LARVAL FISH ECOLOGY IN RESTORED HABITATS IN THE LOWER MISSOURI RIVER

By

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Abstract: Anthropogenic alterations to aquatic ecosystems have greatly reduced and homogenized riverine habitat, especially those used by larval and juvenile fishes. Creation of shallow-water habitats is used as a restoration technique in response to altered conditions in several studies globally, but only recently on large rivers in the United States. In the summer of 2012, the U.S. Army Corps of Engineers sampled larval and juvenile fishes at six paired sites (mainstem and constructed chute shallow-water habitats) along a section of the Missouri River between Rulo, NE and St Louis, MO. From those samples, we enumerated and identified a total of 7,622 fishes representing 12 families. Community responses of fishes to created shallow-water habitats were assessed by comparisons of species richness and diversity measures between paired sites and among sampling events. Shannon entropy measures were transformed and γ diversity (total diversity) was partitioned into two components, α (within community) and β (between community) diversity using a multiplicative decomposition method. Mantel test results suggest site location, time of sampling event, and habitat type were drivers of larval and juvenile community structure. Paired t-test results indicated little to no differences between in beta diversity between habitat types, however, chute habitats had significantly higher alpha and gamma as well as increased numbers of Asian Carp larvae when compared to mainstem shallow-water habitat. Feeding patterns of Freshwater Drum indicate a greater number of chironomid larvae were consumed in mainstem reaches when compared to chutes. Chute habitats also had significantly greater numbers of individuals with empty stomachs.

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CHAPTER I

COMMUNITY STRUCTURE OF AGE-0 FISHES IN PAIRED MAINSTEM AND CREATED SHALLOW-WATER HABITATS IN THE LOWER MISSOURI RIVER

Introduction

Large rivers around the world, along with their accompanying riparian zones and floodplains, often contain a disproportionately high amount of total regional biodiversity (Ward et al. 2001). However, many large rivers have been significantly altered due to transportation, flood control, irrigation, and hydroelectric power generation. In North America, Europe, and the former Soviet Union, 71% of large rivers are affected by dams, reservoirs, interbasin diversions, and water abstraction (Dynesius and Nilsson 1994). Over 85% of large rivers in the United States alone are fragmented by impoundments that alter hydrologic variability, reduce sediment transport, homogenize in-stream habitat, block dispersal of aquatic organisms, and decrease connections with floodplain habitat (Poff 1997; Luttrell et al. 1999; Tockner and Stanford 2002; Hughes et al. 2005; Nilsson et al 2005). In particular, severing floodplain connections through construction of dams and levees has led to habitat loss for aquatic fauna and altered exchanges of organic and inorganic matter between main-channel and floodplain habitats (Beechie et al. 2001). This loss of habitat alters the dynamic processes occurring at several trophic levels at the floodplain channel interface (Junk et a. 1989).

In response to these degraded conditions, river-floodplain restoration and rehabilitation projects are developed; however, there are few published examples of such projects in Europe and the United States (Schropp and Bakker 1998; Schiemer et al. 1999; Bernhardt et al. 2007). In the United States, reconnection of isolated floodplain habitat techniques are at the early stage of development (Lister and Finnigan 1997; Cowx and Welcomme 1998; Solazzi et al. 2000) while in Europe, studies by Simon et al. (2001) and Chovanec et al. (2002) both concluded that man-made channels reconnecting floodplain habitats in the Rhine and Danube rivers provided suitable habitat for rheophilic invertebrates and fishes. Floodplain restoration is a new and emerging science and results from projects are essential for evaluating and choosing appropriate restoration techniques (Pess et al. 2005). Currently, several restoration programs in the United State have focused on mitigation on the lower Missouri River (Galat et al. 1998) where the river channel has been greatly restricted due to flood-levee construction and bank stabilization (Galat 2005).

The Missouri River has a long history of degraded instream habitat. The 1912 Bank Stabilization and Navigation Project (BSNP), along with the 1945 Rivers and Harbors Act authorized the U.S. Army Corps of Engineers (USACE) to maintain a 300 ft (91 m) wide and 9 ft (3 m) deep navigation channel on the Missouri River from Sioux City, IA to its confluence with the Mississippi River (USACE 1983; Ferrell 1995). These activities have altered the mainstem by flow regulation from upstream reservoirs, channelization, and floodplain levees (Galat et al. 2011) as well as reducing the total surface area of the Missouri River by over 67% (Morris et al. 1968).

Existing alterations to the Missouri River have compromised the integrity of biodiversity in the system. In 2000, the U.S. Fish and Wildlife Service (USFWS) issued a Biological Opinion (BiOp) to the USACE stating that BSNP operating procedures on the Missouri River had jeopardized the continued existence of three federally endangered species: piping plover (*Charadruis melodus*), least tern (*Sternula antilarum*) and pallid sturgeon (*Scaphirhynchus* *albus*). Under Section 7 of the Endangered Species Act (Endangered Species Act,16 United States Code [U.S.C.] 1532 et sequ. 1973), the agency responsible for actions threatening the continued existence of endangered species must take reasonable and prudent measures to minimize negative impacts. The final BiOp issued in 2003 listed several recovery actions for implementation including water management, species propagation/stocking, and restoration of shallow water habitat (SWH) (USFWS 2000, 2003).

Shallow-water habitats (depths < 1.5 m and current velocities < 0.61 m/s) (USFWS 2000, 2003) of the lower Missouri River include constructed chutes, revetment chutes, backwaters, and main channel modifications that mimic shallow floodplain habitats lost through impoundment and channelization. Constructed chutes are of particular interest to USACE and USFWS biologists because they represent an active management strategy to hasten the pace of SWH creation. Chutes are surface-fed side-channels maintaining connection to the mainstem at their upstream and downstream ends, creating an island in between (Pess et al. 2005). Similar habitats have been shown to be critical retention zones for maintaining riverine biodiversity and biological processes (Schiemer et al. 2001). For instance, chute habitats have been shown to provide spawning and rearing habitat for salmonid fishes (Sheng et al. 1990; Lister and Finnigan 1997) and adult rheophilic fish species (Chovanec et al. 2002). Shallow-water habitats usually exhibit higher retention of organic matter and zoobenthos (O'Niell and Thorp 2001), which are the foundation of aquatic food webs. Also, physical characteristics of SWH provide optimal thermal conditions and refuge from water-level fluctuations for larval fishes as well as possible areas of retention for larval fishes as they settle out of the mainstem channel (Schiemer et al. 2001; Schiemer et al. 2003). Constructed chutes are an effort to mimic natural SWH and as these chutes mature, they accrue benefits of SWH for age-0 fishes through the natural dynamic processes of sedimentation, erosion, forestation, and growth of vegetation (Schropp and Bakker 1998). Thus, SWH is vital for fish communities in large rivers and constructed chutes are mechanisms to restore this lost ecosystem function. Comparisons of age-0 fish species assemblages between restored and unrestored reaches of riverine habitats have been done in other settings with promising results (Langler and Smith 2001).

In the lower Missouri River, several fish species have been identified as in critical need of SWH. However, non-native species may also benefit from constructed habitat. Thus, there is a need to evaluate how these habitats affect the total fish community, especially in terms of early-life history phases. We examined measures of assemblage structure such as species richness and effective number of species to assess larval fish community structure within created chute SWH in relation to their adjacent mainstem SWH reaches. We also examined spatiotemporal factors that could be influencing assemblage structure within and between habitat types.

Methods

Sample Collection.- Six sites along the lower Missouri River in the state of Missouri, from Holt County (rkm 807) to St. Louis (rkm 17), were sampled for age-0 fishes by the USACE in summer 2012 (Figure 1). Sites consisted of paired off-channel created chute SWH as well as adjacent mainstem SWH. Two chute habitats were formed naturally (Lisbon and Littles) whereas the remaining four (Rush Bottoms, Worthwine, Jameson and Overton) were constructed, although all were fitted with control structures at the upstream and downstream ends making them “constructed” to varying degrees.

Each paired chute and mainstem SWH were sampled for fishes bimonthly from May to July (Littles site was only sampled once in May) by the USACE using dual, bow-mounted ichthyoplankton nets (750µm mesh) fitted with General Oceanics (General Oceanic Inc., Miami, FL, USA) model 2030R mechanical flow meters. Nets were deployed fixed in the current or gently pushed through areas with little to no velocity. Multiple subsamples were taken until a minimum volume of $500m^3$ was sampled. Specimens were preserved in 70% ethanol, and identified to at least family (genus or species when possible), using larval fish keys developed by Auer (1982) and Holland-Bartels et al. (1990). Due to the difficulty associated with larval fish identification, a “species” in some instances was represented by genus (e.g. *Hiodon* spp.) or family (e.g. Catostomidae) (Wolf et al. 1996; Pepin and Penney 2000). Genetic samples of all *Scaphirhynchus* spp. caught were sent to USFWS to identify to species. Temperature and dissolved oxygen were measured at each site each sample day using a Hydrolab (Hach Hydromet, Loveland, CO, USA) sonde. Current velocity was measured with a torpedo-mounted SonTek (SonTek YSI Inc., San Diego, CA, USA) acoustic doppler velocimeter in the vicinity of the collection.

Data Analysis.- To determine differences between mainstem and chute SWH, we used paired analyses (t-tests when data met assumptions of normality and Wilcoxon signed-rank tests when they did not). All analyses were performed with the “vegan” package (Oksanen et al. 2013) in Program R 3.0.2 (R Core Team 2013) and results were considered significant at $\alpha < 0.05$. We tested for differences in abiotic factors such as depth temperature, dissolved oxygen, velocity densities of age-0 fishes (n/m^3), and fish diversity between habitat types.

Fish diversity was described with two separate metrics, species richness (S) and Shannon entropy (expressed as effective number of species [ENS]). Definitions of diversity using conventional indices like Shannon entropy fail to yield an ecologically meaningful interpretation without further mathematical adjustments (Jost 2006). For example, if the diversity of two communities, one with five equally abundant species and the other with ten, is measured using Shannon entropy the resulting values would be 1.609 and 2.302. These values lack units and are highly non-linear. More importantly, the community with ten equally abundant species is functionally twice as diverse as the community with five (Hill 1973), but this is not the result interpreted from Shannon entropy measures (i.e. $2.302 \neq 1.609 \times 2$). To resolve these shortcomings, Jost (2006) proposed the 'numbers equivalents' of traditional diversity indices. This is done by taking the exponential of Shannon entropy measures ($\exp - \sum_{i=1}^{S} p_i \ln p_i$), where S is the number of species and p is the proportional abundance of species i, producing a measure of diversity that is linear and has units (ENS). This measure of diversity can be interpreted as the number of distinct communities in a group (Jost 2006) and can be directly compared between habitat types (Kanno et al. 2012).

Measures of diversity based on both S and ENS, were partitioned into alpha (α), beta (β), and gamma (γ) components using a multiplicative decomposition method (Jost 2007). All components were examined separately for spatiotemporal patterns between habitat types (Silvano et al. 2000). For example, temporal alpha diversity was measured at one site per sampling event, whereas spatial alpha diversity was measured for a sampling event among sites. Temporal gamma diversity was measured as total diversity found at a site across all sampling events, whereas spatial gamma diversity was measured as a sampling event across all sites. Beta diversity was thus defined as the turnover in species at a site across all sampling events (temporal) as well as turnover in species within a sampling period across sites (spatial). For each dimension (temporal and spatial), beta was estimated with the equation $\beta = \gamma/\alpha$ (Jost 2007; Kanno et al. 2012). Using both S and ENS, we calculated mean α values for each site and event along with the estimated β and γ values and tested for differences between SWH types using pairwise tests as previously described (paired t-tests and Wilcoxon sign-rank tests). Individuals that could not be identified to species level such as *Hiodon* spp. and *Hypopthalmichthys* spp. were treated as a "species" for all diversity partitioning.

Relationships between habitat assemblages, and dimension (time and space) were assessed with Mantel tests. For the temporal dimension, a Bray-Curtis matrix (Bray and Curtis 1957) was calculated using the relative proportion of each species per sampling event and then compared to the Euclidean distance of time between sampling events, whereas the spatial dimension used the Euclidean distance of rkm between sites. Mantel tests (1000 permutations each) were run for all sites combined, then separately for chutes and mainstem sites only.

Results

Despite no significant difference in abiotic factors (Table 1), we found that age-0 fish communities differed between habitat types and as a function of longitudinal location in the river. We identified 7622 individuals from six paired habitat sites (Table 2). Although some individuals could only be identified to family or genus, we treated them all as "species" for analysis (e.g.. *Hiodon* spp.). Genetic results of sturgeon samples indicated all sturgeon caught were shovelnose sturgeon (USACE, unpublished data). Across all six sites, thirteen species were caught in chute habitats and twelve in mainstem areas. Fishes caught in chute habitat comprised 73% of total catch; however 67% of these chute-dwelling individuals were collected in just two samples. These samples were not representative of fish density in this habitat type and were removed from subsequent analyses. Age-0 fish densities were, on average, more than two times greater in chute habitats ($0.77/m^3$) than mainstem sites ($0.3/m^3$), but this was not significantly different (Paired t (31) = -1.53, P = 0.14). Asian carp (*Hypopthalmichthys* spp.), suckers (Catostomidae) freshwater drum (*Aplodinotus grunniens*) and minnows (Cyprinidae) made up 34%, 34%, 13%, and 13% of total chute density, respectively. In comparison, freshwater drum, suckers, Asian carp, and minnows made up 38%, 26%, 15%, and 14% of total mainstem density, respectively.

Temporal differences in diversity.- Differences in fish diversity between habitat types varied according to the method (richness or expected number of species) and metric (α, β, γ). When considered temporally (at a site over time), alpha diversity based on species richness (S_α) was higher in chute habitats (3.52) than mainstem (2.72) (Wilcoxon signed-rank: V= 312, P =

0.02) (Table 3). However, when using ENS, which is sensitive to abundances, we found no significant differences in alpha diversity between chute and mainstem habitats (Paired-t(34) = 0.93, P = 0.36). Beta diversity was not significant for either measure (S_{β}, Paired-t(5) = 0.79, P = 0.47) (ENS, Paired-t(5) = 0.04, P = 0.97), with approximately 2 distinct fish communities at each site over time. Gamma diversity measures were significantly higher in chute habitat when based on richness (S_{γ} Paired-t(5) =3.10, P = 0.03) but not when quantified with ENS (Paired-t(5)=1.13, P = 0.31).

Mantel test results for all sampling events from both habitats pooled showed no significant relationship (Mantel: r = 0.19, P = 0.23) (Figure 2). However, when habitats were separated, chutes (Mantel: r = 0.84, and P < 0.01) and mainstem habitats (Mantel: r= 0.75, and P < 0.01) were significantly autocorrelated, being more dissimilar with increasing time between sampling events.

Spatial differences in diversity.- When considered spatially, alpha diversity based on species richness (S_{α}) was significantly higher in chute habitats than mainstem (3.31 versus 2.57) (Wilcoxon signed-rank: V= 312, P = 0.01) (Table 4). Using the ENS method, we found no significant differences in alpha diversity between chute and mainstem habitats (Wilcoxon signed-rank: V=278, P = 0.19). No significant differences were found between beta diversities regardless of method (S_{β}, Paired-t: t = -0.01, df = 5, P = 0.99) (ENS, Wilcoxon signed-rank: V=10 P = 1.00), with less than 2 distinct communities over sites within any one sampling event. Similarly, no significant differences were found between gamma diversities regardless of method (S_{γ}, Paired-t(5) =1.95, P = 0.11) (ENS, Paired-t(5) = 0.68, P = 0.53).

Mantel test results for all sites of both habitats pooled showed no significant relationship (Mantel: r = 0.08, P = 0.23) (Figure 3). However, when habitats were separated, chutes (Mantel:

$r = 0.75$, and $P < 0.01$) and mainstem habitats (Mantel: $r= 0.75$, and $P < 0.01$) were significantly autocorrelated, being more dissimilar with increasing distance between sites.

Discussion

Greater alpha richness (temporally and spatially) as well as greater gamma diversity (temporally) of chute SWH provide evidence of the importance of this habitat type for age-0 fishes; however, with the caveat that these conclusions are drawn from one year of data. Whether habitat quality or quantity is the primary determinant is unknown. On one hand, species richness generally increases simply as a function of area as described by island biogeography theory (MacArthur and Wilson 1963). If constructed chute habitats provided greater SWH per area than its mainstem counterpart, we would expect to find a greater number of species on this basis alone (Angermeier and Schlosser 1989). Alternatively, habitat quality in terms of prey availability or heterogeneity, for example, may also play a role even though the abiotic parameters did not vary between habitat types sampled (O'Neill and Thorp 2001; Simons et al. 2001). Whether these juvenile fishes are receiving less predation pressure, increased foraging return, or other fitness advantages in these constructed chute habitats is unknown and further study is needed. Compared to slack, backwater habitats, constructed chutes appear to offer less forage for juvenile fishes, in terms of phytoplankton and zooplankton, but few comparisons have been made between chute and mainstem sites (Dzialowski et al. 2013). Additional research related to prey availability, diet and body growth is planned for the future and this could be help elucidate these mechanisms.

One of the main "benefits" of this SWH appeared to be for invasive Asian carp, whose larvae composed 30% of age-0 density in chute SWH compared to only about 5% in mainstem habitats. However, approximately 99% of Asian carp larvae collected in chutes were caught at one chute alone (Jameson). Adult Asian carp species often frequent shallow, slow moving waters in larger rivers during the summer (DeGrandchamp et al. 2008) and many were observed leaping out of slow moving waters in chute habitat throughout this study (A. Civiello, personal observation). Spawning of Asian carp species occurs in spring and summer and peak larval densities have been linked to rising discharge and water temperatures (Lohmeyer and Garvey 2009). Post-spawn, eggs drift for approximately 100 kilometers (Gorbach and Krykhtin 1980) and then settle out in flooded backwater areas that act as nursery habitat (Schrank et al. 2001). It is plausible that created SWH is acting as suitable nursery habitat and thus enhancing recruitment of Asian carp species, but possibly dependent on location along the river's length. Similarly, some pelagic spawning cyprinids with reproductive ecology similar to Asian carp species may also be benefitting from these habitats, however; difficulties identifying cyprinid specimens below the family level may have prevented detection of larvae of these species (Hoagstrom and Turner 2013). This, as well as other possible benefits to invasive species, should receive further study and be considered in future restoration projects.

Our results could have been biased because of exclusion of two outlier samples from Jameson chute, which contained over 67% of total fish captured. However, the species in these samples were representative of species captured in other samples (e.g., freshwater drum, suckers, minnows, *Hiodon* spp., sunfishes, and Asian carp), resulting in no change to richness values. Including these results in ENS calculations could have skewed results higher for chute habitats, where we found no significant differences when they were excluded. Our conservative conclusions without these outliers suggest higher diversity in chutes, so including them would only strengthen that conclusion.

Mantel tests suggest strong spatial and temporal autocorrelation; but dependent on habitat type, which could have several implications for future restoration projects. First, unidentified mechanisms related to habitat type (chute or mainstem SWH) appear to be providing different advantages to age-0 fishes. For instance, chute habitats exhibited a much stronger spatial autocorrelation than mainstem SWH. Community dissimilarity approached 0.6 at the most physically separated sites and peaked at sites in the middle section of the river (Lisbon, Jameson and Overton). In comparison, dissimilarity observed at mainstem sites was about one-half of chute sites, approaching 0.3 at the most physically separated sites. In essence, chute SWH are operating more like isolated islands of habitat than mainstem sites. Dissimilarities at chute sites were due to increased densities of freshwater drum in downstream sites along with decreased densities of shortnose gar, *Hiodon* spp., and sucker species. In contrast, the smaller values of dissimilarity observed in mainstem habitats were due to increases in freshwater drum, shortnose gar, and sunfishes in downstream sites as well as a reduction in temperate basses and suckers. While dissimilarity and date of sampling event were significantly correlated in both habitat types, the magnitude of change in temporal autocorrelation was about the same for both habitats. As a result, restoration efforts targeted for a particular species, either threatened or endangered (e.g. pallid sturgeon) or invasive (e.g. Asian carp) should take river location into consideration. For example, restoring reaches near an impoundment may not benefit large river species as much as restoring lower reaches because of the large influx of non-native species from the reservoir (Penczak and Kruk 2004). The impoundment may be altering mechanisms that structure fish populations along the river continuum in a manner described by the serial discontinuity concept (Ward and Stanford 1983). Longitudinal recovery from these alterations may make restoration efforts at downstream sites more favorable to early life-stages of riverine fishes (Kinsolving and Bain 1996). In this study, shovelnose sturgeon were caught exclusively at three sites (Lisbon, Jameson, and Overton) located approximately midway down the river, at least 954 rkm from Gavins Point Dam. Using these results, managers could focus restoration efforts for these species at sites farther from the dam, however; it should be noted the gears used in this study were not as well suited for targeting benthic larval fishes as others, such as mini-Missouri trawls. Also, management and eradication efforts of Asian carp could be focused at sites around Jameson chute where a majority of this species was collected. Lastly, distance from treatment sites may be an important factor to consider when selecting the location of reference sites for future studies. Control and reference sites help account for a portion of natural background variation that masks detection of a true response to restoration (Roni et al. 2005). If control or reference sites are located farther away from treatment sites and spatial autocorrelation exists, detection of responses to restoration could be obscured (Tobler 1970).

In conclusion, chute SWH generated varied results for age-0 fish community structure when compared to adjacent reaches of mainstem SWH. The significance of species richness values indicates that chute SWH provides larger or higher quality habitat types to early life stages of fishes. But this benefit is "species"-specific and benefits a subset of the assemblage. Difficulty with taxonomic resolution of specimens could have masked or obscured other patterns. For instance, it is possible chute SWH are benefitting cypinid and castostomid species that were found in higher numbers in chutes. These particular groups have shown long-term decline in the Missouri River (Pflieger and Grace 1987) and higher taxonomic resolution would allow specific responses of these groups to be quantified. Location of restoration segments may have many implications for management of restoration projects, future construction of restoration reaches, as well as future studies investigating floodplain restoration effectiveness.

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TABLE 1. Mean (SD) values of abiotic measurements taken at six sites in two habitat types (chute and mainstem) of the lower Missouri River in May-July 2012 at each sampling event. Mean differences in abiotic variables were tested with paired t-tests.

Site	rkm[1]	Temperature (C°)	Velocity (m/s)	Dissolved Oxygen (mg/L)	Depth (m)
Rush bottoms chute	806	24.1 (4.7)	0.60 (0.21)	7.5 (1.0)	1.2 (0.3)
Rush bottoms main		24.1 (4.7)	0.30 (0.20)	8.0 (0.9)	1.4 (0.4)
Worthwine chute	737	23.9 (5.3)	0.62 (0.11)	6.9 (1.5)	1.8 (0.7)
Worthwine main		24.1 (4.9)	0.53 (0.17)	7.8 (0.8)	1.2 (0.3)
Lisbon chute	351	25.1 (3.4)	0.42 (0.09)	6.8 (1.0)	1.2 (0.4)
Lisbon main		24.8 (3.6)	0.61 (0.14)	7.3 (0.7)	1.2 (0.2)
Jameson chute	344	24.9 (3.4)	0.81 (0.08)	7.4 (1.3)	1.2 (0.4)
Jameson main		24.9 (3.5)	0.49 (0.05)	7.5 (1.5)	1.3 (0.3)
Overton chute	301	26.7 (3.2)	0.47 (0.05)	7.8 (0.9)	1.5 (0.5)
Overton main		26.0 (3.1)	0.58 (0.14)	7.5 (1.3)	1.4 (0.3)
Littles chute	17	27.4 (2.2)	0.25 (0.18)	7.8 (0.1)	1.1 (0.5)
Little main		27.7(2.6)	0.32 (0.26)	8.1 (0.7)	1.0 (0.4)
Mean difference between chute and mainstem		-0.05	0.08	-0.56	0.08
DF		31	32	32	34
t-statistic		-0.79	1.34	-1.79	1.28
P-value		0.43	0.19	0.08	0.08

[1] river kilometers from confluence with Mississippi River

TABLE 2. Total numbers and proportion (percentage of total catch) that each species (in this case genera and families were counted as a species) contributed to total density at each habitat type. *Indicates species was not caught in habitat

Fish Identity		Chute SWH		Mainstem SWH	
Common name	Scientific name	Number caught	Prop. Of Density	Number caught	Prop. Of Density
Asian carp	*Hypopthalmichthys spp.*	1891	33.6	309	14.8
suckers	*Catostomidae*	1847	33.5	549	26.3
minnows	*Cyprinidae*	707	12.8	287	13.8
freshwater drum	*Aplodinotus grunniens*	719	12.7	788	37.8
goldeye and mooneye	*Hiodon spp.*	322	5.8	125	5.9
sunfishes	*Centrarchidae*	17	<1	16	<1
shortnose gar	*Lepisosteus platostomus*	20	<1	3	<1
shovelnose sturgeon	*Scaphirhynchus platorhynchus*	4	<1	1	<1
herrings	*Clupeidae*	3	<1	*	*
temperate basses	*Moronidae*	3	<1	1	<1
yellow bullhead	*Ameiurus natalis*	1	<1	*	
blue catfish	*Ictalurus furcatus*	*	*	1	<1
channel catfish	*Ictalurus punctatus*	1	<1	1	<1
pikes	*Esocidae*	1	<1	*	
paddlefish	*Polyodon spathula*	*	*	4	<1

TABLE 3. Partitioning of diversity for age-0 fish species richness and effective number of species between both habitat types by site over time. Different superscript letters indicate significant differences between chute and mainstem averages for each diversity measure.

	Species richness (S)			Effective number of species (ENS)		
Site	α^1	β	γ	α^1	β	γ
Rush bottoms chute	2.67	2.63	7.00	1.83	1.63	2.98
Rush bottoms main	2.00	2.50	5.00	1.31	2.26	2.97
Worthwine chute	3.67	3.00	11.00	2.19	1.95	4.28
Worthwine main	1.50	3.33	5.00	0.93	1.60	1.49
Lisbon chute	3.50	2.29	8.00	1.89	2.01	3.80
Lisbon main	2.67	2.25	6.00	1.88	1.97	3.72
Jameson chute	4.33	2.54	11.00	2.33	1.43	3.32
Jameson main	3.00	2.67	8.00	1.72	1.97	3.38
Overton chute	3.17	2.84	9.00	1.83	2.24	4.10
Overton main	3.67	1.91	7.00	2.73	1.44	3.92
Littles chute	2.40	2.50	6.00	0.96	1.75	1.67
Little main	2.60	2.31	6.00	0.92	1.71	1.58
Chute average	3.52^{2a}	2.63^{a}	8.67^{a}	1.84^{2a}	1.84^{a}	3.36^{a}
Mainstem average	2.72^{2b}	2.50^{a}	6.17^{b}	1.58^{2a}	1.83^{a}	2.84^{a}

[1] Mean values reported for a site over six sampling periods excluding Littles which was only sampled five times (n = 6, n = 5 for Littles).

[2] Mean values reported for all sites and sampling periods (n = 35).

TABLE 4. Partitioning of diversity for age-0 fish species richness and effective number of species between both habitat types by day of collection over sites. Different superscript letters indicate significant differences between chute and mainstem averages for each diversity measure.

	Species richness (S)			Effective number of species (ENS)		
Sampling Event	α^1	β	γ	α^1	β	γ
I. May chute	3.60	1.94	7.00	1.96	1.63	3.18
I. May main	3.00	2.00	6.00	1.98	1.67	3.30
II. May chute	4.83	1.66	8.00	2.57	1.31	3.38
II. May main	2.67	1.50	4.00	1.64	1.63	2.68
I. June chute	4.50	2.22	10.00	2.57	1.54	3.95
I. June main	4.00	2.25	9.00	2.15	1.66	3.58
II. June chute	3.67	1.64	6.00	2.58	1.23	3.17
II. June main	2.67	2.25	6.00	1.55	1.42	2.20
I. July chute	2.17	1.85	4.00	1.04	1.44	1.50
I. July main	1.83	1.64	3.00	1.57	1.22	1.92
II. July chute	1.17	2.57	3.00	0.46	2.90	1.34
II. July main	1.33	2.25	3.00	0.78	2.33	1.83
Chute average	3.31^{2a}	1.98^{a}	6.33^{a}	1.90^{2a}	1.67^{a}	2.75^{a}
Mainstem average	2.57^{2b}	1.98^{a}	5.17^{a}	1.60^{2a}	1.66^{a}	2.59^{a}

[1] Mean values reported for a sampling event over six sampling sites excluding Littles which was only sampled five times (n = 6, n = 5 for Littles).

[2] Mean values reported for all sites and sampling periods (n = 35).

FIGURE 1. Missouri River extending across the state of Missouri along with sites of paired created-chute and mainstem shallow-water habitats sampled for age-0 fishes in 2012.

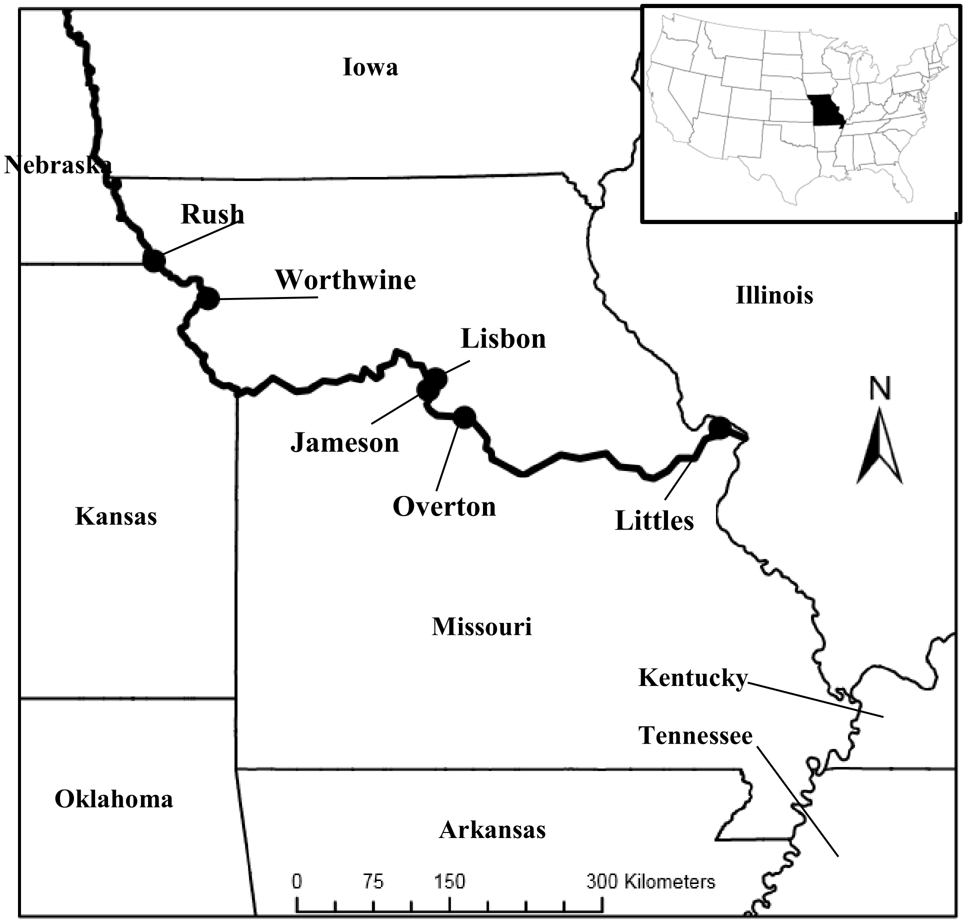


FIGURE 2. Mantel test results comparing Bray-Curtis dissimilarity measures based on age-0 fish community structure at each sampling event and time between events for chute habitat only (r = 0.75, P<0.01) mainstem habitat only (r = 0.75, P<0.01) and both mainstem and chute habitats together (r = 0.19, P=0.09).

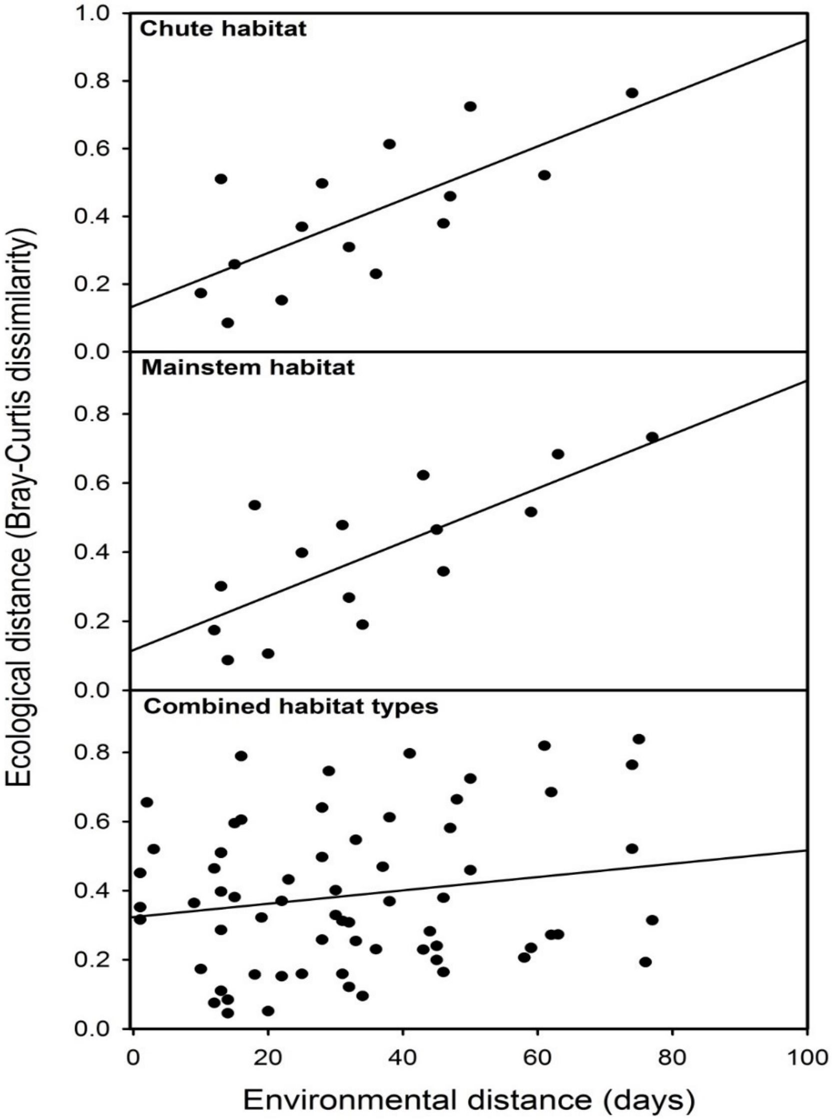
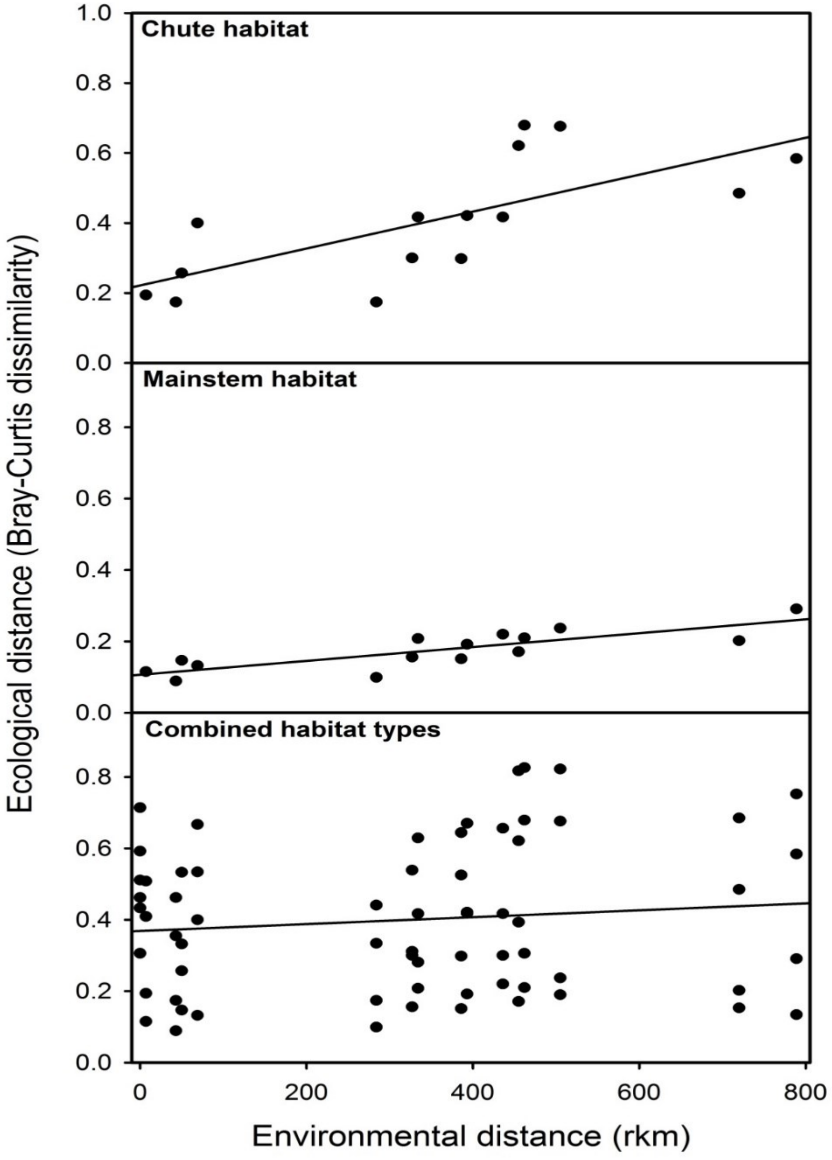


FIGURE 3. Mantel test results comparing Bray-Curtis dissimilarity measures based on age-0 fish community structure at each site and distance between sites for chute habitat only (r = 0.84, P<0.01) mainstem SWH only (r = 0.75, P<0.01) and both mainstem and chute habitats together (, r = 0.08, P = 0.23).



Appendix 1. Larval species identified from USACE sampling on the lower Missouri River in the summer of 2012 (including outlier sample that was removed from analysis).

Date	Species	Abundance
	Jameson Chute	
5/9	*Catostomid* spp.	1324
5/9	*Centrarchid* spp.	2
5/9	*Cyprinid* spp.	464
5/9	*Hiodon* spp.	75
5/9	*Hypopthalmichthys* spp.	1876
5/9	*Macrhybopsis storeriana*	1
5/9	*Pylodictus olivaris*	1
5/9	Unidentified spp.	2789
5/24	*Aplodinotus grunniens*	19
5/24	*Catostomid* spp.	33
5/24	*Centrarchid* spp.	4
5/24	*Clupeid* spp.	1
5/24	*Hiodon* spp.	68
5/24	*Lepisosteus platostomus*	1
6/7	*Aplodinotus grunniens*	1
6/7	*Catostomid* spp.	55
6/7	*Cyprinid* spp.	61
6/7	*Esocidspp.*	1
6/7	*Hiodon* spp.	11
6/7	*Ictalurus punctatus*	1
6/7	*Scaphirhynchus platorynchus*	1
6/7	Unidentified spp.	21
6/20	*Aplodinotus grunniens*	6
6/20	*Catostomid* spp.	58
6/20	*Cyprinid* spp.	13
6/20	Unidentified spp.	11
7/12	*Aplodinotus grunniens*	29
7/12	*Catostomid* spp.	5
7/12	*Centrarchid* spp.	1
7/26	*Aplodinotus grunniens*	31
7/26	*Catostomid* spp.	1
	Jameson Mainstem	
5/9	*Catostomid* spp.	277
5/9	*Centrarchid* spp.	6
5/9	*Cyprinid* spp.	220
5/9	*Hiodon* spp.	18

5/9	*Hypopthalmichthys* spp.	290
5/9	*Polyodon spathula*	1
5/9	Unidentified spp.	123
5/24	*Aplodinotus grunniens*	4
5/24	*Catostomid* spp.	8
5/24	*Centrarchid* spp.	1
5/24	*Hiodon* spp.	59
6/7	*Catostomid* spp.	23
6/7	*Cyprinid* spp.	17
6/7	*Ictalurus punctatus*	1
6/7	*Lepisosteus platostomus*	1
6/7	*Polyodon spathula*	1
6/7	*Scaphirhynchus platorynchus*	3
6/7	Unidentified spp.	9
6/20	*Catostomid* spp.	12
6/20	*Cyprinid* spp.	2
6/20	*Ictalurus punctatus*	1
7/12	*Aplodinotus grunniens*	10
7/12	*Catostomid* spp.	8
7/26	*Aplodinotus grunniens*	25
7/26	*Centrarchid* spp.	1
	Lisbon Chute	
5/8	*Catostomid* spp.	160
5/8	*Cyprinid* spp.	66
5/8	*Cyprinus carpio*	1
5/8	*Hiodon* spp.	42
5/8	*Hypopthalmichthys* spp.	18
5/8	Unidentified spp.	62
5/23	*Aplodinotus grunniens*	1
5/23	*Catostomid* spp.	2
5/23	*Clupeid* spp.	1
5/23	*Hiodon* spp.	105
5/23	*Scaphirhynchus platorynchus*	2
5/23	Unidentified spp.	1
6/6	*Catostomid* spp.	77
6/6	*Cyprinid* spp.	36
6/6	*Hiodon* spp.	7
6/6	*Hypopthalmichthys* spp.	1
6/6	Unidentified spp.	11
6/21	*Aplodinotus grunniens*	7
6/21	*Catostomid* spp.	3

6/21	*Cyprinid* spp.	16
6/21	*Lepisosteus platostomus*	1
7/11	*Aplodinotus grunniens*	16
7/11	*Catostomid* spp.	3
7/11	*Macrhybopsis* spp.	1
7/11	*Moronid* spp.	1
7/25	*Centrarchid* spp.	1
7/25	*Pimephales* spp.	1
	Lisbon Mainstem	
5/8	*Catostomid* spp.	96
5/8	*Hiodon* spp.	3
5/8	*Hypopthalmichthys* spp.	39
5/8	Unidentified spp.	22
5/23	*Catostomid* spp.	2
5/23	*Hiodon* spp.	23
6/6	*Catostomid* spp.	25
6/6	*Cyprinid* spp.	18
6/6	*Ictalurus furcatus*	1
6/6	Unidentified spp.	1
6/21	*Aplodinotus grunniens*	2
6/21	*Catostomid* spp.	15
6/21	*Cyprinid* spp.	1
7/11	*Aplodinotus grunniens*	17
7/11	*Catostomid* spp.	19
7/11	*Cyprinid* spp.	8
7/25	*Aplodinotus grunniens*	12
7/25	*Catostomid* spp.	2
	Littles Chute	
5/31	*Aplodinotus grunniens*	114
5/31	*Catostomid* spp.	19
5/31	*Hiodon* spp.	7
5/31	Unidentified spp.	1
6/14	*Aplodinotus grunniens*	187
6/14	*Catostomid* spp.	2
6/14	*Cyprinid* spp.	4
6/14	*Hiodon* spp.	10
6/14	*Hypopthalmichthys* spp.	1
6/14	*Lepisosteus platostomus*	4
6/27	*Aplodinotus grunniens*	92
6/27	*Catostomid* spp.	5
6/27	*Cyprinid* spp.	5
6/27	*Hiodon* spp.	1

7/17	*Aplodinotus grunniens*	1
7/31	*Catostomid* spp.	1
7/31	*Centrarchid* spp.	1
7/31	*Lepomis macrochirus*	1
	Littles Mainstem	
5/31	*Aplodinotus grunniens*	156
5/31	*Catostomid* spp.	16
5/31	*Centrarchid* spp.	4
5/31	*Hiodon* spp.	1
5/31	Unidentified spp.	1
6/14	*Aplodinotus grunniens*	279
6/14	*Catostomid* spp.	7
6/14	*Cyprinid* spp.	1
6/14	*Hiodon* spp.	9
6/14	*Lepisosteus platostomus*	1
6/27	*Aplodinotus grunniens*	92
6/27	*Catostomid* spp.	8
6/27	*Cyprinid* spp.	2
6/27	*Hiodon* spp.	1
6/27	Unidentified spp.	1
7/17	*Catostomid* spp.	2
7/17	*Cyprinid* spp.	1
7/17	*Macrhybopsis gelida*	2
	Overton Chute	
5/15	*Catostomid* spp.	68
5/15	*Cyprinid* spp.	64
5/15	*Hiodon* spp.	4
5/15	*Moronid* spp.	1
5/15	*Scaphirhynchus platorynchus*	2
5/15	Unidentified spp.	4
5/30	*Aplodinotus grunniens*	9
5/30	*Catostomid* spp.	17
5/30	*Centrarchid* spp.	2
5/30	*Clupeid* spp.	1
5/30	*Cyprinid* spp.	3
5/30	*Hiodon* spp.	20
5/30	*Ictalurus furcatus*	1
5/30	*Lepisosteus platostomus*	1
6/13	*Aplodinotus grunniens*	21
6/13	*Catostomid* spp.	8
6/13	*Hiodon* spp.	4
6/26	*Aplodinotus grunniens*	7

6/26	*Catostomid* spp.	9
7/16	*Aplodinotus grunniens*	4
7/30	*Aplodinotus grunniens*	1
	Overton Mainstem	
5/15	*Catostomid* spp.	35
5/15	*Cyprinid* spp.	37
5/15	*Hiodon* spp.	4
5/15	*Polyodon spathula*	1
5/15	Unidentified spp.	23
5/30	*Aplodinotus grunniens*	4
5/30	*Catostomid* spp.	2
5/30	*Centrarchid* spp.	2
5/30	*Hiodon* spp.	5
5/30	Unidentified spp.	1
6/13	*Aplodinotus grunniens*	5
6/13	*Catostomid* spp.	8
6/13	*Centrarchid* spp.	2
6/13	*Cyprinid* spp.	1
6/13	*Hiodon* spp.	2
6/13	*Lepisosteus platostomus*	1
6/13	*Polyodon spathula*	1
6/13	*Scaphirhynchus platorynchus*	1
6/13	Unidentified spp.	13
6/26	*Aplodinotus grunniens*	11
6/26	*Catostomid* spp.	8
7/30	*Aplodinotus grunniens*	4
7/30	*Catostomid* spp.	2
	Rush Chute	
5/3	Unidentified spp.	2
5/22	*Catostomid* spp.	6
5/22	*Centrarchid* spp.	1
5/22	*Hiodon* spp.	5
5/22	Unidentified spp.	4
6/4	*Catostomid* spp.	1
6/4	*Cyprinid* spp.	1
6/4	*Hiodon* spp.	1
6/4	*Ictalurus punctatus*	1
6/4	Unidentified spp.	1
6/18	*Ameiurus natalis*	1
6/18	*Aplodinotus grunniens*	7
6/18	*Catostomid* spp.	5
6/18	*Cyprinid* spp.	1

6/18	*Hiodon* spp.	1
6/18	*Lepisosteus platostomus*	11
6/18	Unidentified spp.	1
7/9	*Aplodinotus grunniens*	84
7/9	*Catostomid* spp.	5
7/23	*Aplodinotus grunniens*	3
7/23	Unidentified spp.	1
	Rush Mainstem	
5/22	*Catostomid* spp.	4
5/22	*Hiodon* spp.	5
5/22	Unidentified spp.	1
6/4	*Aplodinotus grunniens*	1
6/4	*Catostomid* spp.	14
6/4	*Cyprinid* spp.	3
6/4	*Hiodon* spp.	1
6/18	*Aplodinotus grunniens*	4
6/18	*Catostomid* spp.	6
6/18	*Moronid* spp.	1
7/9	*Aplodinotus grunniens*	18
7/9	*Catostomid* spp.	7
7/9	Unidentified spp.	3
7/23	*Aplodinotus grunniens*	4
7/23	*Notropis stramineus*	1
	Worth Chute	
5/1	*Catostomid* spp.	2
5/1	*Centrarchid* spp.	3
5/1	*Hiodon* spp.	1
5/1	Unidentified spp.	1
5/21	*Aplodinotus grunniens*	1
5/21	*Catostomid* spp.	4
5/21	*Cyprinid* spp.	1
5/21	*Hiodon* spp.	5
5/21	*Ictalurus punctatus*	1
6/5	*Catostomid* spp.	10
6/6	*Centrarchid* spp.	2
6/7	*Cyprinid* spp.	4
6/8	*Hiodon* spp.	2
6/9	Unidentified spp.	1
6/19	*Aplodinotus grunniens*	3
6/19	*Catostomid* spp.	7
6/19	*Cycleptus elongatus*	1
6/19	*Cyprinid* spp.	1

6/19	*Lepisosteus platostomus*	2
7/10	*Aplodinotus grunniens*	83
7/10	*Catostomid* spp.	1
7/10	*Centrarchid* spp.	2
7/10	*Moronid* spp.	1
7/24	*Aplodinotus grunniens*	13
7/24	*Catostomid* spp.	1
	Worth Mainstem	
5/2	*Cyprinid* spp.	1
5/2	*Polyodon spathula*	1
6/5	Unidentified spp.	2
6/19	*Aplodinotus grunniens*	5
6/19	*Catostomid* spp.	2
6/19	*Ictalurus punctatus*	5
7/10	*Aplodinotus grunniens*	133
7/10	*Catostomid* spp.	5
7/24	*Catostomid* spp.	1

CHAPTER II

TROPHIC RESPONSES OF JUVENILE FISHES IN SHALLOW-WATER HABITATS OF THE LOWER MISSOURI RIVER

Introduction

Degradation of freshwater habitats has prompted several large-scale restoration projects aimed at recovering lost ecosystem structure and function (Roni 2005; Berhardt et al. 2007). In large rivers, restoration projects have often been aimed at recovering losses of habitat complexity, biodiversity, and dynamic ecological processes at the river floodplain interface, which represents a species-rich environment that drives productivity and energy exchange at multiple trophic levels (Junk et al. 1989; Ward et al. 1999). However, difficulties with post-monitoring make gauging the relative "success" of restoration problematic (Roni at al. 2005). A multitude of restoration metrics encompassing abiotic and biotic characteristics exist (Pess et al. 2005) and the responses by each may vary considerably.

In the lower Missouri River, few differences in fish community metrics were found between mainstem and constructed chute shallow-water habitat (Chapter 1). There, constructed chutes were hypothesized to have greater species richness and effective number of species metrics than adjacent mainstem habitats due to either greater habitat quality or quantity. Trophic responses by individual fish species may provide another mean to assess the effect of chute construction on the fishes that rely on the natural environments of the river. Diet analysis of fishes collected from differing habitat types is one method for assessing responses to these restoration activities (Jud et al. 2011; Jordan and Arrington 2014). This approach helps identify predator-prey relationships, trophic linkages among adjacent habitats, and other aspects of ecosystem function (Jordan and Arrington 2014).

In the current study, I assessed diet metrics of age-0 Freshwater Drum (*Aplodinotus grunniens*) and Goldeye and Mooneye (*Hiodon* spp.) to determine trophic responses by fishes to creation of chute shallow-water habitats on the lower Missouri River. River. These two species were consistently abundant in both habitat types and most sites in the river, allowing for such an assessment (Chapter 1 [Appendix 1]). Freshwater Drum spawn in open water where eggs float for one to two days before hatching (Daiber 1953). Diet studies of age-0 Freshwater Drum have been limited to lakes and reservoirs, where they have been described as consuming cladocerans, chironomid larvae, mayfly larvae, and occasionally larvae of other fishes (Swedburg and Walburg 1970; Clark and Pearson 1979; Bur 1982). Both Goldeye and Mooneye are thought to spawn midwater where eggs and newly hatched larvae float freely (Battle and Sprules 1960). Age-0 Goldeye were found to consume primarily calanoid copepods and cladocerans in the Peace-Athabasca Delta, Alberta (Donald and Kooyman 1976).

Determinants of feeding patterns could be attributed to habitat quality (e.g. prey availability or habitat heterogeneity [Tews et al. 2004]) or habitat quantity (e.g. species increasing as a function of available habitat [MacArthur and Wilson 1963; MacArthur and Wilson 1967]). The current study aims to address the former, as previous studies have shown the utility of diet analysis to explain differences among habitat types (Jud et al. 2011; Jordan and Arrington 2014). Furthermore, because there is a paucity of information on prey use by these two species at early-life history stages, I use my findings to fill this gap. To quantify potential differences in resources use, I examined prey richness, numbers of empty stomachs, and proportional prey-use by Freshwater Drum and *Hiodon* spp. in chute and mainstem habitats in the lower Missouri River.

Methods

Sample Collection.- Age-0 Freshwater Drum and *Hiodon* spp. were collected by USACE biologists in the summer of 2012 at six sites in the Missouri River spanning the state of Missouri from Holt County (rkm 807) to St. Louis (rkm 17) (Figure 1). Each site consisted of paired mainstem and created chute SWH. Four chute habitats were constructed by the USACE (Rush Bottoms, Worthwine, Jameson and Overton) while two were formed naturally (Lisbon and Littles). However, all chutes were fitted with control structures at the upstream and downstream ends to maintain connection with the mainstem, which makes them all "constructed" to varying degrees. Paired sites were sampled bimonthly from May to July using dual, bow-mounted ichthyoplankton nets (750μm mesh) and a benthic sled (750μm mesh) (Yocum and Tesar 1980). Both were fitted with General Oceanics (General Oceanics Inc., Miami, FL, USA) model 2030R mechanical flow meters. Nets were deployed stationary or pushed through areas that met the standard for SWH (depths <1.5m and current velocities <0.6 m/s) (USFWS 2000, 2003) until a minimum volume of $500m^3$ was sampled. Samples were preserved in 70% ethanol and identified using the larval fish key developed by Auer (1982). I lumped Goldeye (*Hiodon alosoides*) and Mooneye (*Hiodon tergisus*) as *Hidon* spp. because they could not be discriminated at these early life stages, although they were most likely to be Goldeye (A. George, U.S. Geological Survey, personal communication).

Juvenile fishes were measured for total length (mm), and dissected to remove digestive tracts. Items found in the digestive tract were removed, quantified, and identified to the lowest practical taxonomic unit using the aquatic insect key developed by Merrit et al. (1996). Prey items that made up less than 1% of all diet items were grouped into a “rare” prey category and analyzed as a single prey type.

Data Analysis-Prey richness, number of empty stomachs, and proportion of each prey type consumed were quantified and analyzed for each species separately (Freshwater Drum and *Hiodon* spp.) to assess differences in habitat types. Paired analysis (paired t-tests) was used to determine size differences of each species between habitat types.

Feeding patterns for each species were assessed using a graphical method to characterize feeding strategy (specialization vs. generalization), relative prey importance (rare or dominant), and niche variation (within or between individual variation) (Costello 1990; Amundsen et al. 1996). The modified graphical method by Amundsen et al. (1996) was used by plotting prey specific abundance (P_i) against frequency of occurrence (O_i). Prey specific abundance was calculated as $P_i = (\Sigma S_i / S_{ti})$ where P_i represents prey specific abundance of prey i, S_i equals the abundance of prey i, in diets, and S_{ti} equals the total abundance of prey in predators that contain prey i. Frequency of occurrence (O_i) was calculated by dividing the number of fish containing prey i by the total number of fish (within a species) with food in their stomachs (Chipps and Garvey 2007). Differences in numbers of empty stomachs between habitat types for each species were assessed using contingency tables. These analyses were performed using the “vegan” package (Oksanen et al. 2005) in Program R 3.0.2 (R Core Team 2013) and results were considered significant at $P < 0.05$.

Differences in the proportions of each prey consumed by habitat and prey type were assessed using the generalized linear mixed-model procedure for each species (PROC GLIMMIX, SAS 2014). The general linear models contained the effects of habitat type (chute or mainstem), prey, and the interaction between habitat type and prey type. These models used a logit link and a binary distribution (prey type eaten = 1, prey type not eaten = 0). Site (Littles, Lisbon, Rush Bottoms etc.) was a random effect. The same model structure was used to test for differences in prey use by species. This model contained the effects of species (Freshwater Drum or *Hiodon* spp.), prey types that contributed more than 1% of total diet items of both species and the interaction between species and prey type. Subsequent testing of main effect differences in significant interaction terms (i.e. habitat type*prey type; species*prey type) were assessed using a SLICE statement within the generalized mixed-model procedure (SAS 2014).

Results

A total of 350 Freshwater Drum and 145 *Hiodon* spp. were collected in constructed chute SWH compared to 262 Freshwater Drum and 78 *Hiodon* spp. in mainstem SWH. Mean total length of fish did not differ between habitat type (Freshwater Drum, Paired-t(16) = 0.33, P = 0.79) (*Hiodon* spp., Paired-t(5) = 0.97, P = 0.38). Mean total lengths of Freshwater Drum were 5.5 mm and 6.0 mm in chute and mainstem habitats respectively while *Hiodon* spp. were 12.6 mm and 12.3 mm (Table 1). Twenty-one *Hiodon* spp. had not begun exogenous feeding (i.e. had full or partial sac yolk) and were excluded from analysis. Approximately 82% of Freshwater Drum and 87% of *Hiodon* spp. had food in their stomachs. The number of empty stomachs was significantly greater in chute SWH for Freshwater Drum (contingency analysis, $\chi^2 = 54.99$, P = 0.03), but not *Hiodon* spp (contingency analysis, $\chi^2 = 1.62$, P = 0.20). Over 13 prey types were found in dissected digestive tracts, however; 8 of these made up less than 1% of total prey consumed, by count, in each species respectively, and were treated as one prey category ("rare prey") for analysis (Table 2). Prey types that contributed at least 1% of the diet of Freshwater Drum included diptera larvae, diptera pupae, trichoptera larvae, copepods, and ephemeroptera larvae. Prey types that contributed at least 1% of the diet of *Hiodon* spp. were diptera larvae, diptera pupae, trichoptera larvae, ephemeroptera larvae, and odonata larvae. Use of specific prey types differed by habitat type for Freshwater Drum ($F_{5, 3061} = 3.29$, $P < 0.01$) but not *Hiodon* spp.

($F_{5,\ 1155} = 0.67$, $P = 0.65$). Of the prey consumed by Freshwater drum, diptera larvae were consumed more often in mainstem SWH than chute SWH (Figure 2) while all comparisons for *Hiodon* spp. were not significantly different (Figure 3). Use of prey types also differed by species ($F_{4,\ 4231} = 24.83$, $P < 0.01$). Freshwater Drum consumed a greater proportion of diptera larvae while *Hiodon* spp. consumed more diptera pupae, ephemeroptera larvae, and trichoptera larvae (Figure 4).

Graphical analysis indicates that Freshwater Drum specialized on diptera larvae, which were also the dominant prey overall (Figure 5); other prey items were rare and moderately preyed upon. *Hiodon* spp. graphs indicated some specialization ($O_i = 0.67$, $P_i = 0.53$) on diptera pupae and larvae while all other prey items were rare and moderately preyed upon (Figure 6).

Discussion

Riverine fishes are the most important indicators for assessing the ecological integrity of rivers, particularly age-0 fishes (Karr 1991; Schiemer 2003). In the current study, few differences in prey by age-0 fishes were found between habitat types, although the numbers of Freshwater Drum without food items in their stomachs found in chute habitats is notable, as significant declines in fish populations have been attributed to starvation and sub-optimal abiotic conditions (Schiemer 2000). As a proxy for “habitat quality”, number of empty stomachs indicates that chute SWH may not provide the hypothesized benefits, such as higher retention rates of organic matter and increased primary and secondary production (Knowlton and Jones 2000; Bunn et al. 2003; O’Neill and Thorp 2011) to early life stages of Freshwater Drum, although this could be very site specific. For instance, the frequency and duration of flood pulses have very strong influences on the emergence timing and abundance of many aquatic invertebrate species that represent important food sources for YOY fishes (Boulton and Lloyd 1992). Some chute SWH sites in the current study (Lisbon and Overton) experienced reduced and, in some instances, no connection with the mainstem Missouri River during periods of low precipitation in the summer of 2012, causing some chute sites to shift from lotic to lentic (Gosch et al. 2015). Reduced flow through chute SWH could impact the invertebrate assemblage characteristics and abundance, foraging characteristics of the species left in the chute, and increase competition for limited food resources (Garvey and Stein 1998; Gosch et al. 2014). In the current study, 32% of Freshwater Drum and 55% of *Hiodon* spp. with empty stomachs came from these two habitats alone.

The higher proportion of diptera larvae consumed by Freshwater Drum in mainstem SWH could be linked with hydrological dynamics. Significant differences in macroinvertebrate community structures were found between lentic and lotic habitat types in the lower Missouri River (Sampson and Hall 2011). Given that diptera larvae, predominately chironomid larvae, tend to dominate invertebrate communities in newly inundated (< 1 month) lotic habitats, we would expect higher abundances of this prey source in mainstem SWHs as they maintain greater hydrological connection than some chutes (Galat et al. 1998; Gosch et al. 2014). It is possible larval Freshwater Drum are actively selecting for chironomid larvae while *Hiodon* spp. were more opportunistically feeding as indicated by the Amundsen graphs. However, Freshwater Drum collected from inundated floodplain habitat in 2011 specialized on cladocerans (Starks and Long, unpublished data) suggesting that hydrologic conditions may affect feeding strategies. Further information on these feeding relationships could help gauge predator-prey dynamics in restored habitats, as well as indicate if SWHs are providing a greater abundances of prey.

Interestingly, differences in prey use were found even though these specimens were collected in the same habitats at the same time, suggesting niche segregation based on foraging behavior (Reinthal 1990). Diet partitioning in freshwater fishes has been shown to result from differences in morphology as well as life history strategy (Spiegel et al. 2011; Sammons 2012). Morphologically, Freshwater Drum have large mouths when compared to body size, allowing them to prey upon other fish larvae at a young age, although no piscivory was documented in this study. It would be expected to find a greater proportion of larger diet items in *Hiodon* spp. due to their size alone. The average size of *Hiodon* spp. at some sites were more than 10 mm larger than Freshwater Drum larvae, allowing them to consume a greater proportion of larger diet items such as odonatan larvae. Interestingly, other studies of larval *Hiodon* spp. diets found a reliance on cladocerans and corixids while none were found to consume odonata larvae. Consumption of odonata larvae could indicate where *Hiodon* spp. are feeding as these organisms are typically found in depositional zones of lotic waters where they are able to conceal themselves by burrowing (Tennessen 1996; Voshell 2002).

In conclusion, created chute SWH does not appear to providing better habitat quality, in terms of forage, for larval Freshwater Drum and *Hiodon* spp. when compared to similar, unrestored habitats in the adjacent mainstem river. The prevalence of specimens with empty stomachs in chute habitats may even point to a lack of forage for these two species, however; the role of hydrology between habitat type and prey availability needs further investigation. The current study provides some information linking drivers of larval fish communities in the lower Missouri River, however; more study is needed to gain a more holistic perspective of how habitat characteristics structure larval fish communities.

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TABLE 1. Mean (SD) total length (mm) of Freshwater Drum and *Hiodon spp.* collected at each site and habitat types (chute and mainstem) in the summer of 2012.

Site	Freshwater Drum		*Hiodon* spp.	
	Chute SWH	Mainstem SWH	Chute SWH	Mainstem SWH
Rush Bottoms	7.5(2.2)	7.2(2.6)	NA	NA
Worthwine	6.2(1.6)	8.4(1.9)	9.3(2.3)	NA
Lisbon	8.2(3.4)	6.9(3.4)	11.4(2.2)	11.1(0.8)
Jameson	5.3(2.3)	4.6(0.8)	11.7(2.3)	12.0(0.7)
Overton	4.6(0.6)	4.8(1.6)	15.2(3.8)	12.2(2.2)
Littles	4.9(0.8)	5.0(0.8)	18.1(1.8)	19.1(7.8)

TABLE 2. Numbers of each species and proportion of prey types consumed by each species in each respective habitat type.

Prey type	Freshwater Drum Chute SWH (n = 287)	Freshwater Drum Mainstem SWH (n = 232)	*Hiodon* spp. Chute SWH (n = 130)	*Hiodon* spp. Mainstem SWH (n = 72)
Diptera larvae	30.4%	35.7%	27.0%	11.0%
Diptera pupae	10.2%	6.3%	30.3%	11.8%
Trichoptera larvae	5.1%	5.1%	6.4%	4.6%
Copepoda	0.7%	2.5%	0.2%	0.4%
Ephemeroptera larvae	1.1%	1.9%	3.9%	1.1%
Cladocera	0.6%	NA	0.2%	0.4%
Amphipoda	0.4%	NA	NA	NA
Odonata larvae	NA	0.1%	1.1%	0.6%
Coleoptera larvae	NA	NA	0.4%	0.2%
Terrestrial invertebrates	NA	NA	0.2%	0.4%
Hemiptera insects	NA	NA	0.2%	NA
Ostracoda	NA	NA	NA	0.2%
Plecoptera larvae	NA	NA	0.2%	NA

FIGURE 1. Paired chute and mainstem sites on the lower Missouri River sampled for larval fishes in the summer of 2012.

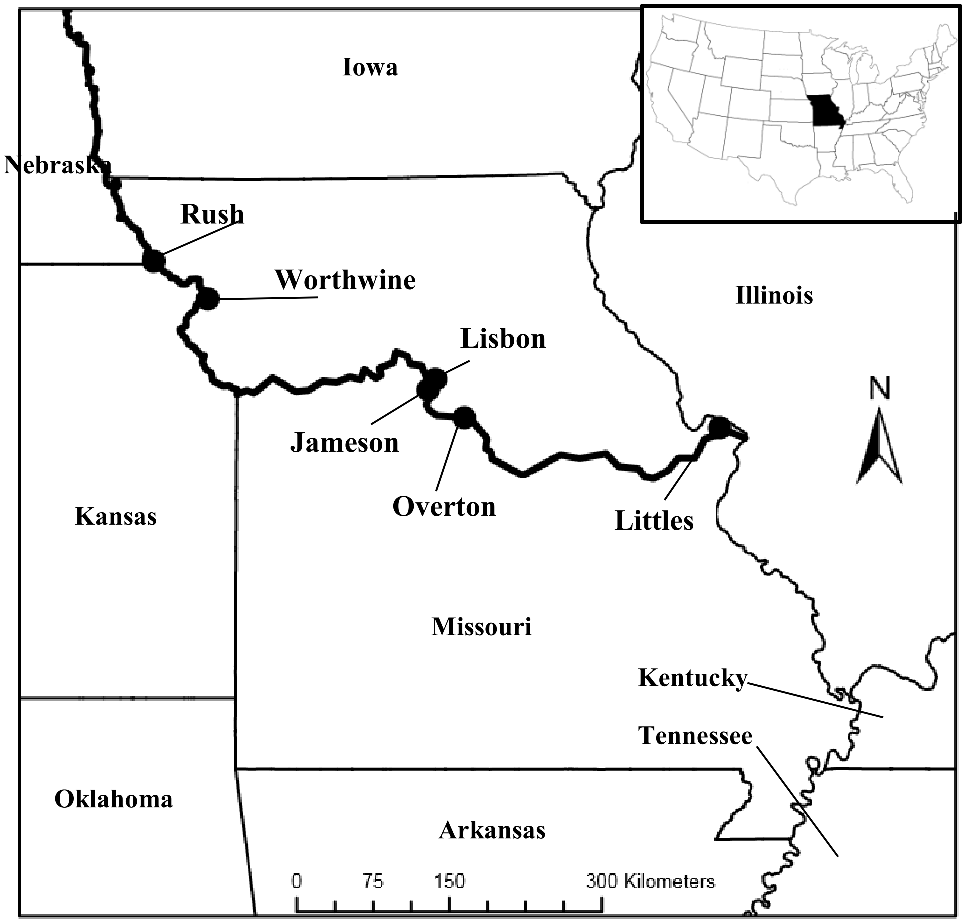


FIGURE 2. Proportions of each individual prey type consumed by larval Freshwater Drum caught in created chute and adjacent mainstem habitats in the summer of 2012.

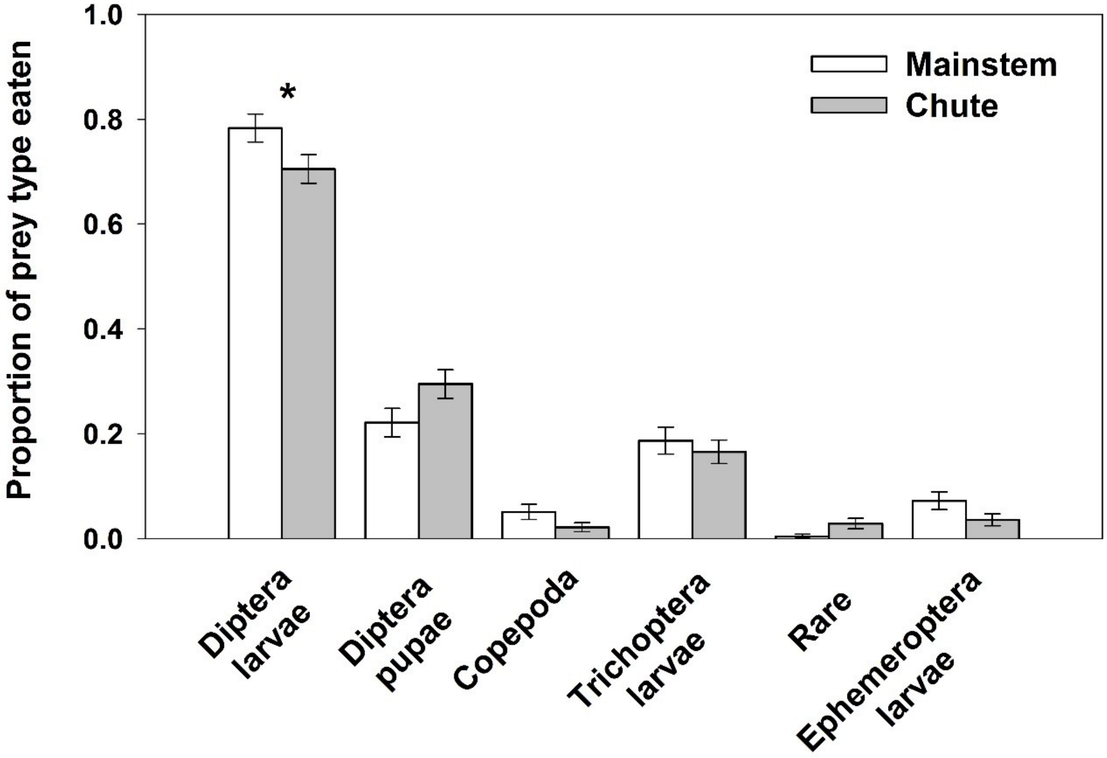


FIGURE 3. Proportions of each individual prey type consumed by larval *Hidon* spp. caught in created chute and adjacent mainstem habitats in the summer of 2012.

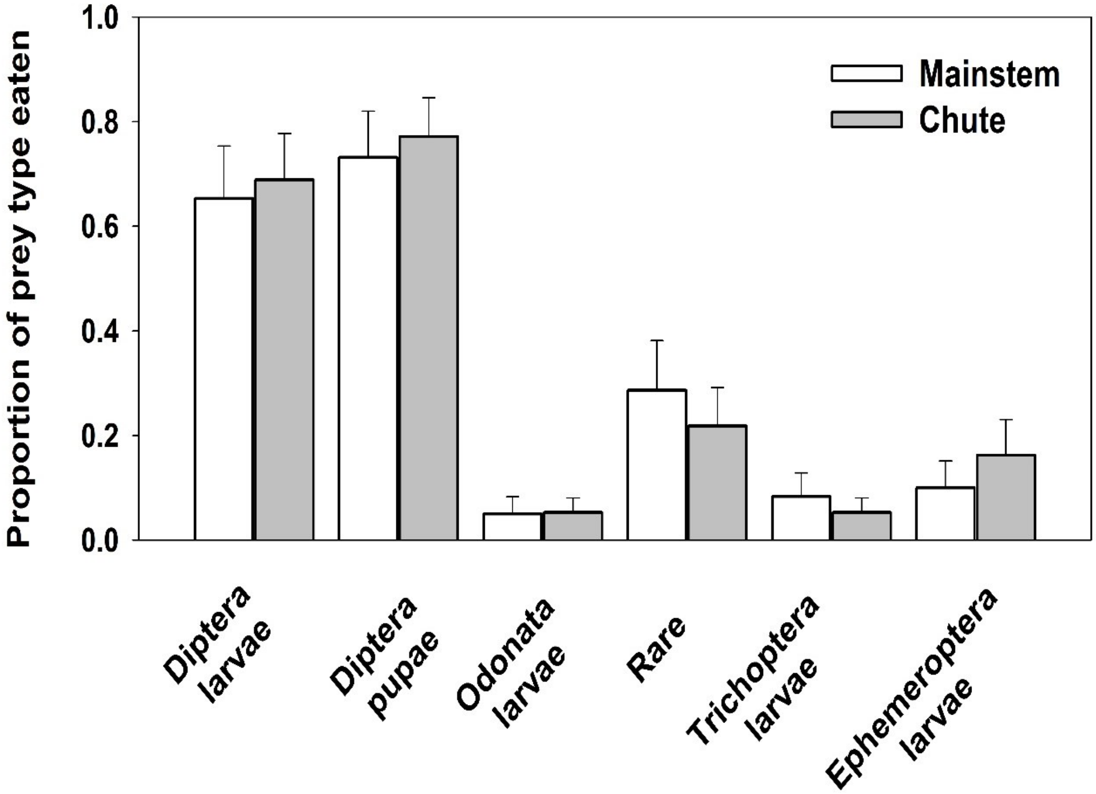


FIGURE 4. Proportions of each individual prey type consumed by larval *Hidon* spp. in comparison to Freshwater Drum caught in shallow-water habitats in the lower Missouri River in the summer of 2012

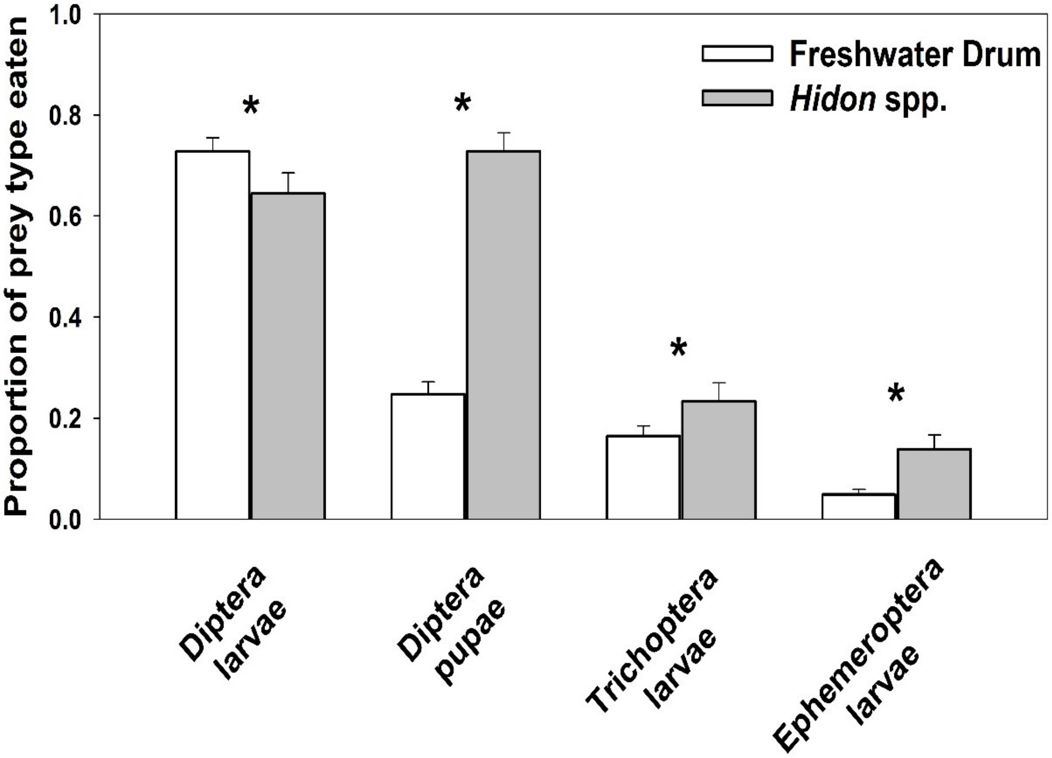


FIGURE 5. Amundsen index graph (along with key) indicating feeding strategy, prey importance, and niche width contribution or prey categories for Freshwater Drum sampled from the lower Missouri River in the summer of 2012.

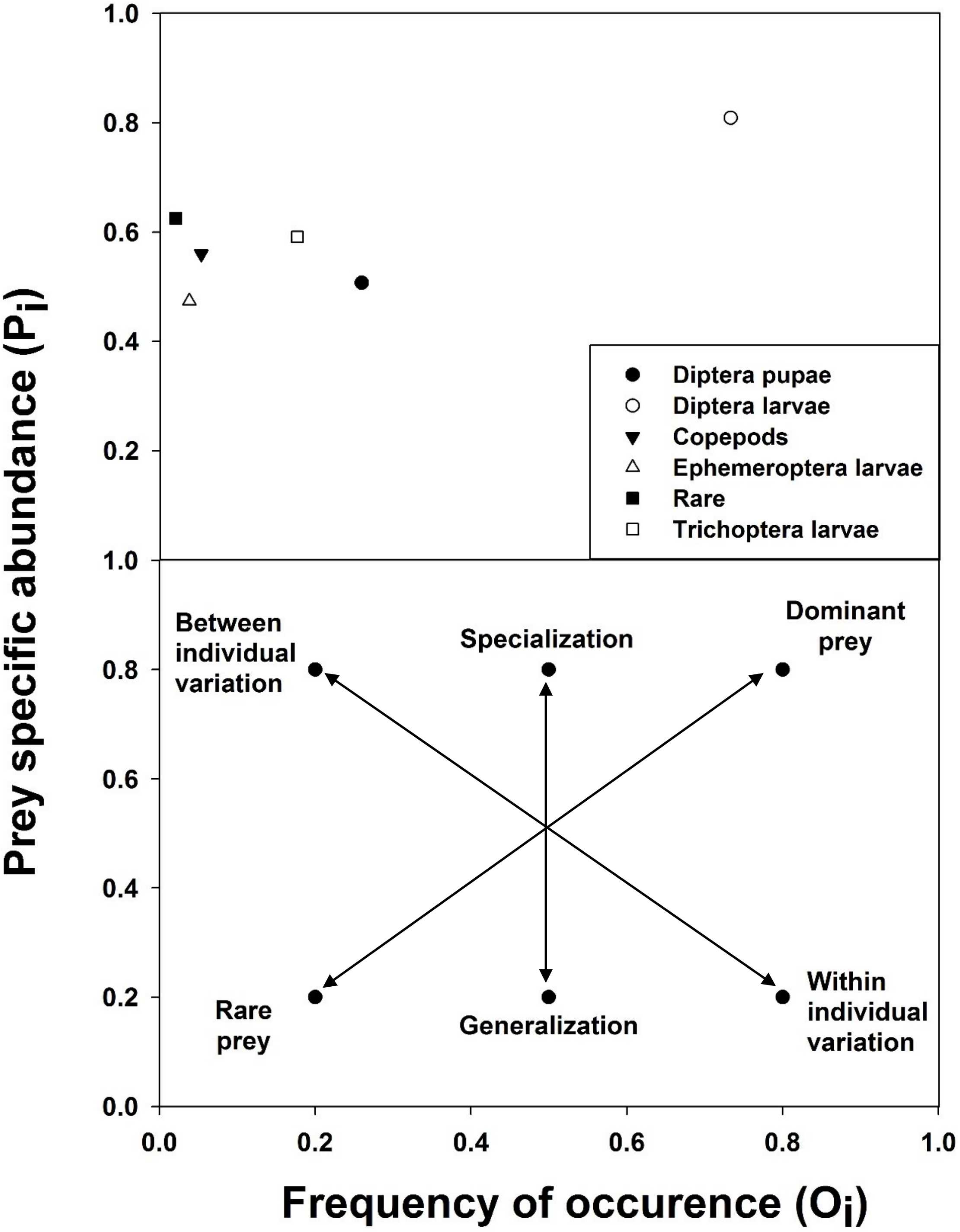
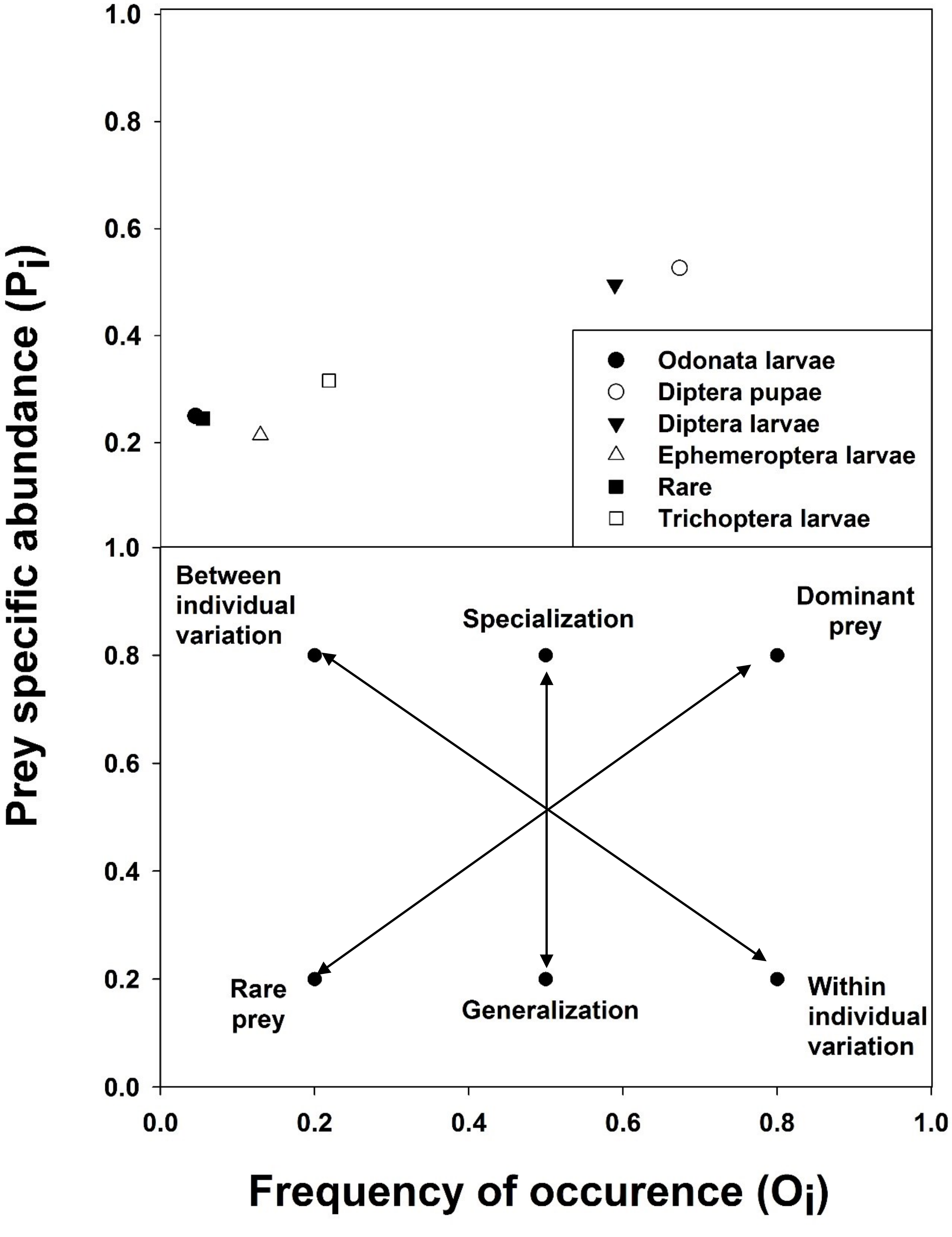


FIGURE 6. Amundsen index graph (along with key) indicating feeding strategy, prey importance, and niche width contribution or prey categories for *Hidon* spp. sampled from the lower Missouri River in the summer of 2012.



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