composition between Tieville Bend (77% of samples) and the remaining sites (0%-17% of samples).

Notable differences in relative abundance were evident in native benthivores such as shovelnose sturgeon *Scaphirhynchus platyrhynchus*, river carpsucker, shorthead

redhorse *Moxostoma macrolepidotum*, and blue sucker *Cycleptus elongatus*, all of which were most abundant in samples at Ponca SP and generally declined in abundance or were not collected at downstream sites (Table 2-7). Flathead catfish catch rates were highest at Indian Cave SP, continuously declined at upstream sites, and were lowest at Ponca SP (Table 2-7). Catch rates for several species including longnose gar *Lepisosteus osseus*, shortnose gar *L. platostomus*, silver carp *Hypophthalmichthys molitrix*, smallmouth buffalo *Ictiobus bubalus*, blue catfish *Ictalurus furcatus*, and channel catfish *I. punctatus* were highest at Schilling WMA near the Platte River/Missouri River confluence (Table 2-7).

Our ability to distinguish among most habitat types in the DFA using depth, velocity, conductivity, and temperature measurements was poor (Table 2-8). “Open Water – Ag” and “Permanent Backwater” habitats were correctly classified in 89% and 63% of observations across sites. Other habitats were less distinct, including “Open Water – Grass” (11% correct placement) and “Vegetation – Trees” (17% correct placement; Table 2-8). Temperature readings were similar across habitats, as less than two degrees Celsius separated the warmest and coolest habitat types (Figure 2-5). More noticeable differences existed in mean depth and velocity measurements across habitat types. “Permanent Backwater” habitats had the greatest mean depth (2.7 m), and “Bankline” and “Open Water – Ag” habitats were the shallowest, averaging 1.7 m deep (Figure 2-5). Mean water velocity was fastest in “Open Water – Ag” habitats, averaging 0.56 m/s, and slowest in “Permanent Backwater” habitats, at 0.19 m/s (Figure 2-5).

Correct placement of electrofishing runs in *a priori* determined habitats using the DFA based on fish catch data was better than the habitat measurement-based DFA for

most habitats (Table 2-9). Correct placement percentages ranged from 28% in “Vegetation – Trees” to 56% in “Open Water – Ag” habitats (Table 2-9). Fish catch data were also poorly correlated to habitat measurements (Table 2-10). The best correlation from the Mantel test related temperature, depth, and velocity to catch data, but still only resulted in a correlation of 0.137. Velocity was incorporated in eight of the top ten variable combinations, suggesting it was one of the most important variables to predict species occurrence.

DISCUSSION

Large river fish communities are shaped by a number of complex biotic and abiotic factors. The altered hydrograph that currently characterizes the Missouri River flow regime has specifically decreased seasonal flow variability, and the resulting decades of separation between the main channel and floodplain habitats have likely influenced the current fish community. Differences in the fish community between Ponca SP and Indian Cave SP are likely caused by differing levels of habitat modification and influence of flow regulation. For example, the river at Indian Cave SP is within the channelized portion of the Missouri River, where off-channel and shallow water habitats have been greatly reduced since the mid-1900s (Whitley and Campbell 1974). Conversely, the river at Ponca SP is unchannelized and maintains several properties of its natural physical habitat, including reconstructed backwaters that provide additional habitat complexity. Ponca SP is, however, more directly influenced by modified water releases from Gavins Point Dam, whereas flow patterns at lower sites such as Indian Cave SP are buffered by inputs from tributaries including the Platte River (Figure 2-6).

Differences in the occurrence of native benthivores between Ponca SP and the other four sites suggest that these native species may be more limited by habitat diversity than flow regulation. Some of these native benthivores, including blue suckers and shovelnose sturgeon, are currently considered species of conservation concern in Nebraska and other states within their range (Keenlyne 1997; Neely et al. 2009). Physical habitat has been identified as an abiotic factor that shapes community composition (Bunn and Arthington 2002), so it is critical that future efforts to recover native fishes in the Missouri River continue to assess habitat associations of these species.

Differences in both community composition and structure at the Tieville Bend site were mostly caused by the abundance of black bullheads. A 17 hectare, three meter maximum depth, floodplain lake was built on the floodplain at Tieville in 2003 (Sterner et al. 2009). Sterner et al. (2009) surveyed the fish community of the lake at Tieville Bend from 2006 through 2008 and found that black bullhead was the dominant species collected. Environmental conditions in the lake resulting from its shallow depth probably allowed only species that are tolerant of extremes in temperature and dissolved oxygen to subsist. Engineering off-channel habitats to allow flow between the lake and main river channel will help buffer extreme environmental conditions and allow fish to move between the river and lake as needed. Connectivity between floodplain lakes and the main channel across a range of within-channel discharges will be especially important if floodplain connectivity in the Missouri River above the Platte River confluence continues to occur as infrequently as it has in the past 60 years. Additionally, studies to assess fish movement through the water control structure at the Tieville Bend lake would help

determine the required flow to allow fish movement between the river and lake as environmental conditions warrant.

Restoring floodplain connectivity in targeted locations could provide several benefits to Missouri River fish and wildlife, and may reduce the risk of flood damage to remaining human interests on the floodplain. Agricultural and municipal land uses have drastically changed the appearance of the Missouri River floodplain over the past 60 years, in part due to an extensive levee system that protected human infrastructure from damaging floods. Recent efforts by the United States Army Corps of Engineers and several cooperating agencies to move levees laterally away from the river have shown potential to reduce flood risk to human establishments and reduce levee maintenance costs, while allowing the river to connect with portions of the floodplain during high water. For example, one project near Frazers Bend (RKM 902.8 – 894.8) moved about 3.2 km of levee 1.1 km away from its original location, allowing over 390 ha of floodplain to reconnect to the river during high water. This project will facilitate lowering the stage of a 100-year flood by nearly 0.5 m (US Army Corps of Engineers, unpublished data). Another project in Copeland Bend (RKM 909.2 – 906.1) and Nebraska Bend (RKM 906.1 – 900.7) moved 7.2 km of levee 0.8 km away from its original location, allowing 310 ha of floodplain to be reconnected with the river during high water. The Copeland/Nebraska Bends project will also facilitate a 0.5 m reduction in 100-year flood stage (US Army Corps of Engineers, unpublished data). Similar techniques to allow river-floodplain connectivity within a defined corridor have been used along several rivers in Europe, Asia, and the USA (e.g., Rapp and Abbe 2003; Piegay et al. 2005). The erodible corridor concept (ECC) has been implemented to allow more natural stream

function, in the form of less-inhibited sediment dynamics and more regular flooding, while providing increased flood protection to remaining human interests on the floodplain. Justification for the ECC includes the economic unsustainability of engineered bank protection and the recognition of ecosystem services provided by erosive processes that had not been realized during initial river modification (Piegay et al. 2005). Continued efforts to move levees away from the river and re-establish aquatic and terrestrial habitats inside the levees will likely aid imperiled fauna through increased habitat heterogeneity that more closely resemble pre-alteration habitats along the Missouri River.

Our analyses targeted at evaluating visual and empirical habitat data to delineate use by fish underline the importance of identifying relevant habitat attributes. Specifically, the DFA based on fish catch data had better overall precision and accuracy than classifications using measured habitat variables. Our findings suggest the habitat classifications were biologically meaningful because we could delineate species composition among habitats where fish were collected. However, our empirical measures were not able to explain differences in habitat use by the fish community on their own. Li and Gelwick (2005) related habitat attributes to community structure in the Brazos River, Texas and found that shallow water fish community structure was most related to depth, velocity, and substrate. Similarly, Copp et al. (1994) surveyed floodplain habitat use by juvenile fish in the Danube River, and identified water velocity and turbidity among the most influential environmental variables to predict species occurrence. Collecting additional data to describe the presence and prevalence of physical habitat features (e.g.,

woody debris, substrate type, etc.) may have increased our ability to specifically link species or guild-specific abundance to empirical habitat measures.

An alternative explanation for our inability to delineate habitat use could be the combined result of drastic and widespread floodplain habitat alteration during the 1800s and 1900s and the infrequency of floods in the past 50 years. Bragg and Tatschl (1977) evaluated changes in vegetation and land use on the Missouri River floodplain in Missouri between 1826 and 1972, and noted that floodplain forest area was reduced by 63%, while agricultural area increased by 65%. This homogenization of physical habitat could have changed the way Missouri River fishes used the floodplain during the 2011 flood, and may have made patterns in habitat use less distinct than they would have been historically. Galat et al. (2005) determined that 60 of the 107 native Missouri River fish species (56%) exhibit significant floodplain habitat use and 23 native fish species (21%) are experiencing concerning declines. Many declining species are either known to rely on elements of the natural flow regime to complete their life cycle or have life cycles that we do not fully understand (e.g., Hesse et al. 1993; Jennings and Zigler 2000). Over 50 years of regulated flows and infrequent or non-existent floodplain access could be causing the fish community to transition toward being largely comprised of species that are not reliant on natural flow variability or floodplain access, and would exhibit relatively uniform floodplain habitat use (Bain et al. 2000; Bunn and Arthington 2002; Barko et al. 2004).

The exact nature of fish community structure and function responses to floods that connect a regulated river to its floodplain are unknown. However, we found that there were some differential uses of the floodplain along the Missouri River during a flood that

connected the river to its floodplain for the first time in over 50 years in some locations. Future efforts to determine the response by fishes in systems where unusually large precipitation events must occur to gain floodplain access are needed.

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Table 2-1. Habitat classifications and descriptions used during floodplain electrofishing on the Missouri River, Nebraska during 2011

Habitat	Description
Bankline	The lateral extent of the inundated floodplain, or the edge of floodplain islands
Open Water - Ag	Areas that had been used for row crop agriculture and had no emergent physical habitat
Open Water - Grass	Areas with no emergent physical habitat and had not been used for row crop agriculture
Vegetation - Shrubs	Habitats dominated by shrubs, with most branches below the water line
Vegetation - Trees	Habitats dominated by trees, with most branches above the water line
Permanent Backwater	Floodplain lakes and off-channel water bodies that are connected to the main channel during normal flows

Table 2-2. Number and percentage of electrofishing samples taken in each of six habitat classifications at five sites along the Missouri River, Nebraska during the 2011 flood.

	Ponca SP Samples (%)	Tieville Bend Samples (%)	Boyer Chute NWR Samples (%)	Schilling WMA Samples (%)	Indian Cave SP Samples (%)
Bankline	23 (34)	27 (29)	30 (35)	38 (39)	23 (40)
Open Water - Ag	4 (6)	0 (0)	8 (9)	0 (0)	3 (5)
Open Water - Grass	19 (28)	34 (37)	22 (26)	24 (25)	15 (26)
Vegetation - Trees	17 (25)	17 (18)	24 (28)	21 (22)	13 (23)
Vegetation - Shrubs	5 (7)	14 (15)	2 (2)	2 (2)	3 (5)
Permanent Backwater	N/A*	N/A	N/A	12 (12)	N/A

* - Permanent backwaters are present at Ponca State Park, but they were too deep to effectively sample during the flood.

Table 2-3. Pairwise analysis of similarity comparisons of fish community structure at floodplain electrofishing sites on the Missouri River, Nebraska during 2011. Comparisons marked with an asterisk were considered significant using criteria from both the Global R and P statistics.

Site	Site	<i>R</i>	<i>P</i>
Boyer Chute NWR	Schilling WMA	0.084	0.001
Boyer Chute NWR	Tieville Bend	0.270	0.001*
Boyer Chute NWR	Ponca State Park	0.083	0.001
Boyer Chute NWR	Indian Cave State Park	0.146	0.001
Schilling WMA	Tieville Bend	0.381	0.001*
Schilling WMA	Ponca State Park	0.165	0.001
Schilling WMA	Indian Cave State Park	0.080	0.001
Tieville Bend	Ponca State Park	0.463	0.001*
Tieville Bend	Indian Cave State Park	0.480	0.001*
Ponca State Park	Indian Cave State Park	0.255	0.001*

Table 2-4. Contribution of species adjusted abundances to community dissimilarities between Indian Cave State Park and Ponca State Park, Nebraska during 2011. Abundance columns represent the mean adjusted fourth-rooted abundance (fish·min⁻¹) of each species collected. “Percent contribution” column is percentage of total dissimilarity in fish community structure between sites contributed by each species.

Species	Ponca Abundance	Indian Cave Abundance	Percent Contribution
Flathead catfish	0.06	0.54	18.17
Common carp	0.06	0.35	10.97
River carpsucker	0.20	0.03	6.07
Channel catfish	0.12	0.09	5.80
Shovelnose sturgeon	0.13	0.02	5.57
Gizzard shad	0.11	0.14	5.57
Goldeye	0.04	0.16	5.20
Shorthead redhorse	0.15	0.00	4.96
Blue catfish	0.00	0.15	4.66
Blue sucker	0.11	0.02	4.42
Smallmouth buffalo	0.08	0.08	4.41

Table 2-5. Contribution of species abundances to community dissimilarities between Tieville Bend and the remaining four floodplain study sites on the Missouri River, Nebraska during 2011. Abundance columns represent the mean fourth-rooted abundance (fish·min⁻¹) of each species collected. “Percent contribution” column is percentage of total dissimilarity in fish community structure between sites contributed by each species.

Comparison	Species	Tieville Abundance	Other Abundance	Percent Contribution
<u>Tieville Bend and Boyer Chute NWR</u>				
	Black bullhead	0.72	0.13	25.73
	Common carp	0.32	0.31	12.70
	Bigmouth buffalo	0.33	0.26	11.94
	Gizzard shad	0.18	0.19	6.75
	Flathead catfish	0.06	0.10	5.89
	Shortnose gar	0.02	0.14	4.84
	Largemouth bass	0.06	0.14	3.90
	Channel catfish	0.01	0.07	3.18
	White perch	0.13	0.02	3.15
<u>Tieville Bend and Schilling WMA</u>				
	Black bullhead	0.72	0.00	22.49
	Common carp	0.32	0.41	11.66
	Bigmouth buffalo	0.33	0.23	9.10
	Channel catfish	0.01	0.25	8.19
	Flathead catfish	0.06	0.20	7.09
	Gizzard shad	0.18	0.19	6.15
	Blue catfish	0.00	0.18	5.06
	White perch	0.13	0.05	3.13
	Red shiner	0.03	0.06	2.82
<u>Tieville Bend and Ponca State Park</u>				
	Black bullhead	0.72	0.01	24.62
	Bigmouth buffalo	0.33	0.06	8.54
	Common carp	0.32	0.06	8.19
	River carpsucker	0.04	0.20	5.68
	Gizzard shad	0.18	0.11	5.16
	Shorthead redhorse	0.03	0.15	5.01
	Shovelnose sturgeon	0.00	0.13	4.86
	Quillback	0.08	0.12	4.61
	Channel catfish	0.01	0.12	4.53
	Blue sucker	0.01	0.11	4.01

Table 2-5 Continued.

Comparison	Species	Tieville Abundance	Other Abundance	Percent Contribution
<u>Tieville Bend and Indian Cave State Park</u>				
	Black bullhead	0.72	0.03	23.96
	Flathead catfish	0.06	0.54	16.93
	Common carp	0.32	0.35	12.23
	Bigmouth buffalo	0.33	0.07	8.57
	Gizzard shad	0.18	0.14	6.14
	Goldeye	0.01	0.16	4.49
	White perch	0.13	0.08	4.38

Table 2-6. Pairwise analysis of similarity comparisons of fish community composition at floodplain electrofishing sites on the Missouri River, Nebraska during 2011. Comparisons marked with an asterisk were considered significant using criteria from both the Global R and P statistics.

Site	Site	<i>R</i>	<i>P</i>
Boyer Chute NWR	Schilling WMA	0.083	0.003
Boyer Chute NWR	Tieville Bend	0.262	0.001*
Boyer Chute NWR	Ponca State Park	0.082	0.001
Boyer Chute NWR	Indian Cave State Park	0.143	0.001
Schilling WMA	Tieville Bend	0.375	0.001*
Schilling WMA	Ponca State Park	0.164	0.001
Schilling WMA	Indian Cave State Park	0.079	0.003
Tieville Bend	Ponca State Park	0.460	0.001*
Tieville Bend	Indian Cave State Park	0.469	0.001*
Ponca State Park	Indian Cave State Park	0.253	0.001*

Table 2-7. Mean and standard error catch rates (fish·min⁻¹) for twenty-four fish species at five sites along the Missouri River, Nebraska during the 2011 flood.

Common name	Ponca SP	Tieville Bend	Boyer Chute NWR	Schilling WMA	Indian Cave SP
	Mean (SE)	Mean (SE)	Mean (SE)	Mean (SE)	Mean (SE)
Shovelnose sturgeon	0.032 (0.012)	0.000 (0.000)	0.004 (0.003)	0.000 (0.000)	0.003 (0.003)
Longnose gar	0.006 (0.004)	0.000 (0.000)	0.009 (0.005)	0.014 (0.008)	0.003 (0.003)
Shortnose gar	0.009 (0.005)	0.004 (0.003)	0.027 (0.011)	0.033 (0.012)	0.010 (0.006)
Goldeye	0.008 (0.005)	0.002 (0.002)	0.002 (0.002)	0.006 (0.003)	0.046 (0.015)
Gizzard shad	0.050 (0.028)	0.100 (0.036)	0.112 (0.061)	0.134 (0.083)	0.161 (0.135)
Red shiner	0.008 (0.005)	0.010 (0.006)	0.004 (0.004)	0.014 (0.006)	0.004 (0.004)
Grass carp	0.003 (0.003)	0.002 (0.002)	0.002 (0.002)	0.008 (0.004)	0.012 (0.007)
Common Carp	0.014 (0.007)	0.226 (0.082)	0.131 (0.040)	0.208 (0.046)	0.141 (0.034)
Silver carp	0.000 (0.000)	0.010 (0.007)	0.007 (0.004)	0.009 (0.004)	0.007 (0.005)
River carpsucker	0.086 (0.032)	0.010 (0.005)	0.009 (0.005)	0.035 (0.015)	0.007 (0.005)
Quillback	0.047 (0.026)	0.025 (0.012)	0.014 (0.007)	0.004 (0.003)	0.003 (0.003)
Blue sucker	0.028 (0.012)	0.002 (0.002)	0.004 (0.003)	0.012 (0.005)	0.003 (0.003)
Smallmouth buffalo	0.006 (0.006)	0.002 (0.002)	0.002 (0.002)	0.024 (0.012)	0.007 (0.005)
Bigmouth buffalo	0.017 (0.009)	0.149 (0.035)	0.156 (0.076)	0.071 (0.017)	0.021 (0.012)
Shorthead redhorse	0.043 (0.015)	0.008 (0.005)	0.000 (0.000)	0.002 (0.002)	0.000 (0.000)
Black bullhead	0.003 (0.003)	0.585 (0.099)	0.030 (0.013)	0.000 (0.000)	0.007 (0.005)
Blue catfish	0.000 (0.000)	0.000 (0.000)	0.008 (0.008)	0.196 (0.084)	0.063 (0.024)
Channel catfish	0.026 (0.009)	0.004 (0.004)	0.013 (0.007)	0.068 (0.016)	0.035 (0.019)
Flathead catfish	0.011 (0.006)	0.020 (0.009)	0.016 (0.007)	0.053 (0.014)	0.255 (0.048)
Green sunfish	0.009 (0.006)	0.009 (0.008)	0.016 (0.009)	0.002 (0.002)	0.000 (0.000)
Bluegill	0.012 (0.007)	0.006 (0.004)	0.004 (0.003)	0.023 (0.010)	0.000 (0.000)
Largemouth bass	0.011 (0.006)	0.017 (0.007)	0.050 (0.021)	0.037 (0.017)	0.003 (0.003)
White crappie	0.000 (0.000)	0.000 (0.000)	0.002 (0.002)	0.029 (0.009)	0.003 (0.003)
Freshwater drum	0.025 (0.011)	0.005 (0.003)	0.007 (0.004)	0.015 (0.010)	0.003 (0.003)

Table 2-8. Linear discriminant function analysis indicating percent correct placement into habitat categories based on measurements taken from floodplain habitats and sample size for each habitat sampled from five sites on the Missouri River, Nebraska during 2011.

Habitat	Percent Correctly Classified	N
Bankline	36	120
Open Water - Grass	11	54
Open Water - Ag	89	9
Vegetation - Shrubs	21	14
Vegetation - Trees	17	52
Permanent Backwater	63	8

Table 2-9. Linear discriminant function analysis indicating percent correct placement of catch data into habitat categories based on species abundance from floodplain habitats in the Missouri River, Nebraska during 2011. Numbers in parentheses represent the number of samples taken in each habitat at each site.

Habitat	Percent Correctly Classified	N
Bankline	50	121
Open Water - Grass	38	55
Open Water - Ag	56	9
Vegetation - Shrubs	29	14
Vegetation - Trees	28	53
Permanent Backwater	50	8

Table 2-10. Results of Mantel correlation test between habitat measurements and fish catch data collected at five locations in the Missouri River, Nebraska during 2011

Correlation	Habitat variables
0.137	Temperature, Depth, Velocity
0.134	Depth, Velocity
0.132	Velocity
0.122	Temperature, Velocity
0.105	Temperature, Conductivity, Depth, Velocity
0.102	Conductivity, Depth, Velocity
0.090	Conductivity, Velocity
0.090	Temperature, Conductivity, Velocity
0.084	Temperature, Depth
0.075	Depth

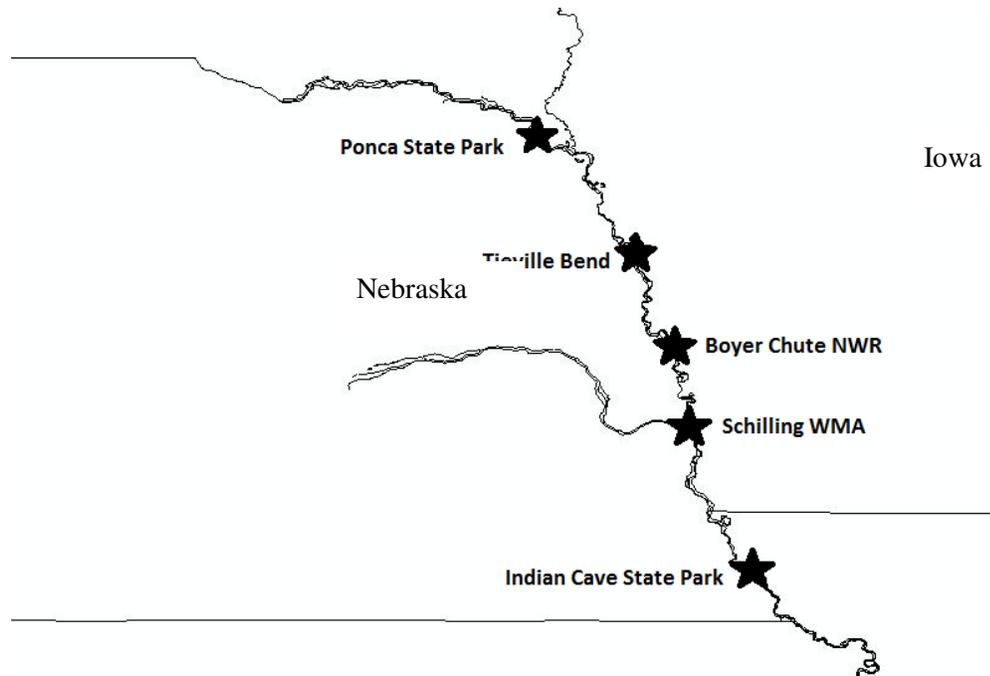


Figure 2-1. Map of Missouri River floodplain study sites for fish community assessment and habitat use during 2011.

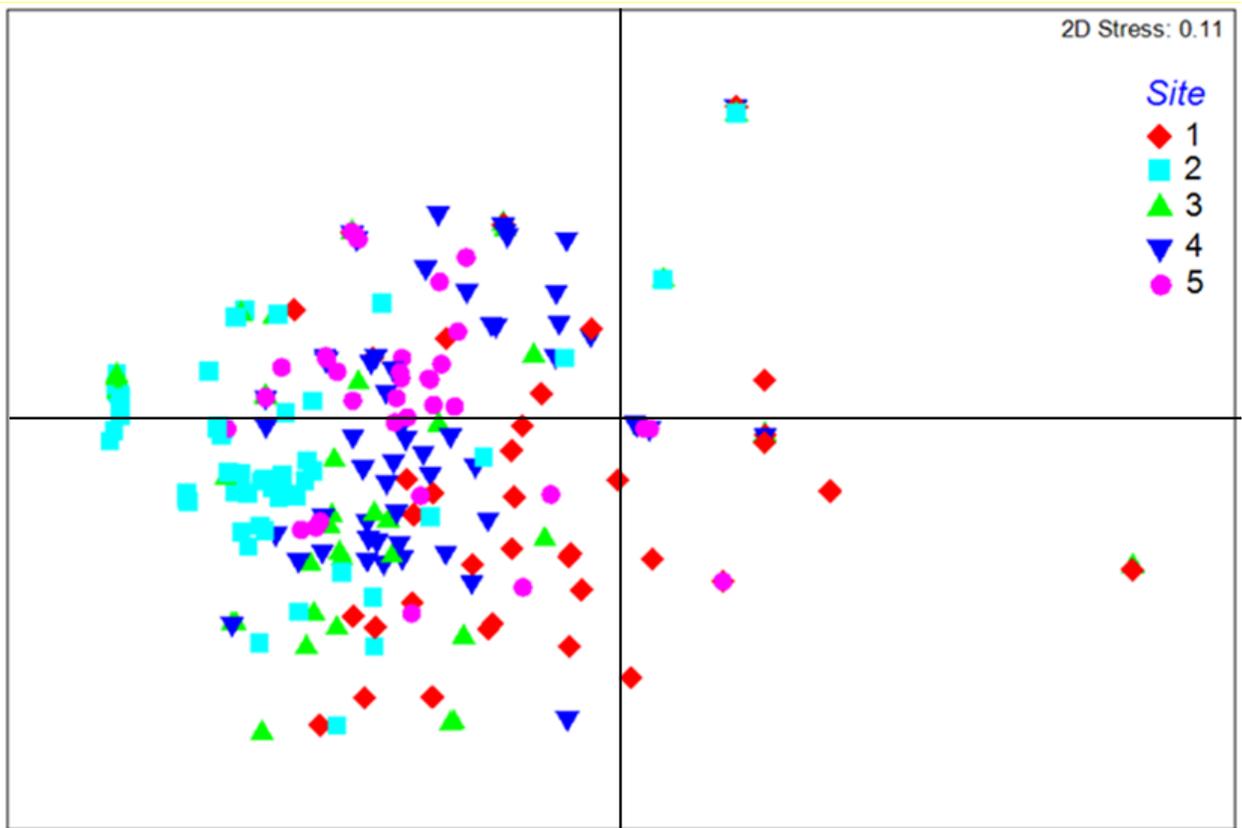


Figure 2-2. Non-metric multidimensional scaling ordination of fish community structure data from five floodplain sites on the Missouri River, Nebraska during 2011. Site 1 = Ponca State Park, Site 2 = Tieville Bend, Site 3 = Boyer Chute NWR, Site 4 = Schilling WMA, Site 5 = Indian Cave State Park. Black lines are for spatial reference.



Figure 2-3. Non-metric multidimensional scaling bubble plots of species with notable contributions to dissimilarity in fish community structure across five floodplain sites in the Missouri River, Nebraska during 2011. Bubble location is consistent with site location in figure 2-2. Bubble size is indicative of relative abundance, with larger bubbles representing higher catch rates. Black lines are for reference with figure 2-2.

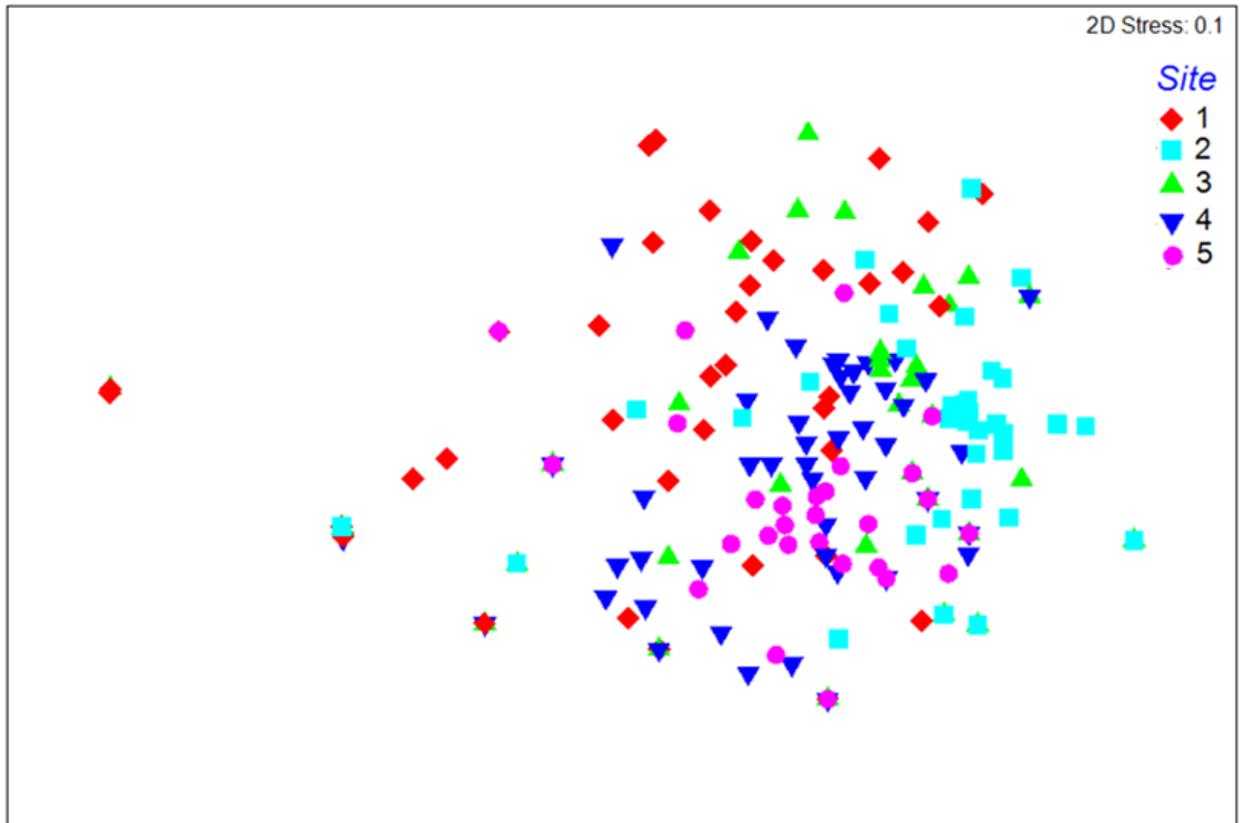


Figure 2-4. Non-metric multidimensional scaling plot of fish community composition data from five floodplain sites on the Missouri River, Nebraska during 2011. Site 1 = Ponca State Park, Site 2 = Tieville Bend, Site 3 = Boyer Chute NWR, Site 4 = Schilling WMA, Site 5 = Indian Cave State Park.

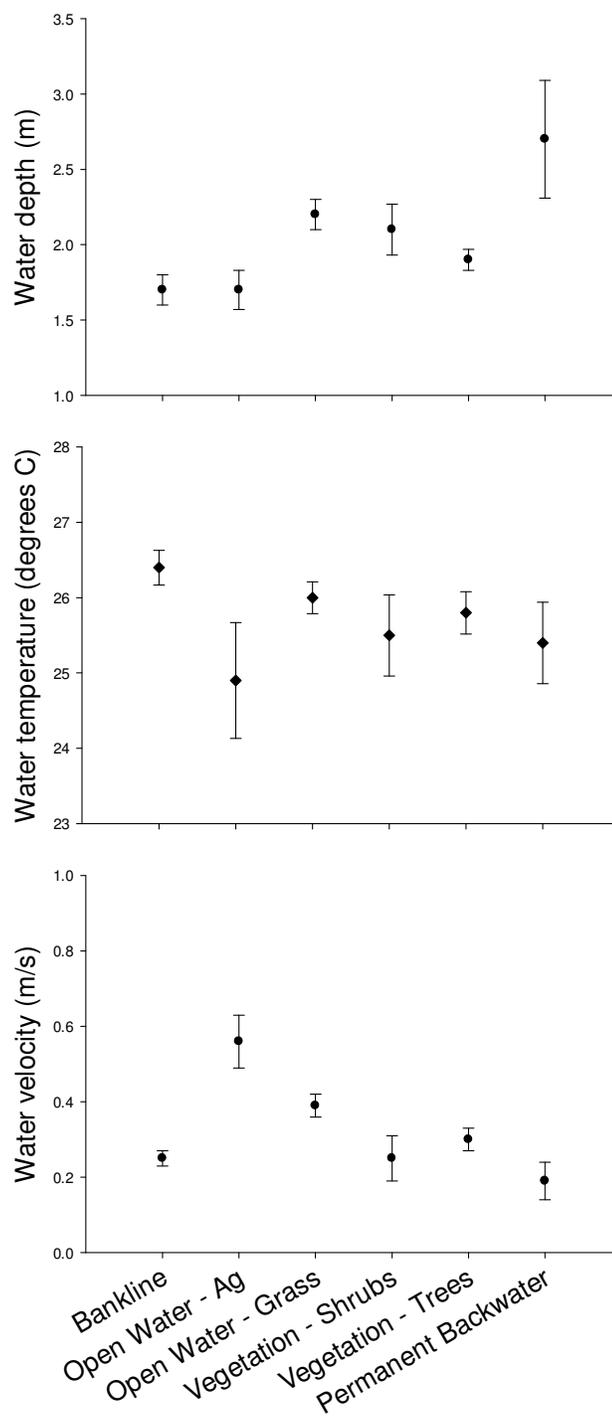


Figure 2-5. Mean \pm SE water depth (top), temperature (middle), and velocity (bottom) across six habitat designations at five floodplain sampling sites on the Missouri River, Nebraska during 2011.

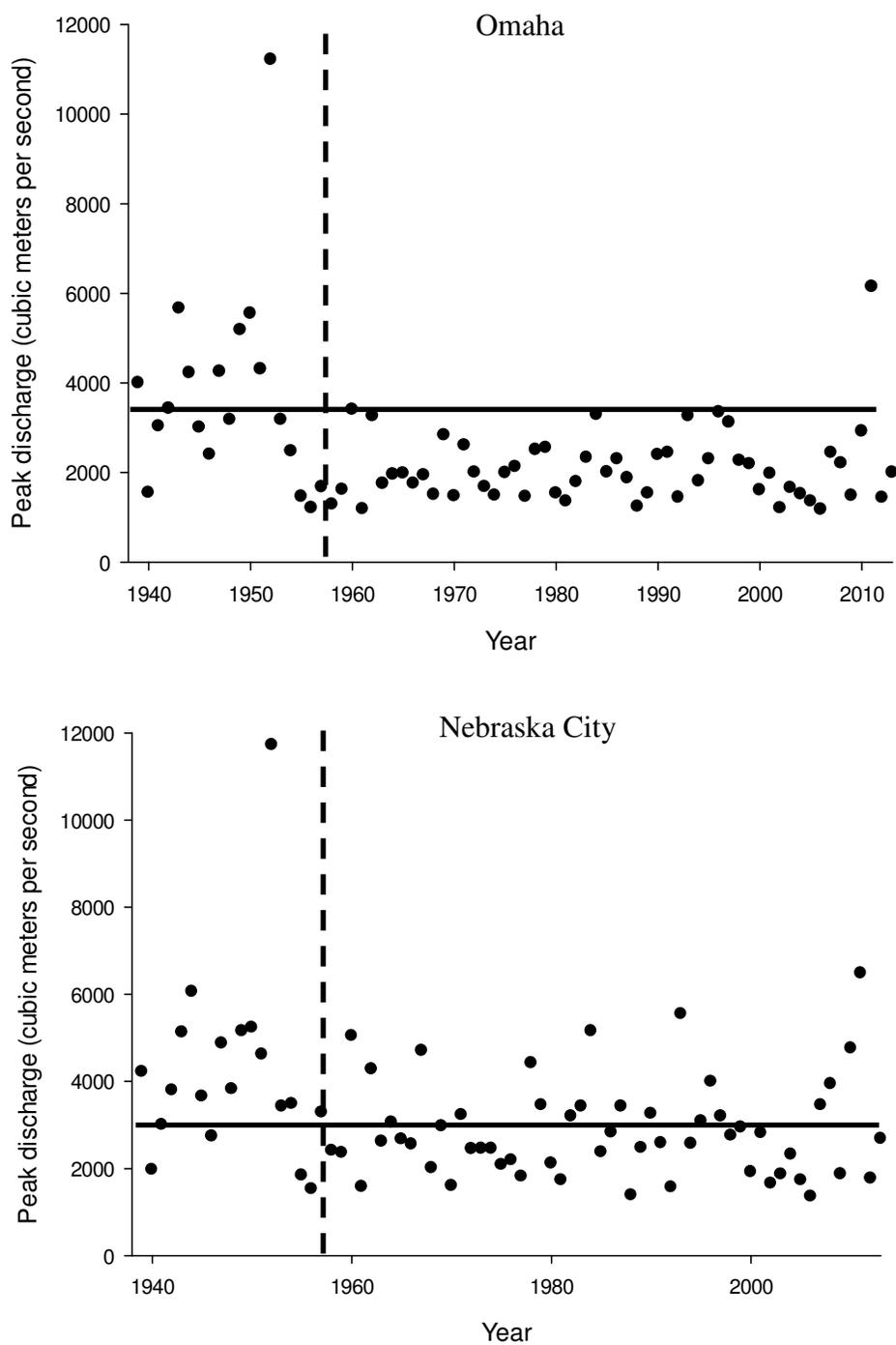


Figure 2-6. Peak annual discharge from 1939-2013 for the Missouri River at Omaha, Nebraska (upstream of the Platte River confluence; top) and Nebraska City, Nebraska (downstream of the Platte River confluence; bottom). The black horizontal line indicates approximate discharge at flood stage at the two locations, and the dashed vertical line represents the beginning of flow regulation by Gavins Point Dam in 1957 (data from United States Geological Survey).

CHAPTER 3

DIETARY RESPONSE OF FLATHEAD CATFISH *PYLODICTIS OLIVARIS* TO FLOODPLAIN CONNECTIVITY IN THE MISSOURI RIVER, NEBRASKA

ABSTRACT

Regular flooding occurs naturally in large rivers and native biotas in these systems have evolved to capitalize on increased production that takes place during floodplain connectivity. Since the mid-20th century, flow regulation and channelization efforts in the lower and middle Missouri River have resulted in almost total confinement of the main channel within incised banks. Record flooding in 2011, however, caused the middle Missouri River to breach levees and inundate its floodplain for much of the summer and fall. Subsequently, near-record drought occurred during 2012, causing the river to remain within its channel all year. We compared flathead catfish *Pylodictis olivaris* diet between 2011 and 2012 to assess differences in mean stomach fullness, frequency of empty stomachs, size at shift to piscivory, and differences in prey taxa consumed during two years with overtly different hydrographs. We collected 629 flathead catfish across both years, and found that stomachs tended to be more full ($P < 0.001$) and fish were in better condition ($P < 0.001$) during the year with floodplain connectivity. We also noted a lower occurrence of empty stomachs ($P = 0.003$), smaller size threshold for when flathead catfish shifted to piscivory during the flood, and differences in overall diet structure in small flathead catfish between 2011 and 2012. Future work to assess large river food web dynamics during floodplain connectivity will help resource managers determine the magnitude and frequency of flooding needed to maintain or enhance large river fisheries.

INTRODUCTION

Large river systems in their unaltered state exhibit dynamic annual and seasonal patterns in discharge, including regular and relatively predictable connectivity with floodplain habitats. Contemporary paradigms in large river ecology predict that regular flooding causes terrestrial and riparian nutrients to become available for primary production in riverine ecosystems (Junk et al. 1989; Schlosser 1991). Increases in production at lower trophic levels during floodplain inundation are believed to result in a bottom-up increase in production across several trophic levels (Junk et al. 1989; Thorp et al. 2008), but few studies have examined the effect of floodplain access on the diet of fish occupying upper trophic levels. Balcombe et al. (2005) compared the diet of nine native fishes in Cooper Creek, an Australian floodplain river, and found that during both flood and non-flood conditions, diets were dominated by aquatic organisms despite increased terrestrial input during floodplain inundation. Balcombe et al. (2005) also noted that stomach fullness was similar during the two conditions, but that prey items were more diverse and of higher nutritional value during a flood. Luz-Agostinho et al. (2009) quantified differences in five South American piscivorous fishes in the upper Paraná River between flood years and dry years and found improved body condition in an ambush predator, which fed regularly during periods of flooding, but poorer body condition in other piscivorous species using other search tactics.

The flathead catfish *Pylodictis olivaris* is an apex predator endemic to the lower Missouri River system that consumes aquatic macroinvertebrates before transitioning to obligate piscivory as adults (Jackson 1999). Minckley and Deacon (1959) studied flathead catfish diet in the Big Blue and Neosho rivers in Kansas, and noted that diet

samples from flathead catfish smaller than 100 mm long were predominately composed of aquatic macroinvertebrates and almost never contained fish, whereas fish larger than 250 mm contained almost exclusively fish and crayfish. Other studies report the ontogenetic shift of flathead catfish from feeding on macroinvertebrates to fish occurs between 250 mm and 360 mm (Jackson 1999; Weller and Robbins 1999; Brewster 2007). Brewster (2007) and Minckley and Deacon (1959) propose that the exact size-at-shift depends on the availability of prey items. Under conditions where fish prey items are frequently encountered and do not require excessive handling time (i.e., the predator is not gape-limited), piscivory resulted in higher energy return and faster somatic growth than invertivory (Galarowicz and Wahl. 2005).

During summer and fall 2011, the mainstem Missouri River was connected with its floodplain after an unprecedented quantity of precipitation occurred in the watershed during the preceding winter and spring. In contrast, widespread drought occurred throughout the basin during 2012, and as a result, the Missouri River remained within its channel all year. The disparity in water conditions between 2011 and 2012 presented an opportunity to investigate the dietary response of flathead catfish to hydrological conditions. Therefore, we investigated the hypothesis that items eaten by flathead catfish differ between a year with floodplain connectivity and a year with no floodplain connectivity. Specifically, we compared stomach fullness, frequency of empty stomachs, condition, size-at-shift to piscivory, and overall taxonomic structure of diet items in flathead catfish between a flood and non-flood year in the Missouri River.

METHODS

Study Area

We collected flathead catfish from five sites along the Missouri River, Nebraska during July, August, and September 2011 and 2012 (Figure 3-1). We sampled each site during 2011 for two consecutive days every two weeks, and along main channel bankline habitats at the same five locations during low water in 2012.

Data Collection

We collected flathead catfish by daytime boat electrofishing using a Smith-Root 5.0 GPP control box at 15 Hz, 5 amp and 60 Hz, 8 amp settings. Fish were measured for total length to the nearest millimeter (mm) and mass to the nearest gram (g). Pulsed gastric lavage (Foster 1977; Light et al. 1983; Kamler and Pope 2001) was used to remove stomach contents from fish ≥ 300 mm, whereas a 500 mL wash bottle was used to remove stomach contents from fish < 300 mm to reduce the risk of stress or injury. All samples were stored in 10% buffered formalin to cease digestive processes. Stomachs containing no animal prey items were recorded as empty, and all fish were released alive immediately after processing.

Data Analysis

Prey items were identified, counted, and weighed to the nearest 0.001 g. Analyses included percent by number and percent by weight for prey items from each taxon within a stomach, and frequency of occurrence for each prey taxon within 25 mm

flathead catfish length groups. All analyses using flathead catfish length groups used only groups where at least five individuals were sampled in a given year.

Stomach fullness (F_s) is a measure of the percentage of maximum stomach capacity containing food items. Stomach fullness for each individual containing food items was calculated as:

$$F_s = \frac{V_s}{C} ,$$

where V_s is the cumulative volume of the stomach contents and C is the maximum capacity of the stomach. Stomach capacity was estimated by plotting sample volume (i.e., the cumulative volume of all prey items in a stomach) by total length similar to Gosch et al. (2009). Fish were divided into their respective length groups, and an exponential regression was fit to the largest data point within each length group to predict maximum stomach capacity based on fish length.

Frequency of empty stomachs (FES) is the proportion of fish in a length group with no prey items in their stomach and is calculated as:

$$FES = \frac{\text{NumE}}{\text{NumT}} ,$$

where NumE is the number of fish collected within a length group with empty stomachs and NumT is the total number of fish sampled in the length group.

Fish condition indices are widely used to assess the well-being of individual fish by comparing their weight-at-length to weight-at-lengths from across the species' distribution. We calculated relative weight for all individuals collected that were larger than the minimum length (130 mm) specified by Bister et al. (2000) using methods outlined by Anderson and Neumann (1996).

Mean stomach fullness, frequency of empty stomachs, and relative weight were compared between years with an analysis of covariance (ANCOVA), using fish length as the covariate. Stomach fullness and frequency of empty stomach values were arcsine square root-transformed prior to analysis. The critical alpha level for determining significance was $\alpha = 0.05$. All tests were done using PROC GLIMMIX in SAS version 9.2 (SAS 2009).

Ontogenetic shift in diet was evaluated using a modified Index of Relative Importance (IRI; Pinkas et al. 1971; Bowen 1996). The original IRI is calculated as:

$$IRI = (N_i + V_i) \times O_i,$$

where N_i is the percent by number of prey item i within a stomach, V_i is the percent by volume of prey item i within a stomach, and O_i is the frequency with which prey item i occurs within a given length group. We substituted percent by volume with percent by mass, as precise measurements of mass are more easily attained for small prey items such as aquatic invertebrates. Percent by mass and percent by volume are often used interchangeably in diet analyses to quantify a prey item's contribution to total prey biomass (Garvey and Chipps 2012).

We assessed ontogenetic shift in diets by consolidating all invertebrate prey items, except crayfish, within each sample to calculate an invertebrate IRI score. We then consolidated all vertebrates and crayfish within each sample to calculate a vertebrate-and-crayfish IRI score. Crayfish and vertebrates were grouped together because the searching and handling processes for flathead catfish preying on crayfish and fish are likely similar, such that these larger prey items probably require more time to find and handle than small aquatic invertebrates (Hoyle and Keast 1987). Mean vertebrate-and-crayfish IRI scores

and invertebrate IRI scores were plotted by flathead catfish length group, and the shift from invertebrate-dominated to vertebrate-and-crayfish-dominated diet was declared when the vertebrate-and-crayfish IRI score surpassed and remained higher than the invertebrate IRI score. Fish smaller than the size-at-diet shift were classified as “invertivores,” and fish larger than the size-at-shift were classified as “piscivores” when comparing flathead catfish diet contents from different size groups between years.

We compared flathead catfish diet structure between high and low water years using analyses in Primer-E version 6 (Clarke and Gorley 2006). Differences between years were tested separately for “piscivorous” classified flathead catfish based on frequency of occurrence data for vertebrate and crayfish prey taxa and “invertivorous” classified flathead catfish based on frequency of occurrence data for invertebrate prey taxa. Frequency of occurrence data were square-root transformed to down weight extreme prey item occurrence values, and Bray-Curtis resemblance matrices were constructed. One way Analysis of Similarity (ANOSIM; Clarke 1993) was used to test for differences between years for each diet group, and a two way ANOSIM was used to test for diet content differences between years and between samples collected upstream and downstream of the Platte River confluence. Global R and P statistics provided by the ANOSIM were used to determine significance, where values of the R statistic typically range from 0 to 1, with values closer to 0 indicating little or no separation among groups (i.e., groupings are no better than randomly placing observations into groups), whereas values closer to 1 indicate more significant separation of *a priori* groups. The P statistic also ranges from 0 to 1, and is interpreted similarly to other frequentist statistical tests such that values less than a pre-determined critical alpha (e.g., 0.05) are assumed

significant. Clarke and Gorley (2006) recommend relying more on the Global *R* statistic to determine significance in cases with uneven sample sizes among groups, because *R* statistics are less sensitive to sample size unevenness across groups than the accompanying *P* statistics. A SIMPER analysis (Clarke 1993) was used to determine taxa-specific contribution to overall dissimilarities between years where differences were significant. Non-metric multidimensional scaling (NMDS) was used to visually inspect separation between years within trophic groups.

RESULTS

A total of 629 flathead catfish was collected during 2011 and 2012. Sampling during 2011 consisted of 585 electrofishing runs lasting approximately 2900 minutes and resulted in the collection of 171 flathead catfish. Sampling during 2012 consisted of 52 electrofishing runs lasting approximately 390 minutes and resulted in the collection of 458 flathead catfish. Prey items were present in 343 (55%) of the captured fish. Stomach contents included organisms from 12 invertebrate families in seven orders and six fish families in four orders. Other vertebrate prey items occurred in two families from two orders. A complete description of prey items collected from all 343 fish is provided in Appendix 2.

A regression of largest stomach sample volume by length group to predict stomach capacity yielded the equation:

$$C = 7.0 \times 10^{-7} (\text{TL})^{2.9726},$$

where *C* is the maximum stomach capacity in ml and *TL* is the fish's total length in mm (Figure 3-2). The results of the ANCOVA accounting for fish length suggest that mean

stomach fullness was greater during 2011 ($F_{1,340} = 79.85$; $P < 0.001$; Figure 3-3). Fish collected during 2011 also had a lower occurrence of empty stomachs (29%) than fish collected during 2012 (52%; $F_{1,30} = 10.18$; $P = 0.003$), and higher mean relative weight in 2011 (109) than in 2012 (96; $F_{1,624} = 88.64$; $P < 0.001$).

The transition from invertebrate-dominated diet to a fish-and crayfish-dominated diet occurred at 200 mm in 2011, but flathead catfish maintained an invertebrate-dominated diet until 350 mm in 2012 (Figure 3-4). However, we did observe at least some unidentifiable fish material (e.g., bones, muscle, scales, etc.) in about 50% of diet samples we classified as invertivorous in 2011 and in about 7% of invertivorous classified samples in 2012 (Table A2-1).

Comparisons of flathead catfish diet based on prey taxa frequency of occurrence between years showed differences in diet structure for flathead catfish primarily consuming invertebrates (Global $R = 0.951$, $P = 0.100$), whereas differences in flathead catfish with fish-and-crayfish dominated diets were not significant between years (Global $R = 0.132$, $P = 0.113$). Non-metric multidimensional scaling reinforces the ANOSIM results where there was separation of invertivores between years and overlap in piscivores between years (Figure 3-5). Prey taxa accounting for differences in “invertivorous” diet structure between years include larval mayflies in family Heptageniidae (0% occurrence in 2011, 18% occurrence in 2012), larval damselflies in family Coenagrionidae (13% occurrence in 2011, 0% occurrence in 2012), and larval mayflies in family Isonychiidae (50% in 2011, 82% in 2012; Table 3-1). Diet contents did not differ among invertivorous (Global $R = 0.212$; $P = 0.008$) or piscivorous (Global

$R = 0.01$; $P = 0.366$) flathead catfish between samples collected upstream and downstream of the Platte River confluence.

DISCUSSION

Large river food webs are comprised of complex and dynamic interactions among organisms, and logistical sampling constraints often preclude scientists from understanding large river food webs in their entirety. However, studies of smaller-scale interactions provide valuable insight to overall food web structure. We found that the size-at-shift to piscivory, mean stomach fullness, and the frequency of empty stomachs differed between 2011 and 2012 for both flathead catfish trophic groups, whereas differences in diet sample composition between 2011 and 2012 were only different for invertivorous flathead catfish. Greater abundances of juvenile fishes were observed during 2010 and 2011 flood events (Steffensen et al *in review*), and differences in flathead catfish diet characteristics between 2011 and 2012 are likely the result of increased prey fish production caused by the incorporation of floodplain resources into the river food web.

Shifting from invertivory to piscivory has bioenergetic benefits including increased growth rate and survival in juvenile fishes (Post 2003; Graeb et al. 2005). If the same is true for juvenile flathead catfish, the dietary transition at a smaller size during 2011 could have positively influenced year-specific growth and survival. We determined flathead catfish to be predominantly piscivorous at 200 mm in 2011 and at 350 mm in 2012 (Figure 3-4). These sizes are close to, but outside the size range over which the dietary transition is reported in the literature (e.g., between 250 mm [Minckley and

Deacon 1959] and 300 mm [Brewster 2007]). The drastic change in size-at-diet-shift between years is likely linked to changes in prey production and availability. Increased catch rates were documented for several age-0 Missouri River fishes during the 2011 flood compared to previous lower water years (Steffensen et al. *in review*). The increased abundance of age-0 fishes in the river during 2011 could have resulted in increased encounter rates between flathead catfish and prey fishes and facilitated the transition to piscivory at a smaller size. Additionally, flow velocities in the main channel during 2011 likely forced fish (both predator and prey) to seek refuge on the floodplain. Small-bodied fishes often occupy shallow (<0.5 m), slow-moving off channel areas during non-flooding periods (Ridenour 2007), and move laterally into floodplain habitats during flooding (Ross and Baker 1983). Prey fishes could have been more susceptible to predation while moving to and among floodplain habitat patches, and thus may have facilitated the shift to piscivory at a smaller size during floodplain inundation.

The disparate minimum sizes of flathead catfish consuming fishes between years may also emphasize their diet plasticity. The 150 mm difference in the size-at-first-piscivory between years likely means that some individual flathead catfish were large enough to be mostly piscivorous in 2011 (i.e., > 200 mm), but were smaller than the minimum piscivorous size (350 mm) in 2012. The ability of predators to adjust their diets to consume the most economical prey (in terms of time required for searching, handling, and energetic benefits received) was first addressed by Emlen (1966), and has been tested in many field studies since (e.g., Chapman et al. 1989; Galarowicz and Wahl 2005; Almeida et al. 2011). Diet shifts are often considered to be permanent, and to occur as increasing gape size allows a predator to efficiently handle larger prey; however,

predators in systems with variable prey production across years or seasons may need to adjust to consuming the most economical prey items that are available under current conditions.

Several methods can be used to estimate stomach capacity, including injecting measured quantities of water into an excised stomach until it is fully distended (Kimball and Helm 1971), and dividing fish into length groups, fitting a regression to the largest stomach content volume across length groups to estimate maximum stomach capacity for a fish of any given length (Gosch et al. 2009). We chose to use the latter method because it is non-lethal and likely gives a more realistic representation of food volumes that fish will consume. It is important to remember that the regression technique we used assigns stomach fullness values greater than 100% to some individuals. This limitation leads us to suggest that stomach capacities derived through this process be viewed as a standard benchmark for comparisons rather than an absolute estimate of maximum capacity.

Increased stomach fullness, condition, and decreased occurrence of empty stomachs during 2011 in flathead catfish is probably the result of increased prey production and flathead catfish shifting to consuming vertebrate prey at a smaller size. Flathead catfish between 200 mm and 325 mm mostly consumed vertebrates and crayfish during 2011, but consumed almost exclusively small invertebrates during 2012. The difference in body size between vertebrate/crayfish prey and invertebrate prey would require flathead catfish to consume considerably more invertebrates to accumulate the same food volume in their stomach as one fish or crayfish. Additionally, larger prey items have been shown to require more time to digest than smaller prey (Garvey and Chipps 2012). Stomach fullness has also been correlated to overall caloric intake (Pope et al.

2001), and fish that spend less time with empty stomachs between feeding events are more likely to maintain a positive energy balance (i.e., consuming more energy than needed for survival; Arrington et al. 2002). Therefore, increased condition and decreased occurrence of empty stomachs during 2011 suggest that flathead catfish were more likely to maintain a positive energy balance and experience increased fitness during the flood as a result of increased food volume and reduced likelihood of having empty stomachs.

Differences in diet structure between years only occurred in flathead catfish primarily consuming aquatic invertebrates. Among the differences noted between years was the overall decrease in taxonomic diversity during low water in 2012 (Table A2-1). Balcombe et al. (2005) also cite decreased diversity in diet composition in fish captured during low water. Diversity and abundance of aquatic macroinvertebrates have been shown to decline during periods of low water, and quickly rebound during years with higher water (Boulton 2003). Similar stomach contents in piscivorous flathead catfish between years is possibly caused by prey fishes remaining susceptible to predation for multiple years post-hatch, making diet structure less reflective of age-0 production in a given year. Flathead catfish with invertebrate-dominated diets, however, are more limited to consuming prey items that were produced during current hydrological conditions, as most larval aquatic insects only spend a portion of one season as larvae before emerging as adults (Daly et al. 1998). The effects of hydrological conditions on prey production and fish diet, therefore, are probably most observable in invertivorous fishes and fishes consuming age-0 fish prey.

We found that organisms in regulated rivers did respond to floodplain connectivity when it occurred, despite decades of flow regulation to prevent overbank

flow on the Missouri River. Determining biotic responses to floodplain connectivity in terms of the frequency, timing, and duration of flooding necessary to restore or maintain large river ecosystems, yet allow large rivers to be used to satisfy human needs is paramount.

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Table 3-1. Similarity (SIMPER) analysis showing prey taxa that account for a cumulative 75% of diet dissimilarity for flathead catfish primarily consuming invertebrates during 2011 and 2012. Occurrence values are square root-transformed frequency of occurrence for each taxon by 25 mm flathead catfish length group represented by at least five individuals. The “Percent Contribution” column reports the percentage of total dissimilarity contributed by each taxon.

Taxon	2011 Average Occurrence	2012 Average Occurrence	Percent Contribution
Heptageniidae mayflies	0.00	4.27	23.60
Coenagrionidae damselflies	3.54	0.00	20.06
Unidentifiable insect	3.54	0.00	20.06
Isonychiidae mayflies	7.07	9.08	11.53

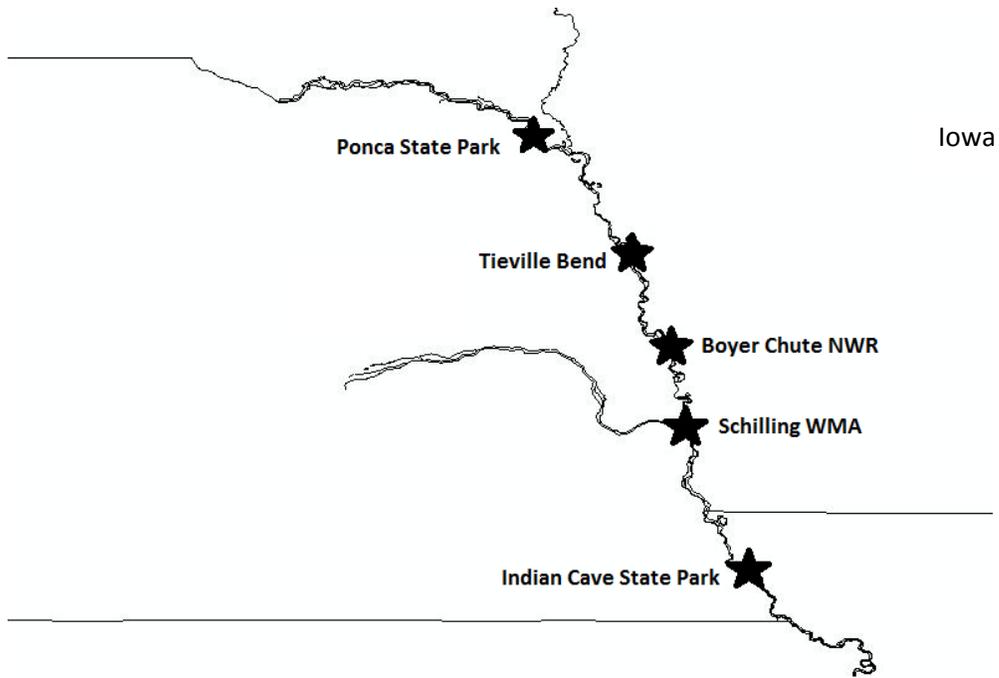


Figure 3-1. Map of study sites used to evaluate flathead catfish diet during 2011 and 2012.

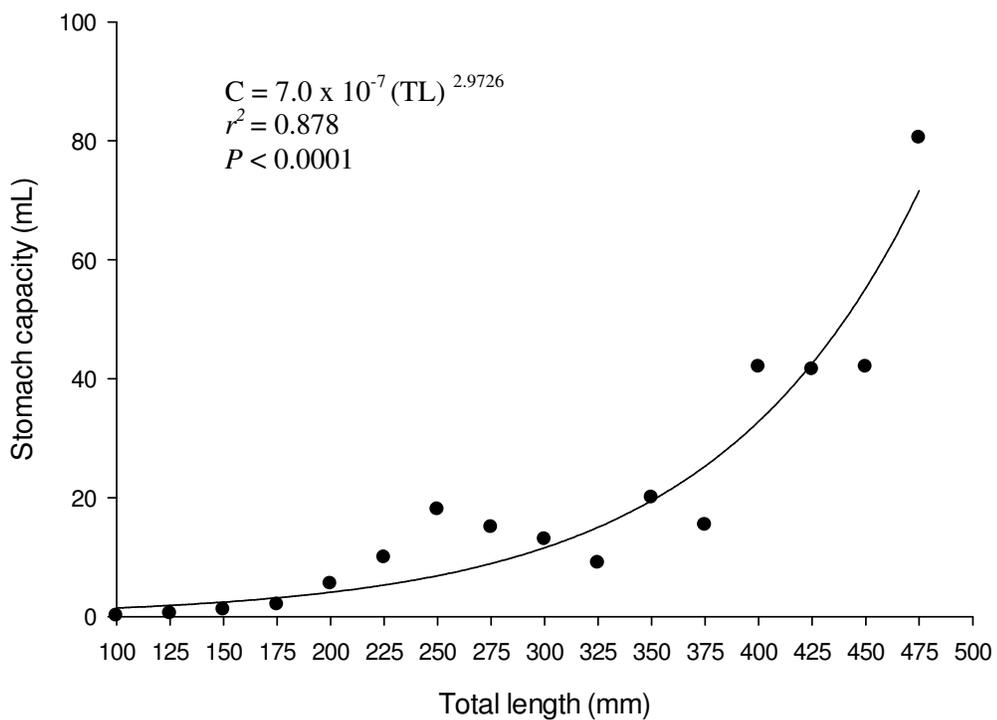


Figure 3-2. Stomach capacity equation generated from flathead catfish stomach sample volumes collected during 2011 and 2012 in the Missouri River, Nebraska.

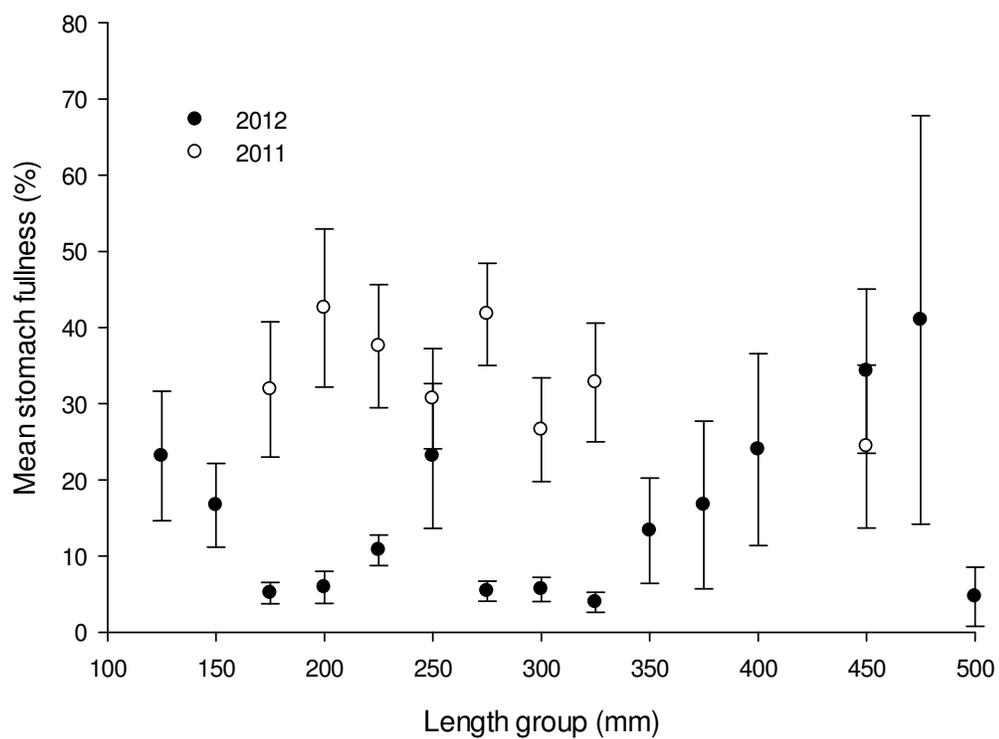


Figure 3-3. Mean \pm SE stomach fullness of flathead catfish collected in the Missouri River, Nebraska during 2011 (open circles) and 2012 (closed circles).

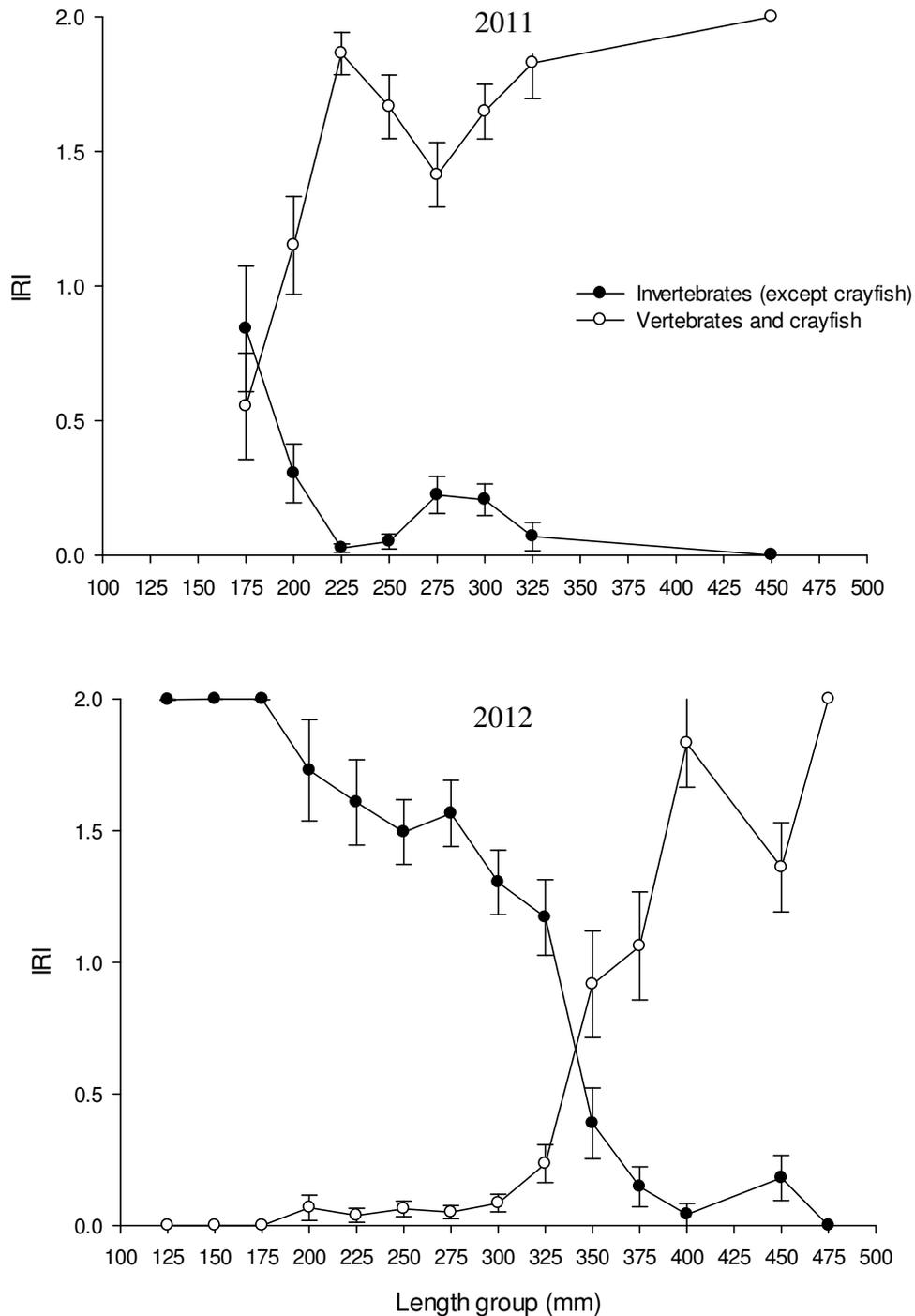


Figure 3-4. Mean \pm SE index of relative importance for invertebrate prey items and vertebrate/crayfish prey items found in flathead catfish stomachs during 2011 (top) and 2012 (bottom). Flathead catfish 200 mm or larger had diets dominated by vertebrates and crayfish in 2011, while flathead catfish 350 mm or larger had diets dominated by vertebrates and crayfish in 2012.

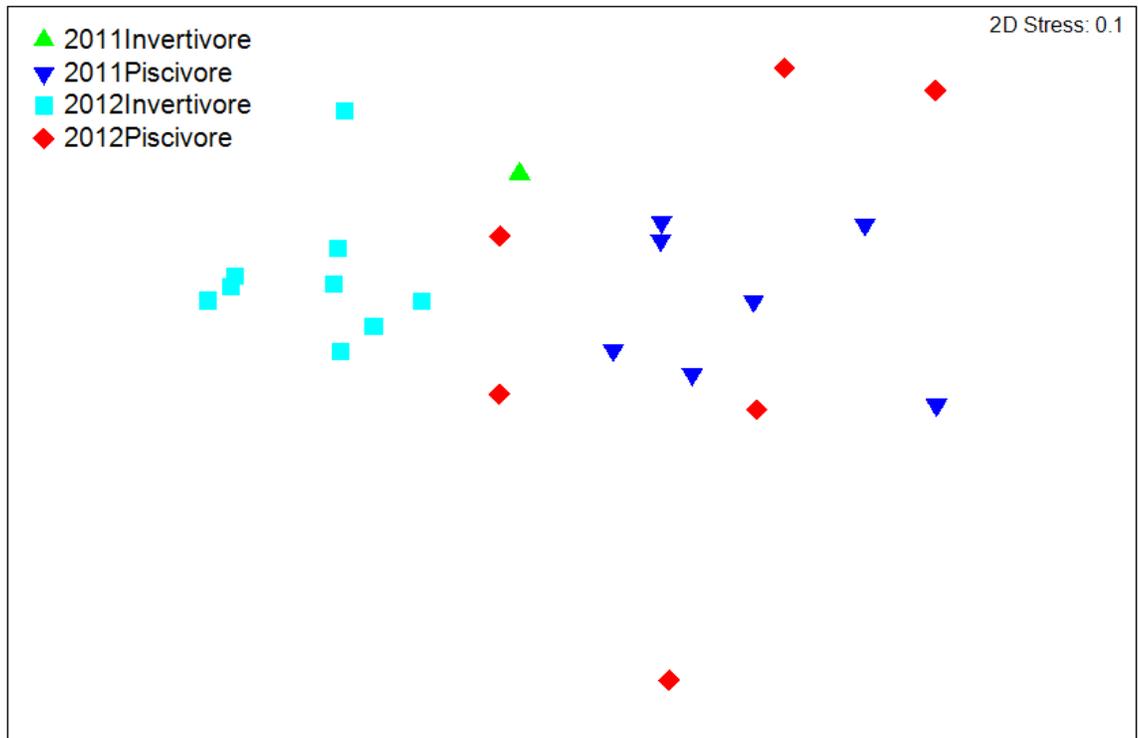


Figure 3-5. Non-metric Multidimensional Scaling plot of diet structure for invertivorous and piscivorous flathead catfish sampled in the Missouri River during 2011 and 2012. Structure data were based on the frequency of occurrence of prey taxa in each 25 mm length group represented by at least five fish.

CHAPTER 4

AGE-0 CHANNEL CATFISH *ICTALURUS PUNCTATUS* GROWTH RELATED TO ENVIRONMENTAL CONDITIONS IN THE CHANNELIZED MISSOURI RIVER, NEBRASKA

ABSTRACT

Flow regimes in large river systems are critical in determining and maintaining instream habitats and biotic community structure. Alteration of large river systems for human use is a worldwide issue threatening global freshwater biodiversity. Modifications to the Missouri River, Nebraska within the past 100 years have drastically reduced shallow water habitat, homogenized the flow regime, and contributed to declines in several native species. Large river paradigms suggest natural flow variability and connectivity with the floodplain and other off-channel habitats foster production and growth of aquatic organisms. The channelized, leveed, and flow-regulated Missouri River is rarely allowed to connect with its floodplain, but several characteristics of the flow regime still differ across years. We related age-0 channel catfish *Ictalurus punctatus* growth to environmental conditions in the channelized Missouri River, Nebraska between 1996 and 2013 using an information theoretic approach. Growth rate was most influenced by the duration of growing season and discharges below the 25th percentile of Missouri River discharge over the last 30 years. Periods of low water may be important for juvenile growth due to channel modifications that limit critical shallow water habitat during higher within-bank flows. The exclusion of peak discharge and peak discharge timing in the best model to predict growth is counter to conventional thoughts on river fish responses to hydrological conditions, but may be reflective of the modified flow regime during our study. Annual peaks in discharge during our study were often not

sufficient to connect the river with its floodplain, where increased growth rates may have occurred. Future efforts to relate juvenile fish growth to environmental conditions can provide guidance for water management in the Missouri River and other regulated North American rivers.

INTRODUCTION

Natural flow regimes have been identified as a critical factor maintaining the ecological integrity of river and riparian ecosystems (Junk et al. 1989; Poff et al. 1997; Bunn and Arthington 2002). A number of interacting attributes define a river's flow regime, including the timing, magnitude, duration, and frequency of high or low flow events (Poff et al. 1997; Rolls et al. 2012). These components of the flow regime result in fluvial geomorphic processes that determine abiotic attributes of a riverscape, including water quality and physical habitat structure. Physical habitat characteristics and water quality, in turn, directly or indirectly determine the structure of floodplain, riparian, and riverine biotic communities (Poff et al. 1997). Specifically, large river fish populations and communities are directly influenced by discharge patterns and the resultant effects on habitat and water quality (Junk et al. 1989; Poff and Ward 1989; Lytle and Poff 2004).

Despite the importance of natural flow regimes to large river structure and function, habitat modifications and flow regulations have been applied to most large rivers worldwide to meet human needs (Dudgeon et al. 2006; Poff et al. 2007). Flow regulation (e.g., reducing flood pulses and increasing flow during periods of low discharge) has been helpful in protecting human establishments on floodplains and providing water for human use all year, but is among the most obtrusive and widespread

modifications to large river ecosystems. Large river impoundments have altered longitudinal connectivity, sediment dynamics, and thermal regimes; whereas seasonal flow homogenization and channel modifications have reduced instream physical habitat heterogeneity and homogenized flow regimes (Poff et al. 2007). Dudgeon et al. (2006) and Vorosmarty et al. (2010) also identified modifications to large river ecosystems as contributing factors to declines in worldwide freshwater biodiversity.

Ecological communities respond quickly to changes in hydrological conditions (e.g., Freeman et al. 2001; Shafroth et al. 2002; Poff and Zimmerman 2010), and characteristics of the flow regime in a given year have been shown to influence year class strength (Schlosser 1985; Freeman et al. 2001; Agostinho et al. 2004) and growth (Weisberg and Burton 1993; Jones and Noltie 2007) of large river fishes. Population-scale growth rates are particularly insightful because they serve as a composite measure of biotic and abiotic conditions that contribute to a population's ecological success (DeVries and Frie 1996; Rypel 2011). Many ecological processes and interactions among aquatic organisms are mediated by body size (Slaughter and Jacobson 2008; Rypel 2011; Chapter 3 of this thesis), and thus, faster growth rate is generally considered ecologically desirable. Therefore, understanding which components of environmental conditions affect fish growth is critical for managing fish populations. The objectives of this study were to investigate relations between age-0 channel catfish growth and environmental conditions experienced by each year's cohort in the channelized Missouri River between 1996 and 2013. Specifically, we tested for differences in growth rate among years and assessed the influence of annual flow pulse magnitude, duration, and timing, and growing season duration on age-0 channel catfish growth.

METHODS

Study area

The Missouri River is among the most drastically altered large river systems in North America. A series of six mainstem impoundments was constructed in the upper and middle Missouri River between 1937 and 1963 to secure water for navigation, irrigation, municipal use, hydropower, and flood control (U.S. Army Corps of Engineers 2006). Operations of these dams have severely altered the Missouri River hydrograph, replacing seasonal flow pulses in the middle and lower river with relatively stable discharge across seasons (Hesse and Mestl 1993; Pegg et al. 2003). Consequently, declines have been noted for several native Missouri River fishes during the post-impoundment era (Galat et al. 2005).

We used channel catfish data collected from the Missouri River between Sioux City, Iowa (river kilometer [RKM] 1178) and the Nebraska-Kansas border (RKM 789; Figure 4-1). Channel catfish growth and environmental data were collected separately from two segments; an upper segment between Sioux City, Iowa and Plattsmouth, Nebraska (RKM 957) and a lower segment between Plattsmouth, Nebraska and the Nebraska-Kansas border. Environmental conditions differ between these segments due to inputs from a major tributary (i.e., the Platte River; Pegg and Pierce 2002). Environmental data were collected by the United States Geological Survey and United States Army Corps of Engineers gaging stations at Omaha, Nebraska (RKM 991) for the upper segment and Nebraska City, Nebraska (RKM 904) for the lower segment.

Data collection and analysis

Age-0 channel catfish data were collected by the Nebraska Game and Parks Commission (NGPC) between 1996 and 2013 and the Pallid Sturgeon Population Assessment Program (PSA) between 2003 and 2013. We used length-frequency analysis (Quist et al. 2012), along with previous length-at-age information for Missouri River channel catfish (Goble 2011), to isolate age-0 channel catfish by plotting total length by capture date and visually inspecting for clusters of points representing annual cohorts. We limited analyses to age-0 fish because they allocate nearly all of their excess caloric intake to somatic growth, and therefore their growth rate is more indicative of annual differences in habitat suitability and food production. Growth in older fish is still affected by annual differences in habitat and food production, but is also affected by changes in behavior and resource allocation toward gamete production (Pegg and Pierce 2001). Visual inspection of length distributions revealed consistent discontinuities (> 20 mm) that first appeared in mid-summer and continued to occur at larger sizes through autumn (Figure 4-2). We assumed this discontinuity represented separation between age-0 and age-1+ channel catfish. Therefore, all fish larger than the length discontinuity were assumed to be age-1+ and were excluded from analysis. Growth rate was estimated as the slope of the regression of total length versus day of year captured for each year using the GLM procedure (SAS 2009; Pegg and Pierce 2001). We used Analysis of Covariance (ANCOVA; SAS 2009) to test homogeneity of slopes across years for each segment to determine whether growth rate differed between years.

Previous studies have used flow attributes including the timing, magnitude, and duration of high and low discharge to characterize flow regimes (e.g., Richters et al.

1996). Additionally, growing season duration has been used as an explanatory variable for fish growth (e.g., Allan 1995; Pegg and Pierce 2001). We used four continuous variables (duration of high discharge, duration of low discharge, peak discharge timing, and growing season duration) and one categorical variable (river segment [Table 4-1]) to construct seven *a priori* linear models after inspecting each pair of continuous variables for autocorrelation. Low and high discharges are identified in several studies as critical components of the flow regime for fish recruitment and growth (e.g., Junk et al. 1989; Humphries et al. 1999). Therefore, our candidate models were constructed to test hypotheses related to low discharge duration and timing and high discharge duration and timing. Growing season duration has been shown to have a direct effect on fish growth (e.g., Arnold et al. 2013), and was included in all models except the null. We ranked candidate models and accounted for each model's uncertainty using Akaike's Information Criterion for small samples (AIC_c ; Hurvich and Tsai 1995; Program R; R Development Core Team 2012). This approach ranks candidate models based on AIC values similar to traditional AIC testing, such that the best candidate model has the lowest AIC_c value, but is less biased in cases where sample size is small. Differences between models are quantified by the ΔAIC_c value, which is calculated as the difference between a lower-ranking model's AIC_c value and the best-fitting model's AIC_c value. Akaike weights (w_i) are computed for each model to indicate how well each model fit the data relative to the other candidate models (Burnham and Anderson 2002).

RESULTS

A total of 11,131 age-0 channel catfish were collected in both segments between 1996 and 2013. The data set for the upper segment included 6,396 age-0 channel catfish; whereas 4,735 age-0 channel catfish were collected in the lower segment. The homogeneity of slopes test revealed different growth rates among years in both segments. Visual inspection of slopes for each year showed a general increase in growth rate between 1996 and 2013 in both segments (Figure 4-3).

The candidate model that best fit age-0 channel catfish growth rates incorporated river segment, growing season duration, and low discharge duration, and is represented by the equation:

$$GR = 1.7768 - 0.1129(RS) - 0.0063(GSD) + 0.0012(LDD),$$

where GR is juvenile channel catfish growth rate (mm/day), RS is river segment (upper = 1, lower = 0), GSD is growing season duration, and LDD is low discharge duration. This model accounted for about 38% of the variability in juvenile channel catfish growth ($r^2 = 0.38$) and carried about 61% of the weight compared to the remaining six models. All seven candidate models and their summary statistics are provided in Table 4-2. Parameter coefficients are provided for all models with ΔAIC_C less than 12. Low discharge duration increased between 1997 and 2006 before a declining trend began in 2007 and continued through 2011 (Figure 4-4). Growing season duration was longest between 2000 and 2006 in both segments, and was longer in the lower segment during most years (Figure 4-5). Flow parameters that were used to construct candidate models, but that were not included in the best-fitting model include duration of discharges greater than the 30-year 75th percentile (Figure 4-6) and peak discharge timing (Figure 4-7).

DISCUSSION

Juvenile fish growth in lotic systems is determined by several aspects of the environment. Our best-fitting model incorporated growing season duration and low discharge duration. An inverse relation between growing season duration and growth rate was not expected, as previous studies have established that juvenile channel catfish grow more quickly within a range of warm temperatures (e.g., Buentello et al. 2000; Arnold et al. 2013), and that growing season duration has a positive effect on fish growth (Neuheimer and Taggart 2007). Our dataset did not include any years with fewer than about 180 days $> 10^{\circ}\text{C}$ (Figure 4-5). It is possible that the optimum growing season duration for fish growth exists at or less than 180 days, and growing seasons much longer or shorter than the optimum result in slower growth. An additional explanation could be a counter-gradient response between growth and growing season. Conover (1990) assessed first-year growth of three North American fishes and found that fish collected in northernmost and southernmost latitudes of their range attained similar sizes by the end of their first growing season, despite the growing season being about 250% longer at lower latitudes. Pegg and Pierce (2001) found similar faster growth rates in adult emerald shiners *Notropis atherinoides* along a latitudinal gradient in the Missouri River. These studies assessed differences in growth rate along a spatial gradient, but no study has investigated a cohort's ability to respond to differences in growing season duration at a single location over time. Future studies to assess the effect of growing season duration on fish growth rate in large rivers will help elucidate the interaction between juvenile fish and their environment.

The best-fitting model also identified the duration of low discharge as important to juvenile catfish growth. The model coefficient for low discharge duration was positive, suggesting faster growth will occur in years with more days of low flow. Large river paradigms such as the Flood Pulse Concept (Junk et al. 1989) and the Natural Flow Regime paradigm (Poff et al. 1997) suggest that properly-timed periods of high discharge are critical for production in floodplain rivers, and that shallow, slow-flowing habitats on the floodplain and in off-channel waterbodies are important for prey production and fish rearing. However, high discharges in the contemporary Missouri River rarely inundate the floodplain due to the levee system and discharge regulation. Therefore, shallow water habitat is generally most abundant in the channelized Missouri River when discharges are low. Low discharges concentrate high velocities in the thalweg, allowing slow velocities and shallow depths to occur in wing dike fields and other limited off-channel habitats. Increased spatial variability in depths and velocities before channelization likely allowed juvenile fishes to locate suitable shallow water areas regardless of discharge. Efforts to reconstruct off-channel habitats may augment juvenile fish growth by providing shallow, slow moving habitats that will be available for use across a range of discharges.

Flow regulation in the channelized Missouri River has resulted in a drastically different flow regime than the pre-altered condition (Pegg et al. 2003). The magnitude and duration of most flood pulses during our study were modified by the reservoir system to prevent flooding, and the resultant lack of seasonal flow variability likely contributed to the low support for flow metrics related to peaks in discharge (i.e., peak discharge magnitude and timing) in the top candidate model. Hence, the catfish response to a properly-timed flood pulse of sufficient magnitude to inundate the floodplain remains to

be tested. Future water management actions should incorporate timing and magnitude of managed flood pulses to determine whether such operations elicit growth responses in juvenile fishes.

Anthropogenic modifications have had widespread effects on the ecological structure and function of many rivers, and future environmental changes are likely to further influence the hydrology and biology of managed rivers. A warming climate is expected to cause higher rain-to-snow ratios, shorter duration of snow and ice cover, earlier and faster snowmelt, changes in precipitation patterns, and warmer thermal regimes; all of which are likely to alter stream flow patterns and influence riverine organisms (Christensen et al. 2004; Ficke et al. 2007; Rahel and Olden 2008). Gradual increases in stream temperature are expected to occur over the next several decades, and are likely to reduce thermally-suitable habitat for cold and cool water fishes (Eaton and Scheller 1996; Rahel et al. 1996). Eaton and Scheller (1996) estimated the effect of warming water temperatures on 57 North American fish species, and predicted the distributions of some cool and cold water species to be reduced by up to 70%, while distributions of some warm water species could increase by up to 33% when atmospheric carbon dioxide concentrations were doubled from their concentration in the 1990s. Aside from instream temperature changes, climate change is projected to influence precipitation patterns in several watersheds. The Colorado River basin in North America, for example, is expected to receive about 82% more precipitation by 2050 than it did in 1960, while the Coppename River basin in South America is projected to receive only about 7% of the water in 2050 that it did in 1960 (Palmer et al. 2008). Differences in water quantity are projected to be less severe in the Missouri River basin, which is expected to receive

2%-5% less precipitation between 2041 and 2060 than it did between 1901 and 1970 (Palmer et al. 2009). Basins receiving relatively consistent precipitation quantities, such as the Missouri River basin, are still subject to hydrological changes caused by earlier and more rapid snowmelt, and the replacement of some snowpack with rainfall. Flow pulse timing and magnitude have long been identified as critical components of a natural flow regime (e.g., Bunn and Arthington 2002), and future changes in temperature regimes and precipitation dynamics could have notable effects on components of the flow regime that are important for aquatic organisms (Lytle and Poff 2004). Water storage systems, such as the Missouri River, may be able to mitigate changes in discharge caused by climate change by altering dam operation to mimic historic discharge patterns. Future water management actions in the Missouri River should be based on the pre-alteration flow regime to maintain important components of flow for aquatic organisms and buffer hydrological shifts caused by climate change. Future research should focus on identifying flow characteristics that are important for the success of aquatic organisms, including the refinement and validation of models relating characteristics such as growth to hydrological conditions.

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Table 4-1. Variables used to construct models explaining age-0 channel catfish growth in the Missouri River, Nebraska

Variable	Description
River segment	Upstream or downstream of Platte River confluence
Low discharge duration	Number of days discharge was less than the 25th percentile of mean daily discharges between January 1, 1984 and January 1, 2013 (Richter et al. 1996)
High discharge duration	Number of days discharge exceeded the 75th percentile of mean daily discharges between January 1, 1984 and January 1, 2013 (Richter et al. 1996)
Peak discharge timing	The day of year when peak discharge occurred
Growing season duration	Number of days each year that water temperature exceeded 10°C (Allan 1995; Pegg and Pierce 2001)

Table 4-2. Candidate models with parameter coefficients, y-intercept (Intercept), Akaike's Information Criterion (AIC_C), number of parameters (k), increase over the lowest AIC_C (ΔAIC_C), Akaike model weight (w_i), and coefficient of determination (r^2) for models fit to age-0 channel catfish growth rates in the channelized Missouri River, Nebraska between 1996 and 2013

Model	Intercept	AIC_C	k	ΔAIC_C	w_i	r^2
growth ~ - 0.1129(seg) - 0.0063(temp) + 0.0012(below)	1.7768	-18.47	5	0.00	0.61	0.38
growth ~ - 0.1428(seg) - 0.0067(temp) - 0.0006(above)	2.0366	-15.94	5	2.53	0.17	0.33
growth ~ - 0.1219(seg) - 0.0641(temp) + 0.0004(timepeak) + 0.0012(below)	1.7327	-15.7	6	2.77	0.15	0.36
growth ~ - 0.1421(seg) - 0.0067(temp) - 0.00002(timepeak) - 0.00006(above)	2.0390	-12.79	6	5.69	0.04	0.30
growth ~ - 0.1251(seg) - 0.0065(temp) + 0.0003(timepeak) - 0.0001(above) + 0.0011(below)	1.7876	-12.37	7	6.10	0.03	0.34
growth ~ 1		-2.65	2	15.82	< 0.01	
growth ~ seg		-0.61	3	17.87	< 0.01	

seg, river segment (upper = 1; lower = 0); temp, growing season duration (days > 10C); above, number of days with discharge > 75th percentile of 30 year discharge; below, number of days with discharge < 25th percentile of 30 year discharge; timepeak, day of year when peak flow occurred.

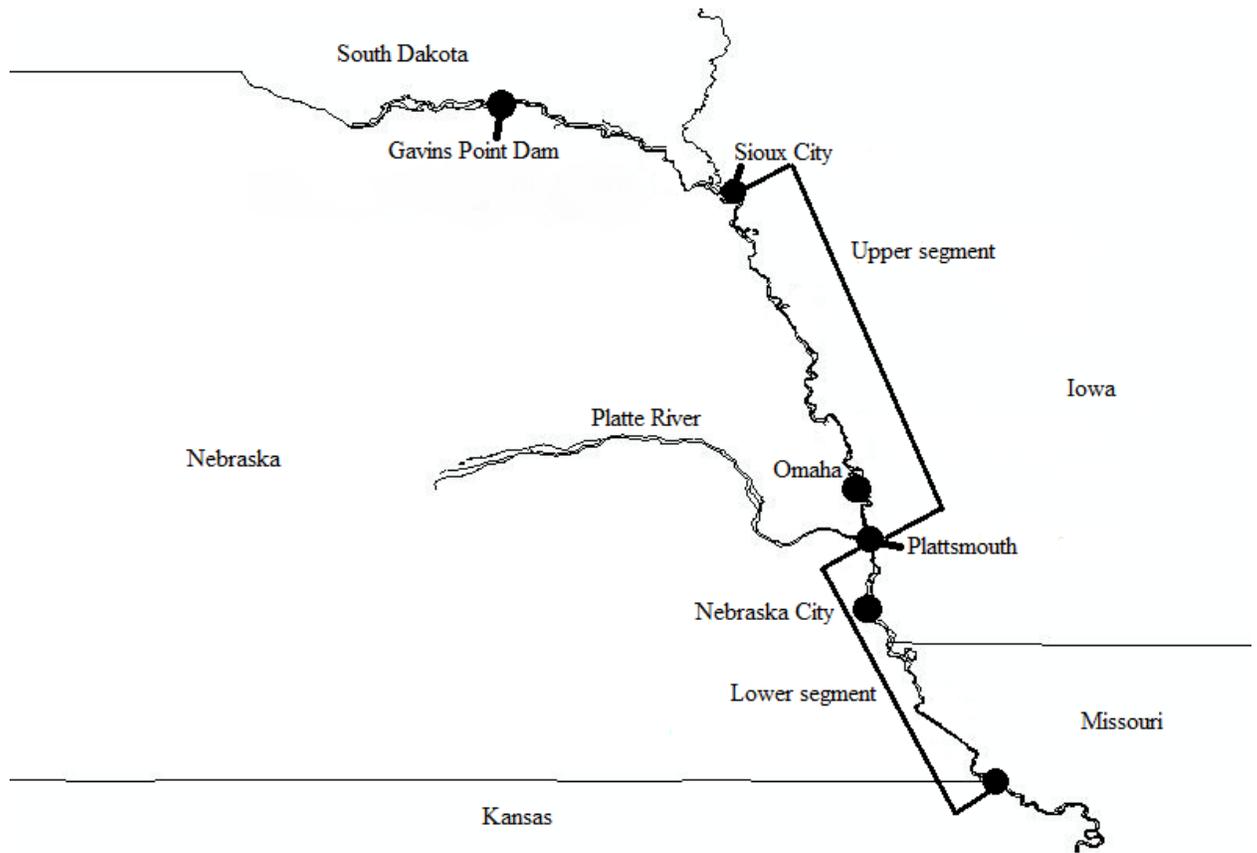


Figure 4-1. Map of the middle Missouri River showing upper and lower segments for age-0 channel catfish collection, Gavins Point Dam, and the location of gaging stations at Omaha, Nebraska and Nebraska City, Nebraska.

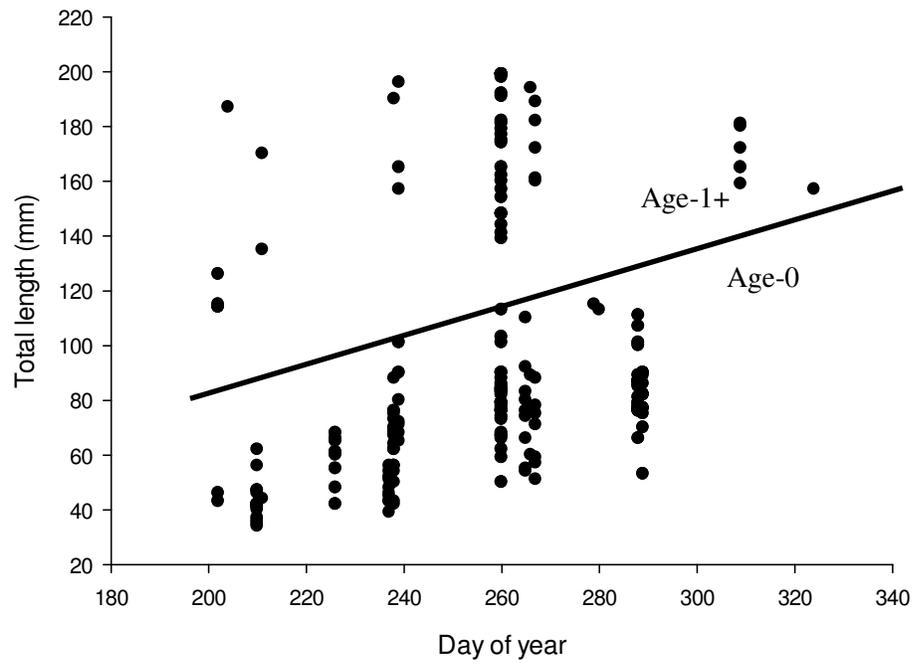


Figure 4-2. Example of length distribution of channel catfish collected in the Missouri River during. Fish below the black diagonal line were assumed to be age-0 and used for analysis; whereas fish above the black diagonal line were assumed to be at least age-1 and excluded from analysis.

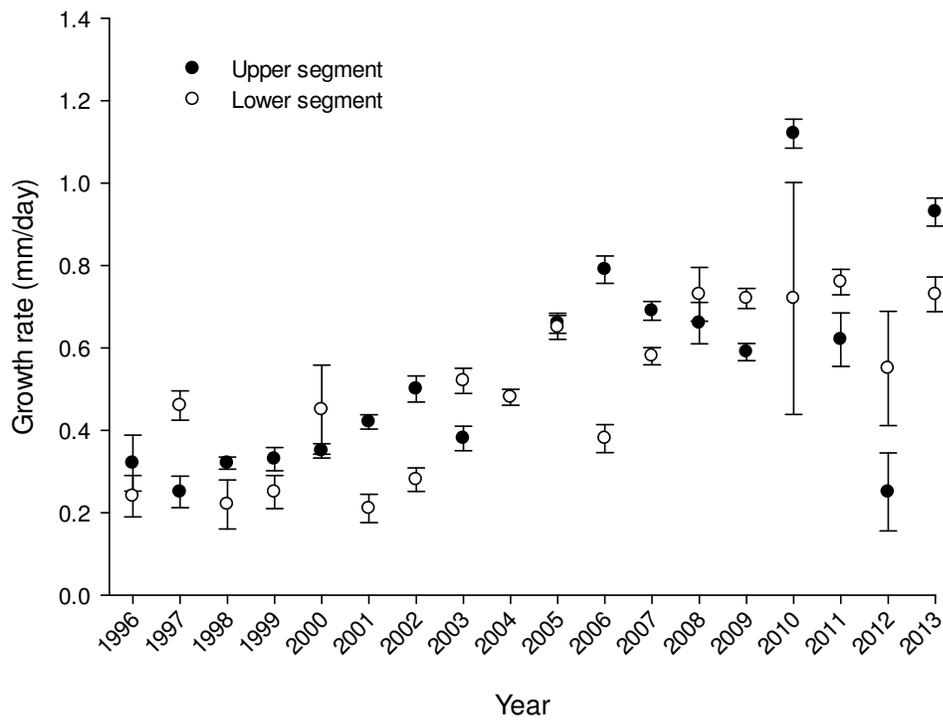


Figure 4-3. Mean \pm SE age-0 channel catfish growth rates by year in the upper (black points) and lower (white points) segments of the channelized Missouri River, Nebraska.

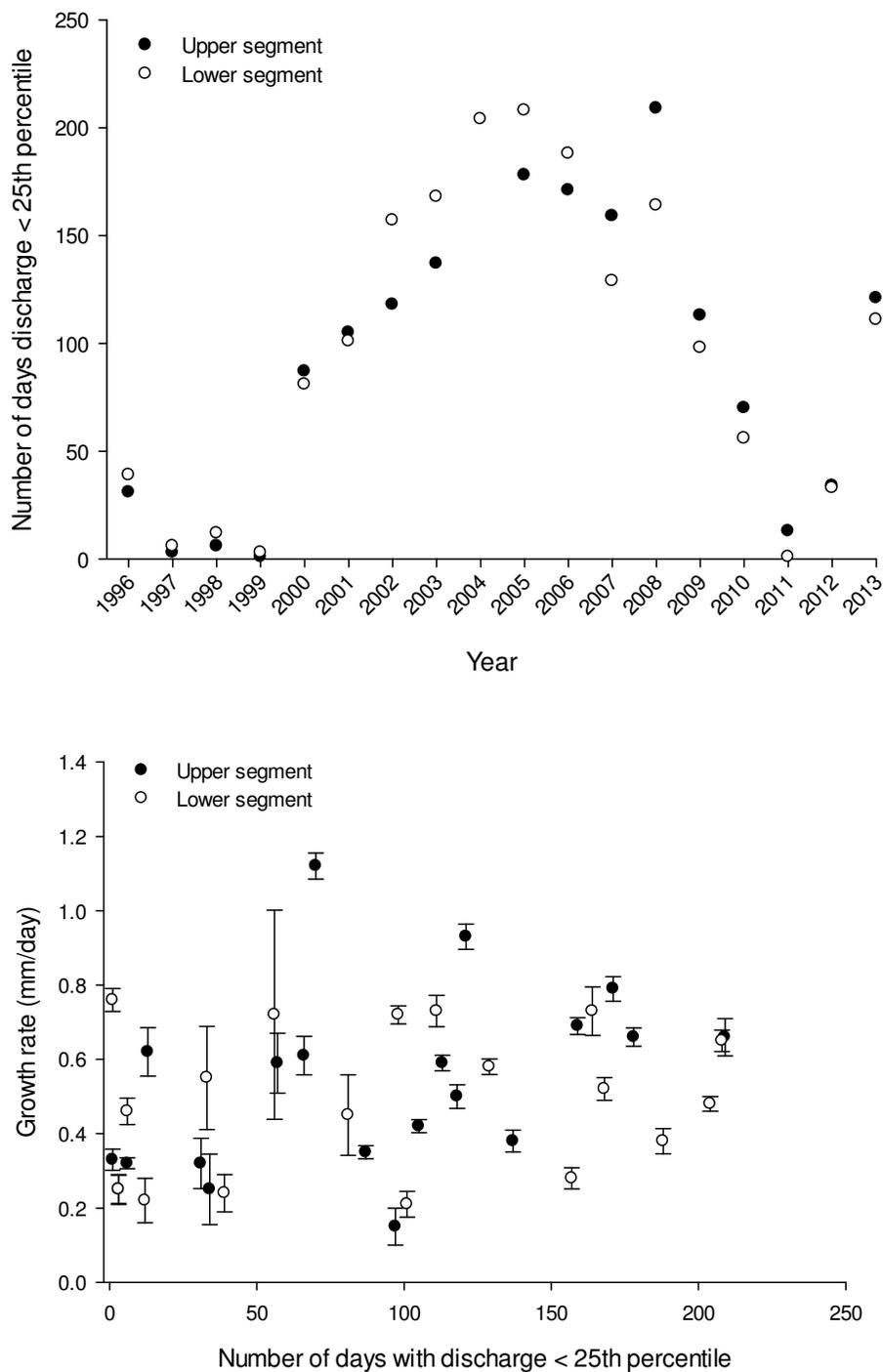


Figure 4-4. Scatterplot of the number of days each year that Missouri River discharge was less than the 25th percentile of mean daily discharges between January 1, 1984 and January 1, 2013 (top figure), and scatterplot of age-0 channel catfish growth rate versus annual duration of discharges less than the 25th percentile of mean daily discharges between January 1, 1984 and January 1, 2013 (bottom figure).

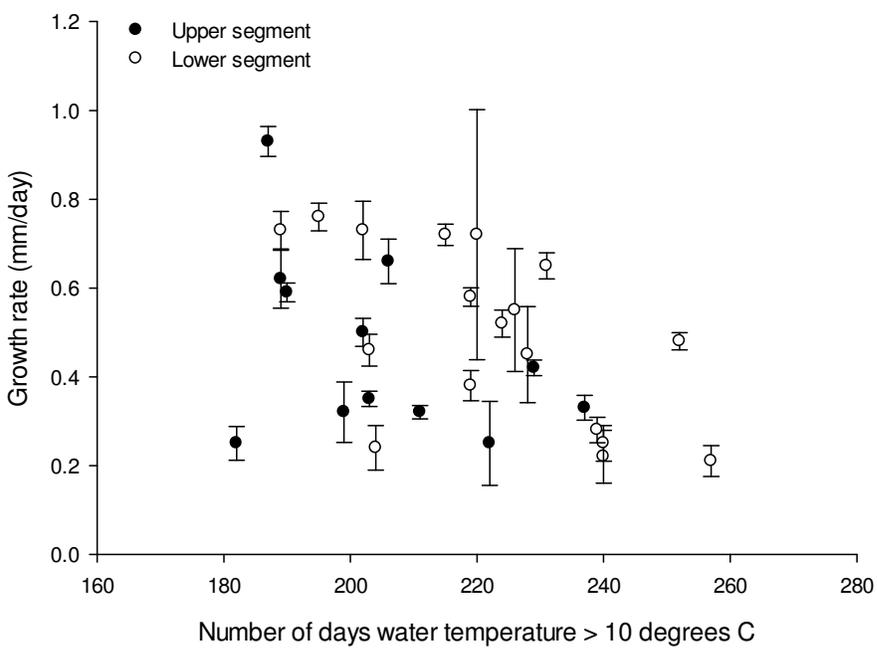
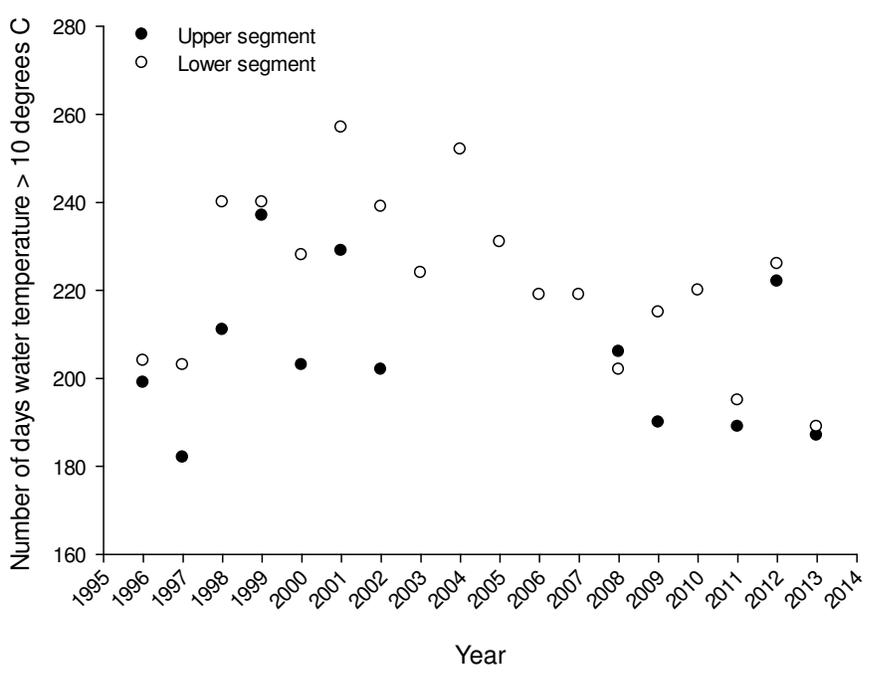


Figure 4-5. Scatterplot of the number of days each year that Missouri River water temperature exceeded 10° C (top figure), and scatterplot of age-0 channel catfish growth rate versus the number of days each year that water temperature exceeded 10° C (bottom figure).

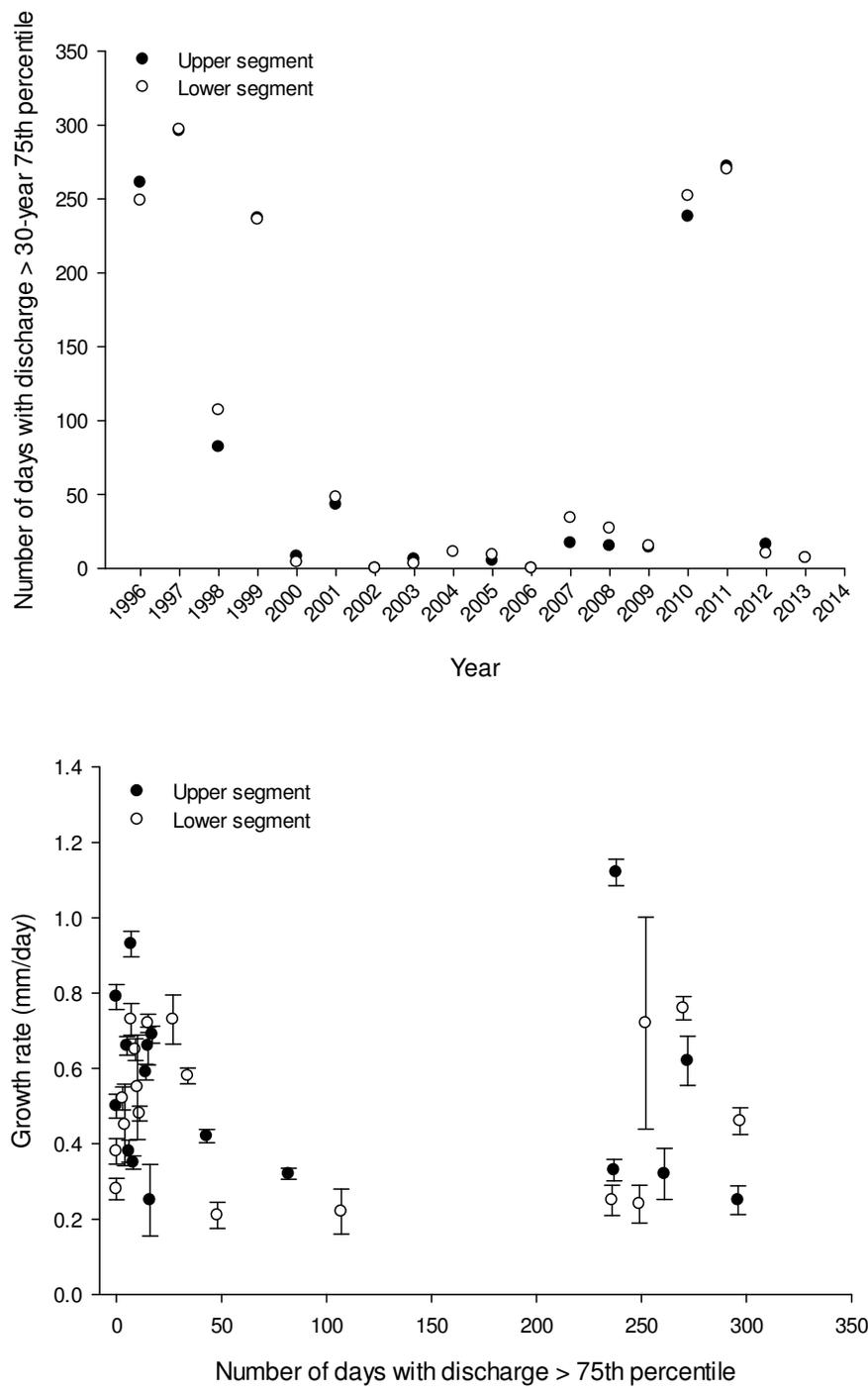


Figure 4-6. Scatterplot of the number of days each year that Missouri River discharge exceeded the 30-year 75th percentile of daily discharge between January 1, 1984 and January 1, 2013 (top figure), and scatterplot of mean \pm SE age-0 Missouri River channel catfish growth rates versus the number of days with discharge exceeding the 30-year 75th percentile daily discharge between January 1, 1984 and January 1, 2013 (bottom figure).

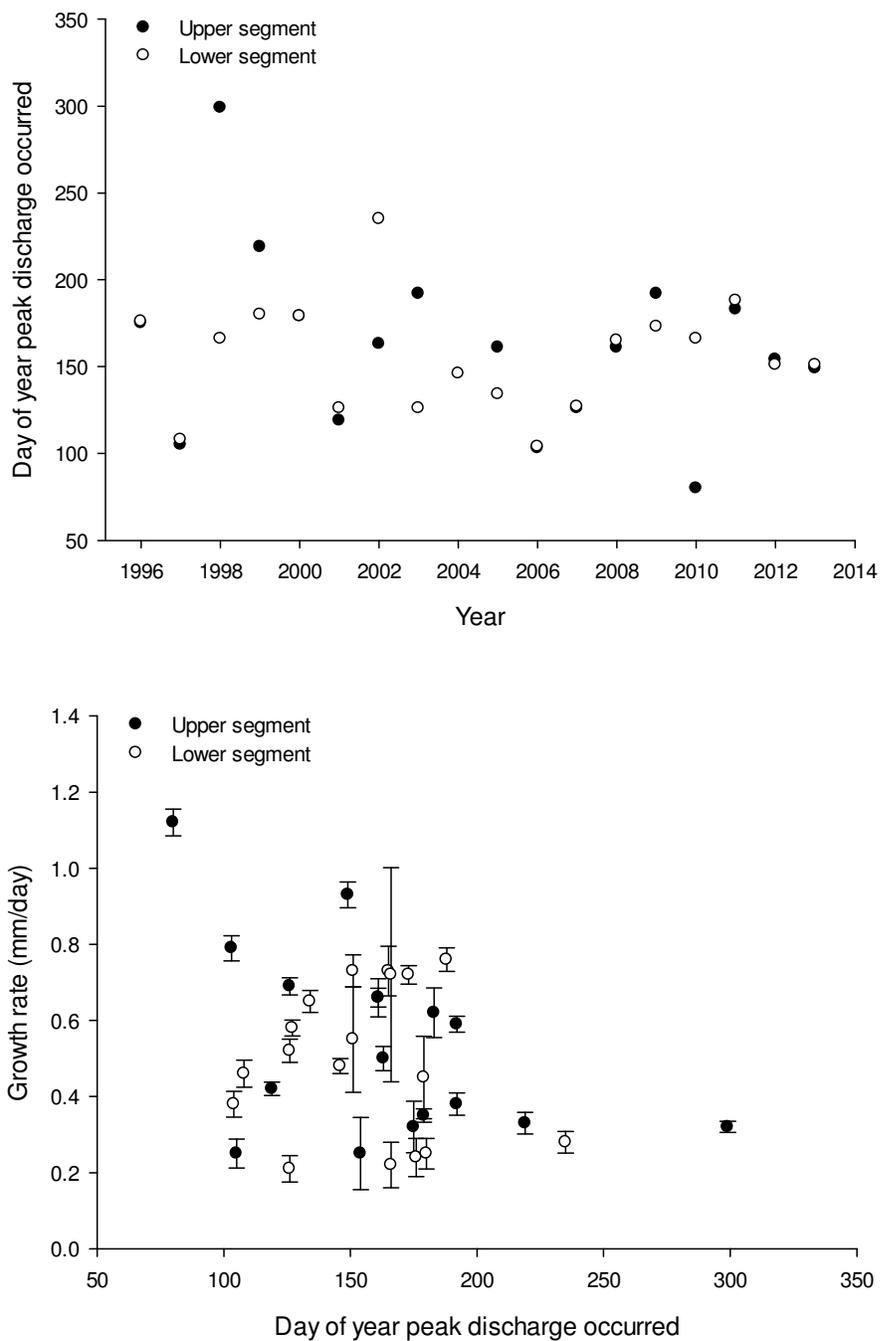


Figure 4-7. Scatterplots of Missouri River peak discharge timing by year (top figure) and mean \pm SE age-0 Missouri River channel catfish growth rates versus the timing (day of year) when peak discharge occurred (bottom figure).

CHAPTER 5

CONCLUSIONS AND MANAGEMENT CONSIDERATIONS

CHAPTER 2

FISH COMMUNITY COMPARISONS AND FLOODPLAIN HABITAT USE AT FIVE SITES IN THE FLOODED MISSOURI RIVER, NEBRASKA

CONCLUSIONS

An understanding of the fish community response to floodplain connectivity may benefit management and recovery in regulated rivers. This study described differences in fish community composition and structure on the Missouri River, Nebraska during the largest flood in over 60 years. Dissimilarity in community composition and structure among sites was generally driven by greater abundances of flathead catfish *Pylodictis olivaris* and common carp *Cyprinus carpio* at Indian Cave State Park and river carpsucker *Carpionodes carpio* and shovelnose sturgeon *Scaphirhynchus platyrhynchus* at Ponca State Park. The Tieville Bend fish community differed from the remaining sites primarily because of higher black bullhead *Ameiurus melas* abundance. Floodplain habitat measurements did not appreciably delineate species abundance or occurrence. This may indicate that appropriate habitat metrics were not recorded, or that homogenized floodplain physical habitat and the infrequency of floodplain connectivity during the past 50 years have led to a loss of habitat specialization among species that had preferred certain physical habitats pre-alteration.

MANAGEMENT CONSIDERATIONS

2.1 Investigate off-channel habitat use for species of conservation concern and monitor water quality of reconstructed off-channel habitats.

Understanding the attributes of off-channel habitat that relate to use by species of conservation concern will guide future habitat restoration efforts for their recovery. The U.S. Army Corps of Engineers initiated the Habitat Assessment and Monitoring Program (HAMP) in 2005 to assess the response of select fish species to recreated shallow water habitats in the lower Missouri River (Bryan et al. 2010). Continuing this research is critical to evaluating the efficacy of reconstructed shallow water areas, and will be especially helpful in evaluating biotic responses to regulated flood pulses in the future. Future research should include seasonal aspects of invertebrate and fish community structure in connected off-channel areas related to hydrological conditions in the main channel to identify areas more likely to be used by species of interest.

Additionally, maintaining sufficient connectivity between the river and off-channel habitats is necessary to ensure that environmental conditions in off-channel habitats remain suitable for fish use. Maintaining flow between habitats is also important to allow fish movement between these areas, even during periods of low river discharge. Insufficient connectivity between these areas, as is likely the case at Tieville Bend, leads to disconnected fish populations in floodplain lakes that are neither ecologically nor socially desirable. Studying fish passage between the river and off-channel areas would help us understand the degree of connectivity necessary to maintain desirable fish communities and ensure that fish passage is possible during high and low river discharges.

2.2 Evaluate additional habitat measures to delineate riverine habitats

Physical habitat measurements correlated poorly with fish catch data in my community analyses, suggesting that other habitat metrics may be needed to delineate

species occurrence. Li and Gelwick (2005) noted that depth, velocity, and substrate type were the most useful attributes in delineating fish habitat use in the Bravos River, Texas. Copp et al. (1994) found that water velocity and turbidity explained the most variability in fish floodplain habitat use in the Danube River. Water velocity was important in both the Bravos River and Danube River studies, and was included in eight of the top ten correlations in my study. Therefore, I recommend continuing to measure water velocity in future habitat use investigations. Smaller-scale velocity measurements (i.e., taking multiple velocity measurements throughout a sampling area to better describe changes in velocity over space) could provide better resolution for microhabitat use studies. Additionally, I recommend collecting substrate type and turbidity to describe these habitat attributes that may influence species occurrence in the Missouri River.

CHAPTER 3

DIETARY RESPONSE OF FLATHEAD CATFISH *PYLODICTIS OLIVARIS* TO FLOODPLAIN CONNECTIVITY IN THE MISSOURI RIVER, NEBRASKA

CONCLUSIONS

Several aspects of flathead catfish diet differed between the flood during 2011 and non-flood during 2012. Flathead catfish captured during 2011 were less likely to have empty stomachs, had more full stomachs, were in better condition, and transitioned to piscivory at a smaller size than fish captured during 2012. Flathead catfish were predominantly invertivorous until they reached 350 mm in 2012, but were predominantly piscivorous at 200 mm in 2011. The bioenergetic benefits associated with all of these factors could lead to influence somatic growth, survival, and reproductive potential, making additional studies of flow-induced changes in Missouri River fish trophic ecology potentially insightful to their management and recovery.

MANAGEMENT CONSIDERATIONS

3.1 Investigate the trophic response of other native fishes to hydrological conditions.

The results of this study suggest that the trophic ecology of flathead catfish differed between two distinct hydrological conditions. Further work to replicate this study during future high and low water events will increase our understanding of the effects of flood pulses on trophic interactions within the Missouri River food web. Additionally, intensive sampling of invertebrates and prey fishes concurrently with diet studies will allow scientists to infer prey selection and how it changes with hydrological conditions. The literature has long suggested that flow conditions influence primary and secondary production in large rivers (e.g., Junk et al. 1989; Power et al. 2008), but further studies to investigate the interaction between flow and production at higher trophic levels are warranted. Specifically, assessing whether floodplain connectivity influences condition, stomach fullness, and the size at which other species make dietary transitions to consuming higher-energy prey could provide insight to their bioenergetics and further aid recovery efforts for imperiled species.

3.2 Assess year-specific fish growth following extreme discharge events.

Jones and Noltie (2007) assessed flathead catfish growth in the upper Mississippi River following the 1993 flood, and found increased yearly growth increments compared to non-flood years. An identical study could be carried out using pectoral spines from flathead catfish and channel catfish in the Missouri River, Nebraska to see if age-specific growth increments were larger during 2010 and 2011 than previous and subsequent years. Together with the year class strength assessment and additional diet studies suggested above, this year-specific growth study would add to a building body of evidence

supporting the theory that even intermittent floodplain connectivity enhances these aspects of large river fish ecology.

CHAPTER 4
AGE-0 CHANNEL CATFISH *ICTALURUS PUNCTATUS* GROWTH RELATED TO
ENVIRONMENTAL CONDITIONS IN THE CHANNELIZED MISSOURI RIVER,
NEBRASKA

CONCLUSIONS

Juvenile catfish growth is variable across years, and likely influenced by a number of interacting conditions related to the environment. Among the most notable observations in my study was the steadily increasing trend in channel catfish growth between about 1995 and 2013. The best model explaining channel catfish growth rate incorporated growing season duration and low discharge duration, and indicated that years with a shorter growing season and more days with discharge below the 20th percentile of a 30 year mean daily discharge results in increased growth rate for age-0 channel catfish. However, variability in environmental conditions during our study were not representative of pre-alteration conditions, and components of the flow regime related to peaks in discharge may affect first-year growth if properly-timed flood pulses were incorporated into a future, managed flow regime. Future work to validate our best-fitting model could be useful in making future water management decisions to aid fish and wildlife recovery.

MANAGEMENT CONSIDERATIONS

4.1 Validate juvenile channel catfish growth model

The results of this chapter suggest that growing season and duration of low discharge influences first year growth of channel catfish. Estimating age-0 channel

catfish growth rate in future years, along with environmental data collection, will allow us to determine the applicability of this model to aid in resource management.

4.2 Evaluate year class strength of native and non-native Missouri River fishes in relation to hydrological conditions.

Past works suggest that some native fishes produce larger year classes in years with floodplain connectivity and unmodified flow and habitat conditions (Jurajda 1995; Balcombe and Arthington 2008). Specifically, King et al. (2009) found increased spawning activity of two native fishes during a flood year in the Murray River, Australia compared to previous low water years, and increased recruitment of two other native species. Steffensen et al. (*in review*) documented higher abundance of some native Missouri River fishes during floods in 2010 and 2011. Additionally, age-0 fish dominated the catch in floodplain habitats both years, suggesting that floodplains provide important rearing habitats for juvenile fishes. Future work to assess flood and drought year production of native species and track their recruitment to the adult population would provide insight to the specific influences of habitat and hydrological conditions on fish production.

4.3 Determine the magnitude, timing, and frequency of flooding needed to elicit changes in fish growth and recruitment.

Managed flooding is a potentially useful tool in the recovery of regulated rivers. The 2011 Missouri River flood was of greater magnitude and duration than most floods prior to flow regulation. It is not reasonable for future managed flood pulses for fish and wildlife benefit to be as large or long-lasting as the 2011 flood, but the ecological benefits of flooding can likely be received during smaller, shorter floods. Stevens et al. (2001) and

Valdez et al. (2001) describe the influence of a seven day regulated flood pulse on the instream habitat and fishes in the Colorado River, Arizona. The Colorado River flood was effective at reconstructing sandbar habitats throughout the river and limiting recruitment of non-native perennial plants (Stevens et al. 2001). The abbreviated flood pulse had no measureable effect on the abundance of native fishes, but caused short-term (~eight month) decreases in non-native rainbow trout *Oncorhynchus mykiss* and fathead minnow *Pimephales promelas* (Valdez et al. 2001). The regulated Colorado River flood pulse was smaller and shorter than historic flood pulses pre-impoundment, and the results of these studies suggest that larger and longer-lasting floods may be required to achieve long-term reduction in undesirable fish species. Similar regulated flood pulses could help scientists determine the magnitude, timing, and frequency of flood pulses needed to benefit Missouri River flora and fauna.

GENERAL CONSIDERATIONS FOR MISSOURI RIVER RECOVERY

1. *Restore components of the Missouri River's natural flow regime.*

Flow in the Missouri River upstream of the Platte River confluence has been particularly homogenized by regulated discharges from Gavins Point Dam. Restoring seasonal variability in flows to resemble pre-alteration conditions, including periods of regular flooding, has been identified as a critical component to ecological recovery in regulated rivers (Galat and Lipkin 2000; Bunn and Arthington 2002), and should be considered an important component of Missouri River recovery efforts. Hesse and Mestl (1993) characterized the natural Missouri River flow regime using stream gage data collected between 1880 and 1899 at Omaha, Nebraska, and noted that the pre-alteration

hydrograph included a distinct peak in March when plains snowpack melted, followed by a larger rise in June when mountain snowpack entered the system. Efforts to restore spring flow pulses have already been met with societal pressure to compare the value of ecosystem services provided by flooding to the economic value of human infrastructure on the floodplain (Jacobson and Galat 2005). Accurately assigning monetary value to ecosystem services is a difficult, if not impossible, task; however, the linkage between flow regime and the integrity of biological communities has been clearly demonstrated (Poff et al. 1997; Marchetti and Moyle 2001; Stromberg 2001).

2. *Continue levee setback efforts to facilitate floodplain connectivity and erosive processes.*

Moving levees farther from the river banks in select locations and restoring areas within the levees to less-altered habitats should provide several benefits to native species and enhance access for outdoor recreation along the river. Constructing levees farther from the river requires more initial expense than repairing existing levees, but reduced maintenance costs make them a desirable long-term option (U.S. Army Corps of Engineers 2012). Substantial legal challenges, including the possible removal of navigation as an authorized purpose for managing the Missouri River, will need to be addressed before the establishment of an erodible corridor can take place. Similar projects to create erodible corridors adjacent to rivers have been successful in reducing flood damages and restoring ecosystem services in several rivers in the United States and abroad (Peigay et al. 2005), and have allowed rivers to regain portions of their structure and function that had been limited previously. Allowing restored flow pulses on the Missouri River to inundate floodplains within setback levees and erode floodplain

sediments will help reinstate the river's natural hydrogeomorphic processes and create complex habitats for fish and wildlife.

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Appendix 1

Fish community composition and structure data for the Missouri River, Nebraska during
the 2011 flood

Table A1-1. Species collected across all five floodplain sites on the Missouri River, Nebraska during 2011 and the percentage contribution of each species to the total collection.

Common name	Scientific name	Count	Percent
Common carp	<i>Cyprinus carpio</i>	313	17.09
Black bullhead	<i>Ameiurus melas</i>	295	16.11
Gizzard shad	<i>Dorosoma cepedianum</i>	200	10.92
Bigmouth buffalo	<i>Ictiobus cyprinellus</i>	185	10.1
Flathead catfish	<i>Pylodictis olivaris</i>	118	6.44
Blue catfish	<i>Ictalurus furcatus</i>	115	6.28
Channel catfish	<i>Ictalurus punctatus</i>	61	3.33
River carpsucker	<i>Carpionodes carpio</i>	57	3.11
Largemouth bass	<i>Micropterus salmoides</i>	53	2.89
White perch	<i>Morone americana</i>	45	2.46
Quillback	<i>Carpionodes cyprinus</i>	38	2.08
Shortnose gar	<i>Lepisosteus platostomus</i>	35	1.91
Smallmouth buffalo	<i>Ictiobus bubalus</i>	27	1.47
Goldeye	<i>Hiodon alosoides</i>	21	1.15
Freshwater drum	<i>Aplodinotus grunniens</i>	21	1.15
Blue sucker	<i>Cycleptus elongatus</i>	20	1.09
Shorthead redhorse	<i>Moxostoma macrolepidotum</i>	20	1.09
Bluegill	<i>Lepomis macrochirus</i>	20	1.09
Red shiner	<i>Cyprinella lutrensis</i>	17	0.93
White crappie	<i>Pomoxis annularis</i>	17	0.93
Silver carp	<i>Hypophthalmichthys molitrix</i>	15	0.82
Green sunfish	<i>Lepomis cyanellus</i>	15	0.82
Shovelnose sturgeon	<i>Scaphirhynchus platyrhynchus</i>	14	0.76
Longnose gar	<i>Lepisosteus osseus</i>	14	0.76
Grass carp	<i>Ctenopharyngodon idella</i>	10	0.55
Emerald shiner	<i>Notropis atherinoides</i>	9	0.49
Black crappie	<i>Pomoxis nigromaculatus</i>	8	0.44
Paddlefish	<i>Polyodon spathula</i>	7	0.38
Yellow perch	<i>Perca flavescens</i>	7	0.38
White bass	<i>Morone chrysops</i>	6	0.33
Fathead minnow	<i>Pimephales promelas</i>	5	0.27
Bighead carp	<i>Hypophthalmichthys nobilis</i>	5	0.27
Grass pickerel	<i>Esox americanus vermiculatus</i>	5	0.27
Sauger	<i>Sander canadensis</i>	5	0.27
Spotfin shiner	<i>Cyprinella spiloptera</i>	4	0.22
Goldfish	<i>Carassius auratus</i>	4	0.22
Brook silverside	<i>Labidesthes sicculus</i>	4	0.22
Yellow bass	<i>Morone mississippiensis</i>	4	0.22
Walleye	<i>Sander vitreum</i>	3	0.16
Suckermouth minnow	<i>Phenacobius mirabilis</i>	2	0.11
Yellow bullhead	<i>Ameiurus natalis</i>	2	0.11
Orangespotted sunfish	<i>Lepomis humilis</i>	2	0.11
River shiner	<i>Notropis blennioides</i>	1	0.05
Sand shiner	<i>Notropis stramineus</i>	1	0.05
Highfin carpsucker	<i>Carpionodes velifer</i>	1	0.05

Table A1-2. Species collected during 2011 Missouri River floodplain electrofishing at Ponca State Park, Nebraska

Common name	Scientific name	Count
River carpsucker	<i>Carpionodes carpio</i>	30
Gizzard shad	<i>Dorosoma cepedianum</i>	18
Quillback	<i>Carpionodes cyprinus</i>	17
Shorthead redhorse	<i>Moxostoma macrolepidotum</i>	15
Shovelnose sturgeon	<i>Scaphirhynchus platyrhynchus</i>	11
Blue sucker	<i>Cycleptus elongatus</i>	10
Channel catfish	<i>Ictalurus punctatus</i>	9
Freshwater drum	<i>Aplodinotus grunniens</i>	9
Emerald shiner	<i>Notropis atherinoides</i>	7
Smallmouth buffalo	<i>Ictiobus bubalus</i>	6
Bigmouth buffalo	<i>Ictiobus cyprinellus</i>	6
Common carp	<i>Cyprinus carpio</i>	5
Flathead catfish	<i>Pylodictis olivaris</i>	4
Bluegill	<i>Lepomis macrochirus</i>	4
Largemouth bass	<i>Micropterus salmoides</i>	4
Shortnose gar	<i>Lepisosteus platostomus</i>	3
Goldeye	<i>Hiodon alosoides</i>	3
Red shiner	<i>Cyprinella lutrensis</i>	3
Spotfin shiner	<i>Cyprinella spiloptera</i>	3
Green sunfish	<i>Lepomis cyanellus</i>	3
Sauger	<i>Sander canadensis</i>	3
Longnose gar	<i>Lepisosteus osseus</i>	2
Black crappie	<i>Pomoxis nigromaculatus</i>	2
Yellow perch	<i>Perca flavescens</i>	2
Paddlefish	<i>Polyodon spathula</i>	1
Grass carp	<i>Ctenopharyngodon idella</i>	1
Highfin sucker	<i>Carpionodes velifer</i>	1
Black bullhead	<i>Ameiurus melas</i>	1
Yellow bullhead	<i>Ameiurus natalis</i>	1

Table A1-3. Species collected during 2011 Missouri River floodplain electrofishing at Tieville Bend, Iowa

Common name	Scientific name	Count
Black bullhead	<i>Ameiurus melas</i>	279
Common carp	<i>Cyprinus carpio</i>	108
Bigmouth buffalo	<i>Ictiobus cyprinellus</i>	70
Gizzard shad	<i>Dorosoma cepedianum</i>	47
White perch	<i>Morone americana</i>	34
Quillback	<i>Carpionodes cyprinus</i>	12
Flathead catfish	<i>Pylodictis olivaris</i>	10
Largemouth bass	<i>Micropterus salmoides</i>	8
Silver carp	<i>Hypophthalmichthys molitrix</i>	5
River carpsucker	<i>Carpionodes carpio</i>	5
Red shiner	<i>Cyprinella lutrensis</i>	4
Fathead minnow	<i>Pimephales promelas</i>	4
Shorthead redhorse	<i>Moxostoma macrolepidotum</i>	4
White bass	<i>Morone chrysops</i>	4
Green sunfish	<i>Lepomis cyanellus</i>	4
Yellow perch	<i>Perca flavescens</i>	4
Bluegill	<i>Lepomis macrochirus</i>	3
Shortnose gar	<i>Lepisosteus platostomus</i>	2
Channel catfish	<i>Ictalurus punctatus</i>	2
Grass pickerel	<i>Esox americanus vermiculatus</i>	2
Freshwater drum	<i>Aplodinotus grunniens</i>	2
Goldeye	<i>Hiodon alosoides</i>	1
Sand shiner	<i>Notropis stramineus</i>	1
Grass carp	<i>Ctenopharyngodon idella</i>	1
Bighead carp	<i>Hypophthalmichthys nobilis</i>	1
Blue sucker	<i>Cycleptus elongatus</i>	1
Smallmouth buffalo	<i>Ictiobus bubalus</i>	1
Black crappie	<i>Pomoxis nigromaculatus</i>	1
Walleye	<i>Sander vitreum</i>	1

Table A1-4. Species collected during 2011 Missouri River floodplain electrofishing at Boyer Chute National Wildlife Refuge, Nebraska

Common name	Scientific name	Count
Bigmouth buffalo	<i>Ictiobus cyprinellus</i>	70
Common carp	<i>Cyprinus carpio</i>	58
Gizzard shad	<i>Dorosoma cepedianum</i>	49
Largemouth bass	<i>Micropterus salmoides</i>	22
Black bullhead	<i>Ameiurus melas</i>	13
Shortnose gar	<i>Lepisosteus platostomus</i>	11
Flathead catfish	<i>Pylodictis olivaris</i>	7
Green sunfish	<i>Lepomis cyanellus</i>	7
Quillback	<i>Carpionodes cyprinus</i>	6
Channel catfish	<i>Ictalurus punctatus</i>	6
Longnose gar	<i>Lepisosteus osseus</i>	4
River carpsucker	<i>Carpionodes carpio</i>	4
Silver carp	<i>Hypophthalmichthys molitrix</i>	3
Freshwater drum	<i>Aplodinotus grunniens</i>	3
Shovelnose sturgeon	<i>Scaphirhynchus platyrhynchus</i>	2
Paddlefish	<i>Polyodon spathula</i>	2
Red shiner	<i>Cyprinella lutrensis</i>	2
Blue sucker	<i>Cycleptus elongatus</i>	2
Orangespotted sunfish	<i>Lepomis humilis</i>	2
Bluegill	<i>Lepomis macrochirus</i>	2
Goldeye	<i>Hiodon alosoides</i>	1
Spotfin shiner	<i>Cyprinella spiloptera</i>	1
Suckermouth minnow	<i>Phenacobius mirabilis</i>	1
Fathead minnow	<i>Pimephales promelas</i>	1
Grass carp	<i>Ctenopharyngodon idella</i>	1
Bighead carp	<i>Hypophthalmichthys nobilis</i>	1
Smallmouth buffalo	<i>Ictiobus bubalus</i>	1
Blue catfish	<i>Ictalurus furcatus</i>	1
Grass pickerel	<i>Esox americanus vermiculatus</i>	1
White perch	<i>Morone americana</i>	1
White crappie	<i>Pomoxis annularis</i>	1
Yellow perch	<i>Perca flavescens</i>	1

Table A1-5. Species collected during 2011 Missouri River floodplain electrofishing at Schilling Wildlife Management Area, Nebraska

Common name	Scientific name	Count
Common carp	<i>Cyprinus carpio</i>	103
Blue catfish	<i>Ictalurus furcatus</i>	96
Gizzard shad	<i>Dorosoma cepedianum</i>	62
Bigmouth buffalo	<i>Ictiobus cyprinellus</i>	34
Channel catfish	<i>Ictalurus punctatus</i>	34
Flathead catfish	<i>Pylodictis olivaris</i>	24
Largemouth bass	<i>Micropterus salmoides</i>	18
Shortnose gar	<i>Lepisosteus platostomus</i>	16
River carpsucker	<i>Carpionodes carpio</i>	16
White crappie	<i>Pomoxis annularis</i>	15
Smallmouth buffalo	<i>Ictiobus bubalus</i>	13
Bluegill	<i>Lepomis macrochirus</i>	11
Longnose gar	<i>Lepisosteus osseus</i>	7
Red shiner	<i>Cyprinella lutrensis</i>	7
Blue sucker	<i>Cycleptus elongatus</i>	6
Freshwater drum	<i>Aplodinotus grunniens</i>	6
Silver carp	<i>Hypophthalmichthys molitrix</i>	5
White perch	<i>Morone americana</i>	5
Black crappie	<i>Pomoxis nigromaculatus</i>	5
Goldfish	<i>Carassius auratus</i>	4
Grass carp	<i>Ctenopharyngodon idella</i>	4
Brook silverside	<i>Labidesthes sicculus</i>	4
Yellow bass	<i>Morone mississippiensis</i>	4
Goldeye	<i>Hiodon alosoides</i>	3
Emerald shiner	<i>Notropis atherinoides</i>	2
Bighead carp	<i>Hypophthalmichthys nobilis</i>	2
Quillback	<i>Carpionodes cyprinus</i>	2
Grass pickerel	<i>Esox americanus vermiculatus</i>	2
White bass	<i>Morone chrysops</i>	2
Paddlefish	<i>Polyodon spathula</i>	1
River shiner	<i>Notropis blennioides</i>	1
Shorthead redhorse	<i>Moxostoma macrolepidotum</i>	1
Yellow bullhead	<i>Ameiurus natalis</i>	1
Green sunfish	<i>Lepomis cyanellus</i>	1
Sauger	<i>Sander canadensis</i>	1

Table A1-6. Species collected during 2011 Missouri River floodplain electrofishing at Indian Cave State Park, Nebraska

Common name	Scientific name	Count
Flathead catfish	<i>Pylodictis olivaris</i>	73
Common carp	<i>Cyprinus carpio</i>	39
Gizzard shad	<i>Dorosoma cepedianum</i>	24
Blue catfish	<i>Ictalurus furcatus</i>	18
Goldeye	<i>Hiodon alosoides</i>	13
Channel catfish	<i>Ictalurus punctatus</i>	10
Smallmouth buffalo	<i>Ictiobus bubalus</i>	6
Bigmouth buffalo	<i>Ictiobus cyprinellus</i>	5
White perch	<i>Morone americana</i>	5
Paddlefish	<i>Polyodon spathula</i>	3
Shortnose gar	<i>Lepisosteus platostomus</i>	3
Grass carp	<i>Ctenopharyngodon idella</i>	3
Silver carp	<i>Hypophthalmichthys molitrix</i>	2
River carpsucker	<i>Carpionodes carpio</i>	2
Black bullhead	<i>Ameiurus melas</i>	2
Sauger	<i>Sander canadensis</i>	2
Shovelnose sturgeon	<i>Scaphirhynchus platyrhynchus</i>	1
Longnose gar	<i>Lepisosteus osseus</i>	1
Red shiner	<i>Cyprinella lutrensis</i>	1
Suckermouth minnow	<i>Phenacobius mirabilis</i>	1
Bighead carp	<i>Hypophthalmichthys nobilis</i>	1
Quillback carpsucker	<i>Carpionodes cyprinus</i>	1
Blue sucker	<i>Cycleptus elongatus</i>	1
Largemouth bass	<i>Micropterus salmoides</i>	1
White crappie	<i>Pomoxis annularis</i>	1
Walleye	<i>Sander vitreum</i>	1
Freshwater drum	<i>Aplodinotus grunniens</i>	1

Appendix 2

Prey items found in flathead catfish stomachs in the Missouri River, Nebraska during
2011 and 2012

Table A2-1. Prey taxa collected from flathead catfish during 2011 and 2012. Flathead catfish collected in 2011 mostly consumed aquatic insects at sizes < 200 mm. Flathead catfish collected during 2012 mostly consumed aquatic insects at sizes < 350 mm. %W = average percent by mass of a prey item across all stomachs within a year/trophic group. %N = average percent by number of prey items across all stomachs within a year/trophic group. %O = percent of stomachs within a year/trophic group containing a given prey item. *N* is the number of individual flathead catfish within a year/trophic group that contained a given prey item.

Prey Taxon	2011								2012							
	<u>Invertivores</u> (<i>N</i> = 15)				<u>Piscivores</u> (<i>N</i> = 107)				<u>Invertivores</u> (<i>N</i> = 159)				<u>Piscivores</u> (<i>N</i> = 62)			
	%W	%N	%O	<i>N</i>	%W	%N	%O	<i>N</i>	%W	%N	%O	<i>N</i>	%W	%N	%O	<i>N</i>
<u>Ephemeroptera</u>																
Isonychiidae	15.9	26.8	40.0	4	1.6	4.7	13.4	15	69.3	68.6	82.2	125	17.9	21.3	23.9	16
Baetidae	11.0	12.5	20.0	2	< 0.1	0.2	0.9	1	2.1	4.1	13.8	21	< 0.1	0.2	1.5	1
Heptageniidae									2.2	3.0	19.7	30	0.1	0.2	3.0	2
Ephemeridae									1.0	0.6	2.0	3	4.4	4.2	6.0	4
Misc. Ephemeroptera					0.8	0.7	0.9	1	< 0.1	0.1	1.3	2				
<u>Trichoptera</u>																
Hydropsychidae	0.2	6.7	20.0	2	2.8	13.3	23.2	26	9.0	11.2	43.4	66	0.7	3.5	9.0	6
<u>Odonata</u>																
Aeshnidae					0.1	0.5	0.9	1								
Coenagrionidae	2.8	1.3	10.0	1												
Libellulidae	2.0	2.5	10.0	1												
Misc. Odonata									< 0.1	0.3	0.7	1				
<u>Diptera</u>																
Chironomidae	1.0	2.5	10.0	1												
Misc. Diptera					0.9	1.1	3.6	4								
<u>Lepidoptera</u>																
Sphingidae					0.6	0.3	0.9	1								
<u>Unidentifiable insect</u>	9.0	6.2	10.0	1	0.1	0.7	1.8	2					1.5	1.5	1.5	1
<u>Megadrilacea</u>	6.0	2.5	10.0	1	0.3	0.4	0.9	1								
<u>Decapoda</u>																
Cambaridae	10.0	10.0	10.0	1	23.6	21.6	31.3	35	5.0	4.6	5.3	8	6.2	6.7	7.5	5

Table A2-1 – Continued.

Prey Taxon	2011								2012							
	<u>Invertivores</u> (N = 15)				<u>Piscivores</u> (N = 107)				<u>Invertivores</u> (N = 159)				<u>Piscivores</u> (N = 62)			
	%W	%N	%O	N	%W	%N	%O	N	%W	%N	%O	N	%W	%N	%O	N
<u>Cypriniformes</u>																
Cyprinidae																
<i>Cyprinus carpio</i>					0.5	0.1	0.9	1								
<u>Siluriformes</u>																
Ictaluridae																
<i>Pylodictus olivaris</i>					2.3	1.3	3.6	4	2.6	2.0	2.6	4	6.2	4.1	7.5	5
<i>Ictalurus</i> spp.					18.4	11.5	23.2	26	2.2	1.0	2.6	4	10.4	10.4	10.5	7
<u>Perciformes</u>																
Centrarchidae																
<i>Micropterus salmoides</i>					1.7	1.7	1.8	2								
Misc. Centrarchidae					0.1	0.2	0.9	1	0.6	0.3	0.7	1				
Percidae																
<i>Perca fulvescens</i>					0.9	0.6	1.8	2					1.5	1.5	1.5	1
<i>Aplodinotus grunniens</i>					6.8	4.9	8.9	10	0.7	0.7	0.7	1	17.3	17.2	17.9	12
<u>Acipenseriformes</u>																
Acipenseridae																
<i>Scaphirhynchus</i> spp.					0.8	0.2	1.8	2								
Polyodontidae																
<i>Polyodon spatula</i>					0.9	0.9	0.9	1								
<u>Unidentifiable fish</u>	42.1	28.0	50.0	5	36.6	33.4	50.0	56	4.4	3.1	6.6	10	31.0	28.4	32.8	22
<u>Anura</u>																
Ranidae																
<i>Rana catesbeiana</i>					0.1	0.1	0.9	1					1.3	0.7	1.5	1
<u>Testudines</u>																
Chelydridae																
<i>Chelydra serpentina</i>													1.5	0.1	1.5	1

