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The Effects of Chanellization and Channel Restoration on Aquatic Habitat and Biota of the Pecos River, New Mexico

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THE EFFECTS OF CHANELLIZATION AND CHANNEL RESTORATION ON AQUATIC HABITAT AND BIOTA OF THE PECOS RIVER, NEW MEXICO

BY

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THE EFFECTS OF CHANNELIZATION AND CHANNEL RESTORATION ON AQUATIC HABITAT AND BIOTA OF THE PECOS RIVER, NEW MEXICO

This thesis is approved as a creditable and independent investigation by a candidate for the Master of Science degree and is acceptable for meeting the thesis requirements for this degree. Acceptance of this thesis does not imply that the conclusions reached by the candidate are necessarily the conclusions of the major department.

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ABSTRACT

THE EFFECTS OF CHANELLIZATION AND CHANNEL RESTORATION ON AQUATIC HABITAT AND BIOTA OF THE PECOS RIVER, NEW MEXICO

DARREL J. MECHAM

2015

River channelization has been shown to negatively impact riverine ecosystems by degrading aquatic habitat conditions, decreasing diversity of both fish and aquatic invertebrate assemblages and impairing fish recruitment. As knowledge of the negative impacts of channelization has increased, so have channel restoration efforts. We evaluated a recent channel restoration project on the Pecos River, New Mexico by comparing abiotic and biotic conditions among five reaches of river, including the restored reach, an unchannelized reach and a channelized reach all prone to streamflow intermittence and an unchannelized reach and channelized reach more perennial in nature.

Our first objective was to assess reach-scale differences in aquatic habitat conditions and fish assemblage structure and diversity, as well as to assess mesohabitat associations of fish species. We found that aquatic habitat availability was greatest in the more perennial unchannelized reach and that little difference in habitat availability existed among other reaches. In addition, we found minimal differences in habitat diversity among reaches. Diversity of fish assemblages showed no difference among reaches and differences in fish assemblage composition among reaches were minimal. We found that pelagic-broadcast spawning species were associated with high velocity habitats of greater depth and volume, while demersal adhesive spawning species and live bearers were associated with low velocity habitats of lesser depth and volume.

Our second objective was to assess reach-scale differences in nursery habitat availability, fish recruitment and abundance of potential prey items (i.e. meiofauna). We found that slackwater availability was greatest in unchannelized reaches and that differences in recruitment were noticeable for two fish species, the plains killifish (*Fundulus zebrinus*) and western mosquitofish (*Gambusia affinis*). Plains killifish exhibited greater recruitment in unchannelized reaches, while western mosquitofish exhibited greater recruitment in channelized reaches. The abundance of total potential prey items was greatest in the more perennial channelized reach.

Collectively, channel restoration showed no improvement to aquatic habitat conditions, fish assemblage diversity and composition, provision of slackwater nursery areas, recruitment of early-juvenile fishes and provision of potential prey items. We feel that periods of low discharge and streamflow intermittence likely impaired our ability to effectively assess the restoration effort. Thus, we suggest that future restoration efforts focus on a more holistic approach that includes both physical channel restoration and flow regime restoration.

CHAPTER 1: RESPONSE OF FISH ASSEMBLAGES AND AQUATIC HABITAT CONDITIONS TO CHANNELIZATION, RESTORATION AND DEWATERING ALONG A SAND-BED RIVER.

This chapter was co-authored by Christopher W. Hoagstrom and Brian D. S. Graeb. It is formatted for River Research and Applications

ABSTRACT

An increasing awareness of the negative impacts of human disturbances to riverine ecosystems has led to an increase in river restoration projects. We provided an evaluation of a recently restored reach of the Pecos River, New Mexico by comparing aquatic habitat conditions and fish assemblage characteristics of the restored reach with unchannelized and channelized reaches. We also assessed mesohabitat associations of fish species collectively. Habitat availability was greatest in the more perennial unchannelized reach, where diversity of velocities was high. Red shiner (*Cyprinella lutrensis*) were most abundant in the more perennial channelized reach and common carp (*Cyprinus carpio*) were most abundant in the restored reach. Red shiner were largest in the channelized reaches and western mosquitofish (*Gambusia affinis*) were larger in the more perennial channelized reach than in the restored reach. Pecos bluntnose shiner (*Notropis simus pecosensis*) and plains killifish (*Fundulus zebrinus)* were larger in the more perennial reaches regardless of channel morphology. Differences in assemblage composition among reaches were minimal and no differences in species diversity metrics were observed among reaches. Pecos bluntnose shiner, speckled chub (*Macrhybopsis aestivalis*), Plains minnow (*Hybognathus placitus*) and Rio Grande shiner (*Notropis jemezanus*) were associated with mesohabitats of greater velocity, depth and volume than all other species sampled. Low discharge and intermittence led to a decline in the aforementioned species and an increase species more tolerant to harsh physico-chemical conditions accompanying periods of low discharge, including red shiner, plains killifish and western mosquitofish. These conditions likely confounded our ability to assess the effects of river channel restoration. We suggest a more holistic approach to river

restoration is necessary, which includes physical channel restoration, as well as mimicry of the natural flow regime and the maintenance of minimum base flows. Without such an approach, the ability to assess the success of physical channel restoration may be masked during periods of low discharge and intermittence.

INTRODUCTION

Human disturbances to riverine ecosystems are widespread in the Desert Southwest USA (Carlson and Muth, 1989; Roberge, 2002) and include the introduction of nonnative species, fragmentation, dewatering, and channelization. The effects of channelization caused indirectly by altered flow regimes or by direct modification to river channels are particularly common (Kennedy and Turner, 2011), and can lead to a decrease in channel complexity by reducing the size and variety of aquatic habitats within the river channel (Morris et al., 1968) and by separating rivers from their floodplain (Jurajda, 1995). These alterations have been shown to negatively impact fishes by decreasing fish diversity and biomass (Huggins and Moss, 1975; Paragamian, 1987; Shields et al., 1994), reducing larger size classes of fish (Oscoz et al., 2005) and reducing recruitment to adult life stages (Jurajda, 1995).

In response to these impacts, channel restoration projects have become more common (Bernhardt et al., 2005) and the amount of funding directed toward these projects has increased (Bernhardt et al., 2007). In North America, restoration projects have shown a consistent increase since the early 1990's (Bernhardt et al., 2005) and expenditures focused on restoration projects, including those focused specifically on channel restoration, have increased to one billion dollars annually (Bernhardt et al., 2007).

The goals and techniques of channel restoration are diverse. Some goals of channel restoration projects are focused on the conservation and recovery of threatened and endangered species (Steppen, 2000), the enhancement of sport fisheries (Muotka et al., 2002), maximizing biodiversity (Lepori et al., 2005), improving stream-bed habitat heterogeneity and enhancing overall ecosystem function (Carline and Klosiewski, 1985). Channel restoration techniques include bank stabilization, floodplain reconnection, channel reconfiguration and the implementation of in-stream habitat improvement structures such as boulders, wood, gyrones, and artificial riffles and pools (Bernhardt et al., 2007; Palmer et al., 2009).

The success of channel restoration projects in relation to their goals is variable. Some projects report being successful (e.g. Nakano and Nakamura, 2008), while others report being unsuccessful (e.g. Pretty et al., 2003). Successful channel restoration projects often focus on the ecology and specific habitat requirements of the target species (e.g. Palm et al., 2007) and are conducted at spatial scales relevant to the target species (Bond and Lake, 2005). Unsuccessful restoration projects often lack this focus or are conducted at spatial scales that may be too small (e.g. Lepori et al., 2005). Additionally, very few restoration efforts provide a post-project evaluation or assessment (Bernhardt et al., 2007). Thus, it is difficult to judge the success of restoration efforts and use successful projects as a model to guide future restoration efforts when the results of restoration projects are rarely evaluated and even more rarely reported.

A recent channel restoration project conducted on the Pecos River, New Mexico presents an opportunity to evaluate the effects of channelization and channel restoration on a declining guild of fishes; the riverine minnows. Riverine minnows are represented in the Pecos River, NM by the federally and state (NM, TX) threatened Pecos bluntnose shiner (*Notropis simus pecosensis*), the Rio Grande shiner (*Notropis jemezanus*) and the speckled chub (*Macrhybopsis aestivalis*). The federally and state (KS, NM, OK, TX) threatened Arkansas River shiner (*Notropis girardi*) and plains minnow (*Hybognathus placitus)* are also members of this guild that occur in the river, but are non-native. Species of this guild primarily utilize mid-depth fluvial main channel habitats of higher velocity (Hoagstrom et al., 2008a; Hoagstrom and Brooks, 2005; Hoagstrom et al., 2010; Hoagstrom et al., 2015). Riverine minnows are found almost exclusively in unchannelized reaches of the river, especially in the case of adult populations (Hoagstrom and Brooks, 2005; Hoagstrom et al., 2008a; 2008b; Hoagstrom et al., 2010; Hoagstrom et al., 2015). Dewatering and channelization have been shown to be influential in their decline (Hoagstrom et al., 2008b) and channel restoration has been suggested as a way to expand the distribution of adult populations by increasing the amount of favorable unchannelized habitat (Hoagstrom et al., 2008a; 2008b).

Although river channel restoration has been suggested to benefit riverine minnows specifically, it is likely that it may also benefit another guild of fishes found in the river, the river's-edge fishes. Common species of the river's-edge guild that occur in the Pecos River, NM include the red shiner (*Cyprinella lutrensis*) the plains killifish (*Fundulus zebrinus*) and the western mosquitofish (*Gambusia affinis*). These species utilize lower velocity and/or deeper habitats that occur near the river's edge, which are often associated with vegetation, debris piles, or cut banks (Matthews and Hill, 1979; Meffe and Sheldon, 1988; Ostrand and Wilde, 2002). In contrast to riverine minnows, few or no studies have been conducted that examine the effects of channelization on river’s-edge fishes that occupy the Pecos River, NM.

The goal of this study is to provide a post-project evaluation of a recent channel restoration project conducted on the Pecos River, NM as it relates to the structuring of habitat conditions and fish assemblages that occupy the river (both riverine minnows and river’s-edge fishes). Our objectives are to: (1) compare aquatic habitat conditions among reaches; (2) compare abundance and size structure of fishes among reaches; (3) compare fish assemblage composition and diversity among reaches; and (4) assess mesohabitat associations of fish. We hypothesized that: (1) channel restoration would create greater availability and diversity of aquatic habitat (i.e. greater wetted width (m) and diversity of depth (m) and velocity (m / s); (2) restoration would increase the distribution of adult riverine minnows; (3) fish assemblages would be most similar among the restored and unchannelized reaches and would be most diverse in these reaches; and (4) that riverine minnows would show more affinity for higher velocity mesohabitats.

METHODS

Study area

The study area is located on the Pecos River within the middle tract of the Bitter Lake National Wildlife Refuge. Before 1940, the river in the Bitter Lake NWR was a dynamic river with large oxbows that were constantly eroding banks and changing course (USBOR, 2009). A river-diversion project that began in 1940 created a channel that bypassed large river bends, shortened the river channel, and created a series of abandoned river meanders that created isolated oxbows. This was done to reduce flood damage to bordering agricultural lands and waterfowl impoundments. This channelization effort cut off the river from its natural floodplain and created a more incised, simplified river channel. Channelization also increased the encroachment of non-native salt cedar (*Tamarix* spp.), which has exacerbated the effects of channelization via bank stabilization (USBOR, 2009).

River-channel restoration efforts were conducted in 2009 in an attempt to restore the channelized section of river. Four techniques were employed in the restoration including redirecting the river into an historic oxbow, removal of non-native vegetation, bank lowering, and reworking channel morphology (USBOR, 2009). Collectively, these efforts were conducted to create a wider, more dynamic river channel, similar to unchannelized reaches upstream from the middle tract of the refuge (USBOR, 2009).

Five study sites were used to assess the effects of the restoration effort on aquatic habitat conditions and on fish assemblages. Sites were selected based on accessibility and to allow for comparisons of the restored site with other channel morphology types. Three sites are located within the Bitter Lake NWR (Figure 1). These include the recently restored site, an unchannelized site approximately 2 km upstream from the restored site and a channelized site approximately 2 km downstream from the restored site. Because the proximity of sites within the Bitter Lake NWR could mask biological and habitat differences, two remote sites were also included in the study. These included an unchannelized site 63 km north of the restored site, hereafter referred to as the “braided” site, as well as a highly channelized site 60 km south of the restored site within the William S. Huey Wildlife Area, hereafter referred to as the “ditched” site.

Field

Field work was conducted at each site during June and August of 2012 and 2013. Attempts were made to sample during stable base flow conditions and to avoid periods of flooding and intermittence in order to facilitate comparisons among sampling periods (Kwak and Peterson, 2007). Despite attempts to sample during periods of stable base-flows, conditions of low discharge were present during sampling trips that were much lower than representative historical conditions based on an assessment of twenty years of pre-dam data (Figure 2). Prolonged periods of intermittence also occurred between sampling trips (Figure 2). Additionally, flooding prevented sampling of the braided site during the final sampling trip.

General habitat conditions were surveyed by establishing a set upstream boundary at each site. Stream length to be sampled was determined by measuring ten wetted widths beginning at the fixed upstream boundary of each site. Afterward, the mean wetted width was calculated. Site length was forty wetted widths. Each site was divided by ten to yield eleven equally spaced transects. Depth, velocity, and stream-bottom temperature were measured at one-meter intervals at each transect (Fitzpatrick et al., 1998) starting on the right bank. The number of riffle-pool sequences found within each site’s length was also recorded and the site water-surface gradient was measured on the stream bottom from thalweg to thalweg across two meander wavelengths. (Fitzpatrick et al., 1998).

A sampling area of two meanders (two riffle-pool sequences) representative of the variety of mesohabitats present in the forty wetted width site length was used to sample fish and to gather data on the physico-chemical nature of each mesohabitat sampled. For our study, we defined mesohabitats as areas of relatively uniform depth and velocity

(Jackson 1975). Due to the disruptive nature of the sampling methods employed, sampling proceeded from downstream to upstream, one mesohabitat at a time.

Fish were seined from all mesohabitats within the two meander sampling area using a 3.0-m seine with 3.2-mm mesh. Fish were sorted and measured upon capture. If fish were gravid or exhibited breeding colors/characteristics, this information was recorded. Specimens collected were released after the aforementioned information was recorded for each fish.

After fish were sampled at each mesohabitat, distance seined, distance to shore, maximum width, maximum depth and maximum length of the mesohabitat were measured and the presence or absence of active or senescent streambed ripples was recorded. All mesohabitats were photographed and given a brief fluvial-geomorphological description. Common mesohabitats sampled included a variety of pools, riffles, runs and slackwaters that were named using nomenclature modified from Polivka (1999) and King (2004).

Statistical analyses

All early-life-stage fishes (individuals < 25-mm SL) were excluded from analyses. This length cutoff was chosen based on the efficiency of our sampling gear. This gear has been used in past studies on the Pecos River (e.g. Hoagstrom and Brooks, 2005; Hoagstrom et al., 2008b) and with it we were confident that we could collect a representative sample of larger fish.

We included two covariates in our models assessing spatial differences in fish abundance, which was rank-transformed due to non-normality. These were mesohabitat volume (m^3) and conductivity (µmhos). These covariates were included in the analyses because both have been shown to affect plains stream fish assemblages (Ostrand and Wilde, 2004; Hoagstrom, 2009). We used a Spearman's ρ test to assess correlation between the covariates prior to inclusion in our analyses and found that they were not highly correlated (i.e. Spearman's ρ values were between -0.43 and -0.01).

We compared habitat availability among sites and sampling trips by comparing discharge (m^3 / s) and wetted width (m) of the river channel. We compared the diversity of depth and velocity point measurements among sites by first grouping depth and velocity measurements into 0.1 (m) intervals (depth) or 0.1 (m / s) intervals (velocity). We then calculated Fisher's α diversity values from the number of values in each interval for both depth and velocity. Resulting diversity values were compared among sites using pairwise diversity permutation tests (Hammer et al. 2001). We assessed the relationships of wetted width and diversity of depth and velocity points to discharge.

Rank abundance of common fish species (i.e. those species comprising at least 5% of the total assemblage) was compared among sites using separate ANCOVA's for each species, with conductivity (µmhos) and volume (m^3) used as covariates. If assumptions of ANCOVA were violated for specific species, ANOVA was used to compare the abundance of those species among sites. If a significant effect was observed, Tukey's HSD (Honestly Significant Difference) tests were used to assess which sites differed significantly from each other with regard to specific species (King, 2004).

Size structure of common fish species was compared among sites using separate Kruskal-Wallis tests. If a significant effect was observed, pairwise Wilcoxon signed-rank tests were used to assess which sites differed significantly from each other with regard to specific species.

Assemblage structure of fish species comprising at least 1% of the total assemblage was compared among sites and trips using non-metric multidimensional scaling (NMS) with Bray-Curtis dissimilarity used as the resemblance measure. Bray-Curtis dissimilarity was chosen because it is a quantitative measure that incorporates species abundance and has been shown to be a robust measure of ecological distance (Faith et al., 1987). All data were Log_{10} transformed prior to analysis. This was done in order to reduce the likelihood of overly abundant species masking community changes with regard to less prevalent species (i.e. riverine minnows), which were our primary interest in this study. The ordination was considered useful for interpretation if the stress value was below 0.20 (Clark and Warwick 2001). Permutational multivariate analysis of variance (PERMANOVA) using 9,999 permutations was used to assess whether patterns depicted in the NMS ordination were statistically significant.

Richness and diversity of fish assemblages was compared among sites using rarefied species richness and Fisher's α diversity. Rarefaction of species richness allows for comparison of species richness among samples of different sizes and was used to account for differences in sampling effort among sites (Kwak and Peterson, 2007). Fisher's α was used for the diversity measure because it is affected most by species of average abundance and is unaffected by sample size (Kempton and Taylor, 1974). Fisher's α diversity values were also compared among sites using diversity permutation tests (Hammer et al. 2001).

Mesohabitat associations of fish were assessed using canonical correspondence analysis (CCA). We assessed associations of species comprising at least 1% of the total assemblage with conductivity (µmhos), mesohabitat volume (m^3), depth (m), velocity (m

/ s), and water temperature (°C). We also assessed relationships of all sites by sampling trip with the same mesohabitat variables. All fish abundance data were Log_{10} transformed prior to analysis to help meet assumptions of CCA.

All analyses used to compare fish abundance and size structure were conducted in JMP 11 (SAS Institute 1989-2007). All analyses used to calculate and compare diversity metrics, assess differences in assemblage structure and to assess associations of sites and fish species with mesohabitat variables were conducted in Program PAST (Version 3.04) (Hammer et al. 2001).

RESULTS

Habitat availability & diversity

Discharge (m^3 / s) and wetted width (m) varied among sites and sampling trips (Figure 3; A-B). Discharge was greater in the braided site than in all other sites. All other sites exhibited similar levels of discharge, especially during the August sampling trips of both years. Wetted width was also greater in the braided site than in all other sites. When comparing all other sites, wetted width was similar. There were no differences in the diversity of depths among sites (Figure 4A) and only one difference in the diversity of velocities was observed, with the braided site exhibiting a greater diversity of velocities than the channelized site (Figure 4B).

Fish

We collected a total of 3,035 juvenile and adult fishes (individuals ≥ 25 mm SL) during the study period. The assemblage was dominated by red shiner (*Cyprinella lutrensis*), plains killifish (*Fundulus zebrinus*) Pecos bluntnose shiner (*Notropis simus* *pecosensis*), western mosquitofish (*Gambusia affinis*) and common carp (*Cyprinus carpio*), which composed 88% of all fish species sampled (Appendix I).

Site scale differences in rank abundance were observed for *C. lutrensis* and *C. carpio* (Figure 5). *C. lutrensis* were more abundant in the ditched site than in the unchannelized site and the channelized site. *C. carpio* were more abundant in the restored site than in the braided site. In addition, *C. lutrensis* and *C. carpio* exhibited a negative relationship with high levels of conductivity (μmhos) (Table 1), while all other common species showed no relationship with conductivity.

Site scale differences in size structure existed for all common species sampled with the exception of *C. carpio* (Figure 6). *N. s. pecosensis* were largest in the braided site and the ditched site, smaller in the unchannelized site and the channelized site and smallest in the restored site. *C. lutrensis* were largest in the ditched site and the channelized site, smaller in the restored site and the unchannelized site and smallest in the braided site. *G. affinis* were larger in the ditched site compared to the restored site. *F. zebrinus* were larger in the braided site than in the unchannelized site and restored site and were larger in both the ditched site and the braided site than in the restored site.

Non-metric multidimensional scaling depicted spatial and temporal differences in fish assemblage composition (Figure 7). Although spatial overlap was present, patterns in assemblage structure showed fish assemblages of the braided site and the ditched site grouping together and fish assemblages of the refuge sites grouping together. Permutational multivariate analysis of variance (PERMANOVA) confirmed differences in assemblage structure among sites (*F* = 1.8, *P* = 0.048). However, post-hoc pairwise comparisons indicated that the only significant difference among sites was between the ditched site and the channelized site ($P < 0.05$). Temporal patterns in the ordination indicated a shift in assemblage structure following the initial sampling trip in June 2012. No differences in rarefied species richness or Fisher's α diversity of fish assemblages were observed among sites (Figure 8).

Canonical Correspondence Analysis (CCA) gave an adequate representation in two dimensions of associations of both fish species and sites with mesohabitat variables (Figure 9). The eigenvalue of axis 1 was 0.132 and the eigenvalue of axis 2 was 0.072. Both axes together explained approximately 85% of the variation in the ordination. Patterns in habitat conditions by site and sampling trip indicated that velocity (m / s), depth (m) and mesohabitat volume (m^3) decreased after June 2012 and levels of conductivity (µmhos) increased concurrently. This change was more apparent for the refuge sites than for the braided site and the ditched site. Patterns in habitat associations of fish species suggest that riverine minnows showed a greater affinity for larger, deeper habitats of higher velocity than river's-edge fishes. Specifically, *M. aestivalis* showed the strongest association with higher velocity, followed by *N. s. pecosensis* and *H. placitus*. *N. jemezanus* showed the strongest association with greater depth (m) and volume (m^3). River's-edge fishes showed a greater affinity for slower moving shallower habitats than riverine minnows. They also showed a greater tolerance for high levels of conductivity (µmhos). This was especially true for *G. affinis* and *F. zebrinus*. Of the river's-edge fishes, *C. carpio* and *C. lutrensis* were most similar to riverine minnows with regard to habitat associations.

DISCUSSION

Habitat availability & diversity

Our finding that the braided site exhibited greater habitat availability than all other sites and greater velocity diversity than the channelized site was expected, as other studies have shown that the river section containing the braided site is more perennial in nature than the section containing the refuge sites (unchannelized, restored and channelized) and has a wider less confined river channel (Hoagstrom et al., 2008a; 2008b). The ditched site is normally characterized by higher levels of discharge than the refuge sites (Hoagstrom et al., 2008b). However, in comparison to historic conditions, extremely low-discharge and intermittence occurred during our study (Figure 2); resulting in similar levels of discharge between the ditched site and the refuge sites. In these sites, we observed that the river began to re-meander within the riverbed. In this context, re-meandering meant that the small ribbon of water remaining within the channel began to move back and forth from bank to bank across the riverbed. This created aquatic habitat conditions that were similar across channel morphologies. Therefore, even though restoration efforts may have lowered banks and widened the river channel, our ability to assess whether this provided greater habitat availability and diversity for fish was likely limited without the presence of adequate base flows following the initial sampling trip in June 2012.

Fish

C. lutrensis and *C. carpio* were the only species that exhibited site scale differences in rank abundance. *C. lutrensis* were more abundant in the ditched site than in the unchannelized site and the channelized site. Carrol et al. (1977) showed that *C. lutrensis* can thrive in channelized reaches of rivers and can sometimes be more abundant in channelized reaches compared to unchannelized reaches. This is a potential explanation as to why they occurred in greater abundance in the ditched site. *C. lutrensis* also showed a negative relationship with conductivity (Table 1). This was expected as this species is less tolerant of high conductivity than other common species we collected (Ostrand and Wilde, 2004). *C. carpio* were more abundant in the restored site than in the braided site (Figure 5). It is likely that this pattern had little to do with habitat conditions. All *C. carpio* collected belonged to a single cohort of age-0 fish that were only collected in June 2012. This species is not common in our study area and may have simply moved out of our study area after the first sampling trip. Another explanation is that they experienced significant mortality after the onset of low discharge conditions. However, Crook et al. (2001) showed that *C. carpio* are highly tolerant to drought conditions. Hence, movement out of the study area may be the most likely explanation.

Site-scale patterns in size structure differed by species. Our hypothesis that restoration would increase the distribution of adult riverine minnows was not supported, as *N. s. pecosensis* exhibited larger sizes in the braided site and the ditched site in comparison to the restored site and the refuge sites in general. The occurrence of larger sizes of *N. s. pecosensis* in the braided site coincides with past research, which has shown that adults of this species are most common here (Hoagstrom et al., 2008b). The presence of larger individuals in the ditched site was unexpected as Hoagstrom et al. (2008b) showed adults to be virtually absent in these reaches. Site scale differences in size structure of *F. Zebrinus* were similar to those observed for *N. s. pecosensis*. This species exhibits the most similar habitat preferences to *N. s. pecosensis* of any species we collected (Rahel and Thel, 2004), which likely explains similar site-scale patterns in size structure. *C. lutrensis* individuals were smallest in the braided site and increased in size with increasing levels of channelization, again this may be due to their ability to thrive in channelized reaches and the presence of more river's-edge habitat in these areas. Size structure of *G. affinis* and *C. carpio* exhibited little to no differences among sites.

Differences in fish assemblage structure among sites were difficult to discern collectively. However, examining differences in assemblage structure through time yielded greater insight. During the initial sampling trip, the unchannelized site and the restored site showed the greatest similarity with the braided site. After the initial sampling trip however, assemblages of these sites shifted away from those of the braided site. The ditched site then became and remained most similar to the braided site for subsequent sampling trips. While discharge was not greater in the ditched site in comparison to the refuge sites, the ditched site did appear to have more standing water and was more lentic in nature in comparison to the refuge sites (personal observation), especially after the first sampling trip. This may have created a refuge area for riverine minnows as these species disappeared from the refuge sites following the first sampling trip, but remained in the ditched site (Appendix I). The decline of riverine minnow abundance, along with the establishment of several tolerant river's-edge species including *G. affinis*, *F. zebrinus* and *C. lutrensis* (Figure 10) was one of the driving factors behind patterns we saw in the ordination and was likely the reason why we saw no differences in rarefied species richness or Fisher's α diversity of fish assemblages among sites.

Observed mesohabitat associations of fish species supported our hypothesis that riverine minnows would associate with higher velocity habitats. This finding has been observed in other studies examining habitat associations of this guild (Hoagstrom and Brooks, 2005; Hoagstrom et al., 2008a). We also found that river's-edge species including *C. lutrensis*, *F. zebrinus*, *G. affinis* and *L. parva* associated with lower velocity habitats than riverine minnows and were found in habitats that had higher conductivity than riverine minnows. This finding is also supported by other studies, which have found that river's-edge species exhibit such habitat associations and are more tolerant of higher levels of conductivity and salinity than riverine minnows (Echelle, 1972; Matthews and Hill, 1979; Meffe and Sheldon, 1988; Ostrand and Wilde, 2004). We also saw decreases in velocity, depth, and mesohabitat volume within the unchannelized, restored and channelized refuge sites after the first sampling trip, while habitat conditions within the braided site and the ditched site remained similar to conditions observed during the first sampling trip. The shift away from habitat conditions favorable for riverine minnows in the refuge sites is likely why riverine minnows became absent from these sites, but persisted in the braided site and the ditched site where aquatic habitat conditions remained more stable through time.

Management implications

Our goal for this study was to assess whether channel restoration could improve general habitat conditions for the benefit of fish assemblages occupying the river, specifically threatened riverine minnow species. While we were able to see some potential benefits of the restoration as it relates to fish assemblage similarity with the braided site during our first sampling trip in June of 2012, our ability to assess benefits of the restoration as it relates to the total fish assemblage, as well as riverine minnow species specifically, was impaired after the first sampling trip. After the initial sampling trip, we saw a subsequent decline in riverine minnows and the establishment of a few dominant river's-edge species. Had adequate base flows remained throughout the study period, we feel that we would have been able to better assess the benefits of physical channel restoration for riverine minnows and potentially other river's-edge species that are less tolerant of periods of low discharge. Indeed, periods of low discharge and intermittence within our study area have been shown to result in the decline of populations of riverine minnows (Hoagstrom et al., 2008b).

Our findings are not unique to the Pecos River, NM however. Conditions of low discharge and intermittence are a threat to riverine minnows and other fishes throughout the plains (Cross and Moss 1987; Ostrand and Wilde, 2004; Durham and Wilde, 2009; Falke et al., 2010; Hoagstrom et al., 2011) and have been shown to reduce diversity and alter the structure of plains stream fish assemblages and lead to extirpation of sensitive species, primarily riverine minnows (Cross and Moss, 1987; Perkin et al., 2014; Perkin et al., 2015). Based on these findings, we suggest that a more holistic approach be taken when planning restoration projects that includes both physical habitat restoration, as well as the mimicry of the natural flow regime, including the maintenance of minimum base flows. This approach has been suggested as a way to benefit riverine minnows of the Pecos River, NM (Dudley and Platania, 2007; Hoagstrom et al., 2008b) and we feel that it would be applicable to riverine minnows throughout the plains. Without such an approach, it is likely that channel restoration itself will not provide the desired effect of creating habitat conditions necessary to enable the conservation of this imperiled guild. In cases where such an approach is not feasible and efforts are limited to physical channel restoration, we recommend continued monitoring in order to assess whether channel restoration is successful at providing favorable habitat conditions during time periods when adequate base flows are present.

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We would like to thank the U.S. Bureau of Reclamation for providing funding for this project. We also thank personnel of the Bitter Lake National Wildlife Refuge, specifically Floyd Truetken and Jeff Sanchez for help in coordinating sampling efforts within the refuge. Additionally, we thank Stephen Davenport for local expertise and Daniel Dembkowski for help with statistical analysis.

REFERENCES

Bernhardt ES. 2005. Synthesizing U.S. river restoration efforts. *Science (Washington)* **308(5722)**:636-637.

Bernhardt ES, Sudduth EB, Palmer MA, Allan JD, Meyer JL, Alexander G, Follstad-Shah J, Hassett B, Jenkinson R, Lave R, Rumps R, Pagano L. 2007. Restoring rivers one reach at a time: results from a survey of U.S. river restoration practitioners. *Restoration Ecology* **15(3)**:482-493.

Bond NR, Lake PS. 2005. Ecological restoration and large-scale ecological disturbance: the effects of drought on the response by fish to a habitat restoration experiment. *Restoration Ecology* **13(1)**:39-48.

Carline RF, Klosiewski SP. 1985. Responses of fish populations to mitigation structures in two small channelized streams in Ohio. *North American Journal of Fisheries Management* **5**:1-11.

Carlson CA, Muth RT. 1989. The Colorado River: lifeline of the American southwest. *Canadian Special Publication of Fisheries and Aquatic Sciences* **106**:220-239.

Carroll JH Jr., Ingold D, Bradley M. 1977. Distribution and species diversity of summer fish populations in two channelized rivers in Northeast Texas. *The Southwestern Naturalist* **22(1)**:128-134.

Clarke K, Warwick RM. 2001. Change in marine communities: an approach to statistical analysis and interpretation.

Crook DA, Robertson AI, King AJ. 2001. The influence of spatial scale and habitat arrangement on diel patterns of habitat use by two lowland river fishes. *Oceologia* **129**:525-533.

Cross FB, Moss RE. 1987. Historic changes in fish communities and aquatic habitats in plains streams of Kansas. *Community and evolutionary ecology of North American stream fishes. University of Oklahoma Press, Norman* **(1987)**:155-165.

Dudley RK, Platania SP. 2007. Flow regulation and fragmentation imperil pelagic-spawning riverine fishes. *Ecological Applications* **17**:2074-2086.

Durham BW, Wilde GR. 2009. Effects of streamflow and intermittency on the reproductive success of two broadcast-spawning Cyprinid fishes. Copeia 2009(1):21-28.

Echelle AA, Echelle AF, Hill LG. 1972. Interspecific interactions and limiting factors of abundance and distribution in the Red River pupfish, *Cyprinodon rubrofluviatilis*. *American Midland Naturalist* **(1972)**:109-130.

Faith DP, Minchin PR, Belbin L. 1987. Compositional dissimilarity as a robust measure of ecological distance. *Vegetatio* **69**:57-68.

Falke JA, Bestgen KR, Fausch KD. 2010. Streamflow reductions and habitat drying affect growth, survival, and recruitment of brassy minnow across a Great Plains riverscape. *Transactions of the American Fisheries Society* **139**:1566-1583.

Fitzpatrick FA, Waite IR, D'Arconte PJ, Meador MR, Maupin MA, Gurtz ME. 1998. Revised methods for characterizing stream habitat in the national water-quality assessment program. U.S. Geological Survey Water-Resources Investigations Report 98-4052.

Hammer, Ø., D. A. T. Harper, and P. D. Ryan. 2001. PAST: Paleontological statistics software package for education and data analysis. Palaeontologia Electronica 4(1): 9pp. http://palaeo-electronica.org/2001_1/past/issue1_01.htm

Hoagstrom CW. 2009. Causes and impacts of salinization in the Lower Pecos River. *Great Plains Research* **19**:27-44.

Hoagstrom CW, Archdeacon TP, Davenport SR, Propst DL, Brooks JE. 2015. Intra-fragment riverscape conservation for an imperiled, small-bodied, pelagic-broadcast spawning minnow: speckled chub, *Macrhybopsis aestivalis*. *Canadian Journal of Fisheries and Aquatic Sciences* **72**:527-537.

Hoagstrom CW, Brooks JE. 2005. Distribution and status of Arkansas River shiner *Notropis girardi* and Rio Grande shiner *Notropis jemezanus*, Pecos River, New Mexico. *Texas Journal of Science* **57(1)**:35-58.

Hoagstrom CW, Brooks JE, Davenport SR. 2008a. Recent habitat association and the historical decline of *Notropis simus pecosensis*. *River Research and Applications* **24**:789-803.

Hoagstrom CW, Brooks JE, Davenport SR. 2008b. Spatiotemporal population trends of *Notropis simus pecosensis* in relation to habitat conditions and the annual flow regime of the Pecos River, 1992-2005. *Copeia* **2008**:5-15.

Hoagstrom CW, Brooks JE, Davenport SR. 2011. A large-scale conservation perspective considering endemic fishes of the North American plains. *Biological Conservation* **144(1)**:21-34.

Hoagstrom CW, Zymonas ND, Davenport SR, Propst DL, Brooks JE. 2010. Rapid species replacements between fishes of the North American plains: a case history from the Pecos River. *Aquatic Invasions* **5(2)**:141-153.

Huggins DG, Moss RE. 1975. Fish population structure in altered and unaltered areas of a small Kansas stream. *Transactions of the Kansas Academy of Science* **77(1)**:18-30.

Jackson RG II. 1975. Hierarchical attributes and a unifying model of bed forms composed of cohesionless material and produced by shearing flow. *Geological Society of America Bulletin* **86**:1511-1522.

JMP©, Version 11 SAS institute Inc., Cary, NC, 1989-2007.

Jurajda P. 1995. Effects of channelization and regulation on fish recruitment in a flood plain river. *Regulated Rivers Research and Management* **10**:207-215.

Kempton RA, Taylor LR. 1974. Log-series and log-normal parameters as diversity discriminants for the Lepidoptera. *The Journal of Animal Ecology* **43**:381-399.

Kennedy TL, Turner TF. 2011. River channelization reduces nutrient flow and macroinvertebrate diversity at the aquatic terrestrial transition zone. *Ecosphere* **2(3)**:1-13.

King AJ. 2004. Density and distribution of potential prey for larval fish in the main channel of a floodplain river: pelagic versus epibenthic meiofauna. *River Research and Applications* **20**:883-897.

Kwak TJ, Peterson JT. 2007. Community indices, parameters, and comparisons. In: Guy CS, Brown ML. Analysis and interpretation of freshwater fisheries data. American Fisheries Society, Bethesda, Maryland. Pages 677-763.

Lepori F, Palm D, Brannas E, Malmqvist B. 2005. Does restoration of structural heterogeneity in streams enhance fish and macroinvertebrate diversity? *Ecological Applications* **15(6)**:2060-2071.

Matthews WJ, Hill LG. 1979. Influence of physico-chemical factors on habitat selection by red shiners, *Notropis lutrensis* (Pisces: Cyprinidae). *Copeia* **1979**:70-81.

Meffe GK, Sheldon AL. 1988. The influence of habitat structure on fish assemblage composition in Southeastern blackwater streams. *American Midland Naturalist* **120(2)**:225-240.

Morris LA, Langemeier RN, Russell TR, Witt A Jr. 1968. Effects of main stem impoundments and channelization upon the limnology of the Missouri River, Nebraska. Nebraska Game and Parks Commission Staff Research Publications Paper 10.

Muotka T, Paavola R, Haapala A, Novikmec M, Laasonen P. 2002. Long-term recovery of stream habitat structure and benthic invertebrate communities from in-stream restoration. *Biological Conservation* **105**:243-253.

Nakano D, Nakamura F. 2008. The significance of meandering channel morphology on the diversity and abundance of macroinvertebrates in a lowland river in Japan. Aquatic Conservation: *Marine and Freshwater Ecosystems* **18**:780–798.

Oscoz J, Leunda PM, Miranda R, Garcia-Fresca C, Campos F, Escala MC. 2005. River channelization effects on fish population structure in the Larraun River (Northern Spain). *Hydrobiologia* **543**:191-198.

Ostrand KG, Wilde GR. 2002. Seasonal and spatial variation in a prairie stream-fish assemblage. *Ecology of Freshwater Fish* **11**:137-149.

Ostrand KG, Wilde GR. 2004. Changes in prairie stream fish assemblages restricted to isolated streambed pools. *Transactions of the American Fisheries Society* **133**:1329-1338.

Palm D, Brännäs E, Lepori F, Nilsson K, Stridsman S. 2007. The influence of spawning habitat restoration on juvenile brown trout (*Salmo trutta*) density. *Canadian Journal of Fisheries and Aquatic Sciences* **64(3)**:509-515.

Palmer MA, Menninger HL, Bernhardt E. 2009. River restoration, habitat heterogeneity and biodiversity: a failure of theory or practice? *Freshwater Biology* **55(s1)**:205-222.

Paragamian VL. 1987. Standing stocks of fish in some Iowa streams, with a comparison of channelized and natural stream reaches in a southern Iowa drift plain. *Proceedings of the Iowa Academy of Science* **94(4)**:128-134.

Perkin JS, Gido KB, Cooper AR, Turner TF, Osborne MJ, Johnson ER, Mayes KB. 2015. Fragmentation and dewatering transform Great Plains stream fish communities. *Ecological Monographs* **85(1)**:73-92.

Perkin JS, Gido KB, Costigan KH, Daniels MD, Johnson ER. 2014. Fragmentation and drying ratchet down Great Plains stream fish diversity. *Aquatic Conservation: Marine and Freshwater Ecosystems* DOI: 10.1002/aqc.2501.

Polivka KM. 1999. The microhabitat distribution of the Arkansas River shiner, *Notropis girardi*: a habitat-mosaic approach. *Environmental Biology of Fishes* **55**:265-278.

Pretty JL, Harrison SSC, Shepherd DJ, Smith C, Hildrew AG, Hey RD. 2003. River rehabilitation and fish populations: assessing the benefit of instream structures. *Journal of Applied Ecology* **40(2)**:251-265.

Rahel FJ, Thel LA. (2004, March 18). Plains killifish (Fundulus zebrinus): a technical conservation assessment. USDA Forest Service, Rocky Mountain Region. Available: http://www.fs.fed.us/r2/projects/scp/assessments/plainskillifish.pdf.

Roberge M. 2002. Human modification of the geomorphically unstable Salt River in metropolitan Phoenix. *The Professional Geographer* **54(2)**: 175-189.

Shields FD Jr., Knight SS, Cooper CM. 1994. Effects of channel incision on base flow stream habitats and fishes. *Environmental Management* **18(1)**:43-57.

Steppen GR. 2000. Green River floodplain habitat restoration for endangered fish species. Proceedings of the Water Environment Federation, Watershed **2000(21)**:2081-2101.

USBOR (U.S. Bureau of Reclamation) (2009) Pecos River channel restoration at the Bitter Lake National Wildlife Refuge, Chaves County, New Mexico, Environmental Assessment. U.S. Bureau of Reclamation, Albuquerque Area Office, Albuquerque, New Mexico.

TABLES

Table I. Results of Analysis of Covariance (ANCOVA) or Analysis of Variance (ANOVA) tests comparing differences in rank abundance of common fish species among sites.

Source	df	*F*	*P*	Source	df	*F*	*P*
C. lutrensis				*G. affinis*			
Site	4	3.1	0.017	Site	4	1.6	0.165
Conductivity (µmhos)	1	11.2	0.001	*N. s. pecosensis*			
C. carpio				Site	4	1.4	0.220
Site	4	4.5	0.002	*F. zebrinus*			
Conductivity (µmhos)	1	14.1	≤ 0.001	Site	4	2.3	0.056

FIGURES

Figure 1. Map of the Pecos River, New Mexico. Stars and Roman numerals indicate sampling sites. Roman numeral I represents the braided site, II* represents the unchannelized site, III* represents the restored site, IV* represents the channelized site and V represents the ditched site. Distances between the unchannelized, restored and channelized sites within the Bitter Lake National Wildlife Refuge (i.e. sites II*, III* and IV*) were approximately 2 km apart and are encompassed by the refuge site star.

Figure 2. Mean daily discharge from the USGS Near Artesia NM gage (08396500). Panels A and B indicate mean daily discharge for our study period (2012-2013 black hydrograph) and historical mean daily discharge (1930-1931 gray hydrograph). Panel A is scaled to show historical peaks in discharge. Panel B is scaled to better illustrate discharge during the study period. The data from 1930-1931 was chosen as representative of mean daily discharge from a 20 year pre-dam period. In panel B, spaces between vertical hashed lines on the x-axis indicate sampling periods for 2012 and 2013.

Figure 3. Habitat availability by site and sampling trip. Panel A: symbols represent mean discharge by site and sampling trip and whiskers represent the standard error. Panel B: symbols indicate mean wetted width by site and sampling trip and whiskers represent the standard error.

Figure 4. Habitat diversity by site. Panel A: symbols represent Fisher’s α diversity values for depth measurements calculated by site. Error bars represent bootstrapped 95%

confidence intervals. Panel B: Symbols represent Fisher's α diversity values for velocity measurements calculated by site. All bootstrapped 95% confidence interval values were the same as the diversity values. Means not sharing the same letter are significantly different based on diversity permutation tests.

Figure 5. Rank abundance of common fish species by site. Open circles indicate ANCOVA adjusted means for *Cyprinella lutrensis* and *Cyprinus carpio* and means for *N. simus pecosensis*, *Gambusia affinis* and *Fundulus zebrinus*. Upper and lower whiskers indicate standard errors for all species. Means not sharing the same letter are significantly different (Tukey's HSD tests: $P < 0.05$). Means without letters did not exhibit any significant differences (ANCOVA site effect: $P > 0.05$).

Figure 6. Standard length (mm) distribution of common fish species by site. Results of separate Kruskal-Wallis tests including the Chi-Square test statistic (H) and the corresponding P-value (P) are indicated in the upper left of each graph. Boxplots indicate the median (middle line), 25th and 75th percentiles (top and bottom of box), 10th and 90th percentiles (upper and lower whiskers), and 5th and 95th percentiles (upper and lower dots). Boxplots not sharing the same letter are significantly different (Pairwise Wilcoxon signed-rank tests: $P < 0.05$). Boxplots without letters did not exhibit any significant differences. Note different y-axis scales for each species.

Figure 7. Non-metric multidimensional scaling ordination of fish assemblages. Centroids represent sampling trips for each site. Numbers indicate the sampling trip represented by each centroid, with 1 = June 2012, 2 = August 2012, 3 = June 2013 and 4 = August 2013.

Figure 8. Fish species diversity by site. Panel A: symbols represent Fisher's α diversity values for fish assemblages by site. All bootstrapped 95% confidence interval values were the same as the diversity values. Panel B: symbols represent rarefied species richness values for fish assemblages by site. Error bars represent the standard error taken as the square root of resampling variances for each value.

Figure 9. Canonical Correspondence Analysis ordination showing relationships of sites by sampling trip and fish species with mesohabitat environmental variables. Axis 1 explained approximately 55% of the variation in the ordination and axis 2 explained approximately 30% of the variation in the ordination. Numbers next to sites indicate sampling trips, with 1 = June 2012, 2 = August 2012, 3 = June 2013 and 4 = August 2013. Names of riverine minnow species are underlined.

Figure 10. Percent abundance of the three most common riverine minnow species (panel A) and the three most common river's-edge species (panel B) by sampling trip.

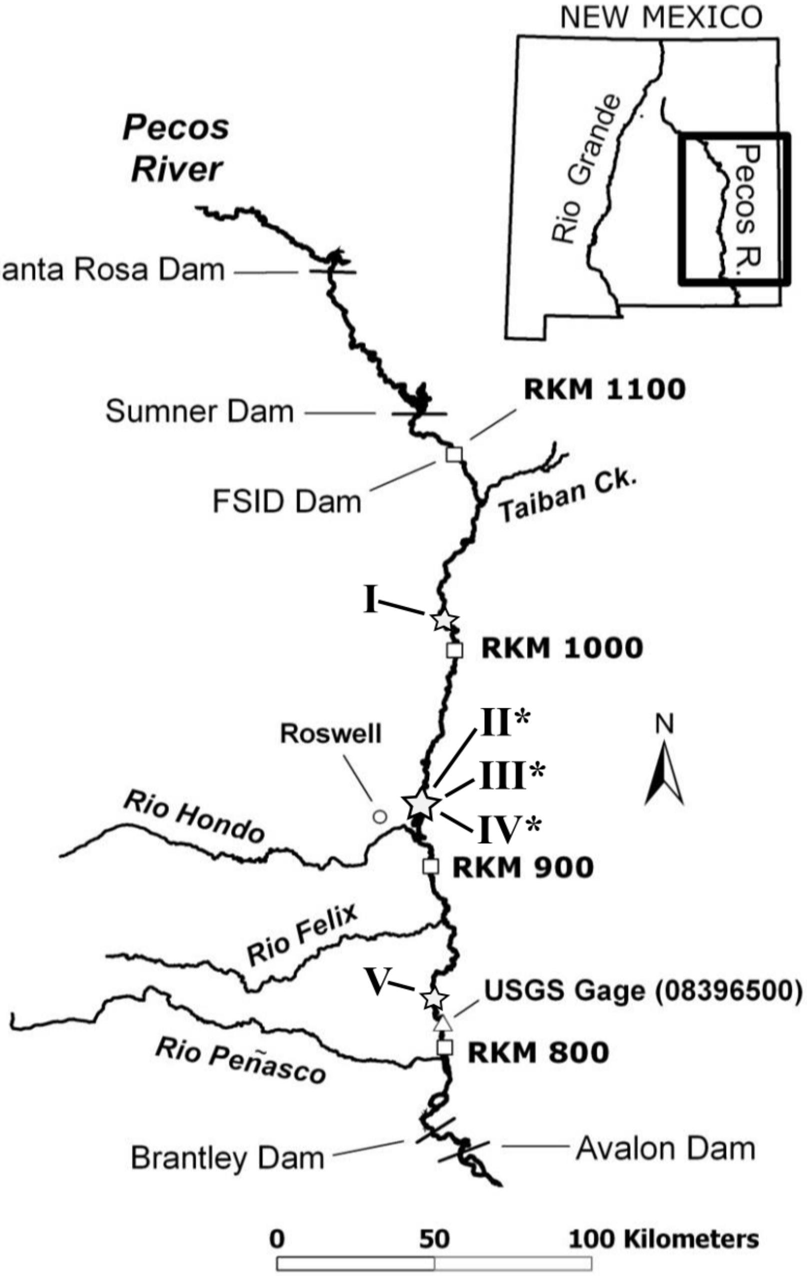
NEW MEXICO
Rio Grande
Pecos R.
Pecos River
Santa Rosa Dam
Sumner Dam
RKM 1100
FSID Dam
Taiban Ck.
I
RKM 1000
Roswell
II*
III*
IV*
N
Rio Hondo
RKM 900
Rio Felix
V
USGS Gage (08396500)
Rio Peñasco
RKM 800
Brantley Dam
Avalon Dam
0
50
100 Kilometers

Figure 1.

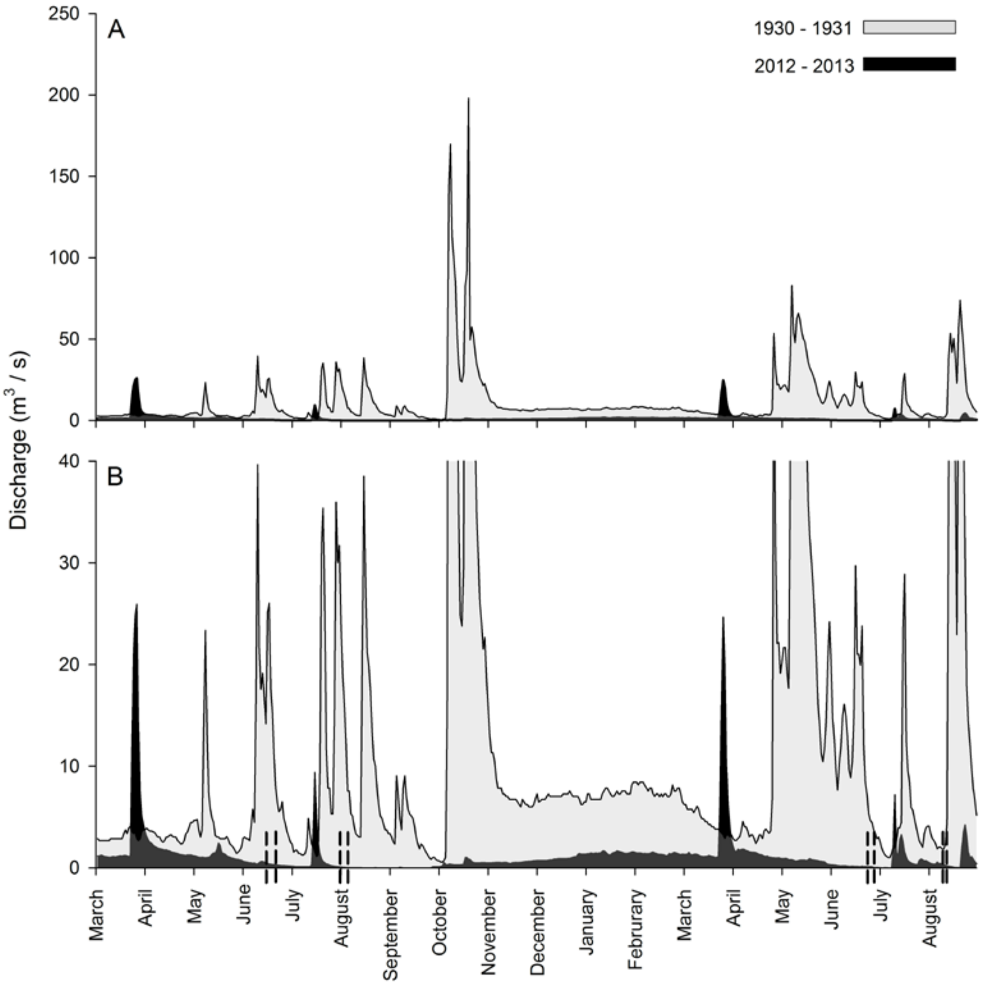
A
B
1930 - 1931
2012 - 2013
Discharge (m^3 / s)
250
200
150
100
50
0
40
30
20
10
0
March
April
May
June
July
August
September
October
November
December
January
Februrary
March
April
May
June
July
August

Figure 2.

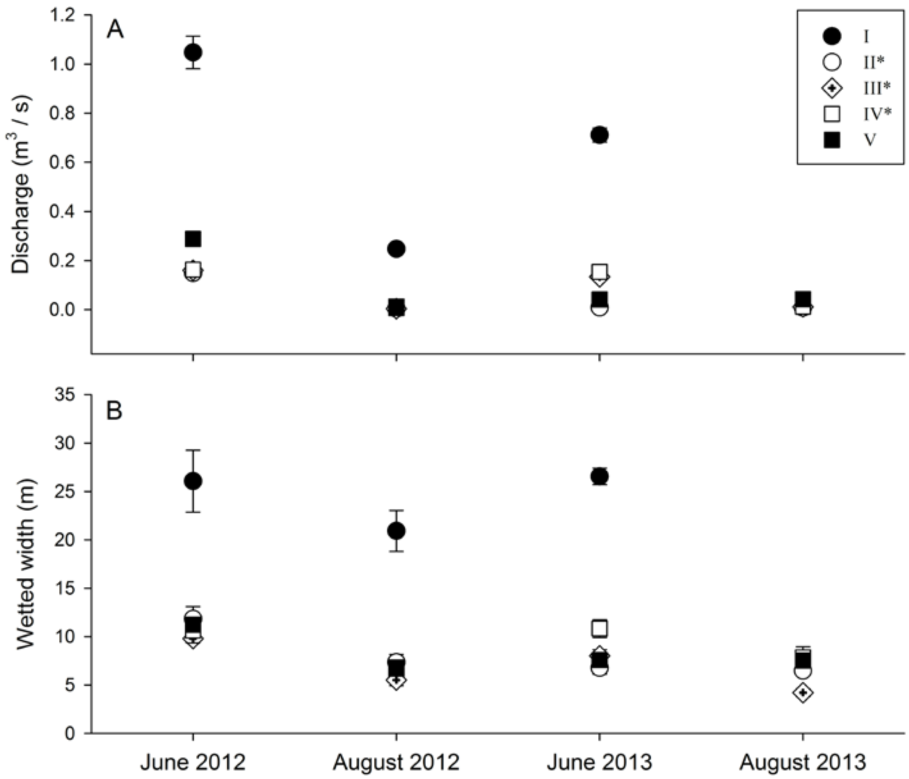
A
B
Discharge (m^3 / s)
Wetted width (m)
1.2
1.0
0.8
0.6
0.4
0.2
0.0
35
30
25
20
15
10
5
0
I
II*
III*
IV*
V
June 2012
August 2012
June 2013
August 2013

Figure 3.

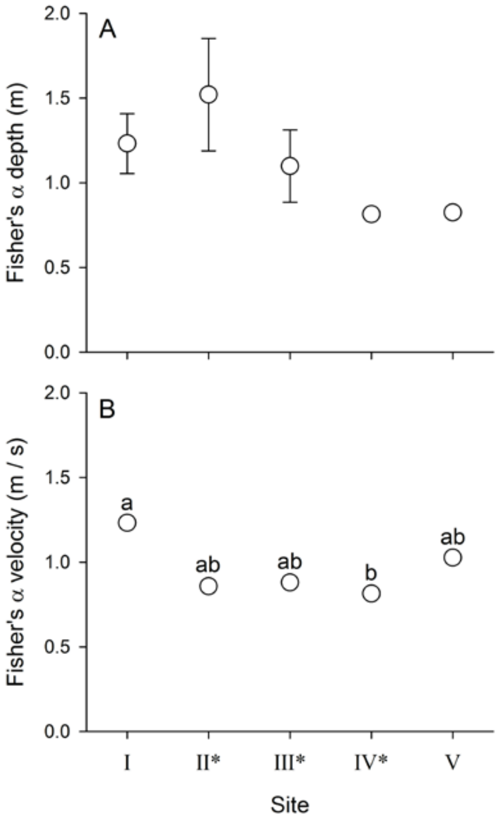
A
Fisher's α depth (m)
2.0
1.5
1.0
0.5
0.0
B
Fisher's α velocity (m / s)
2.0
1.5
1.0
0.5
0.0
a
ab
ab
b
ab
I
II*
III*
IV*
V
Site

Figure 4.

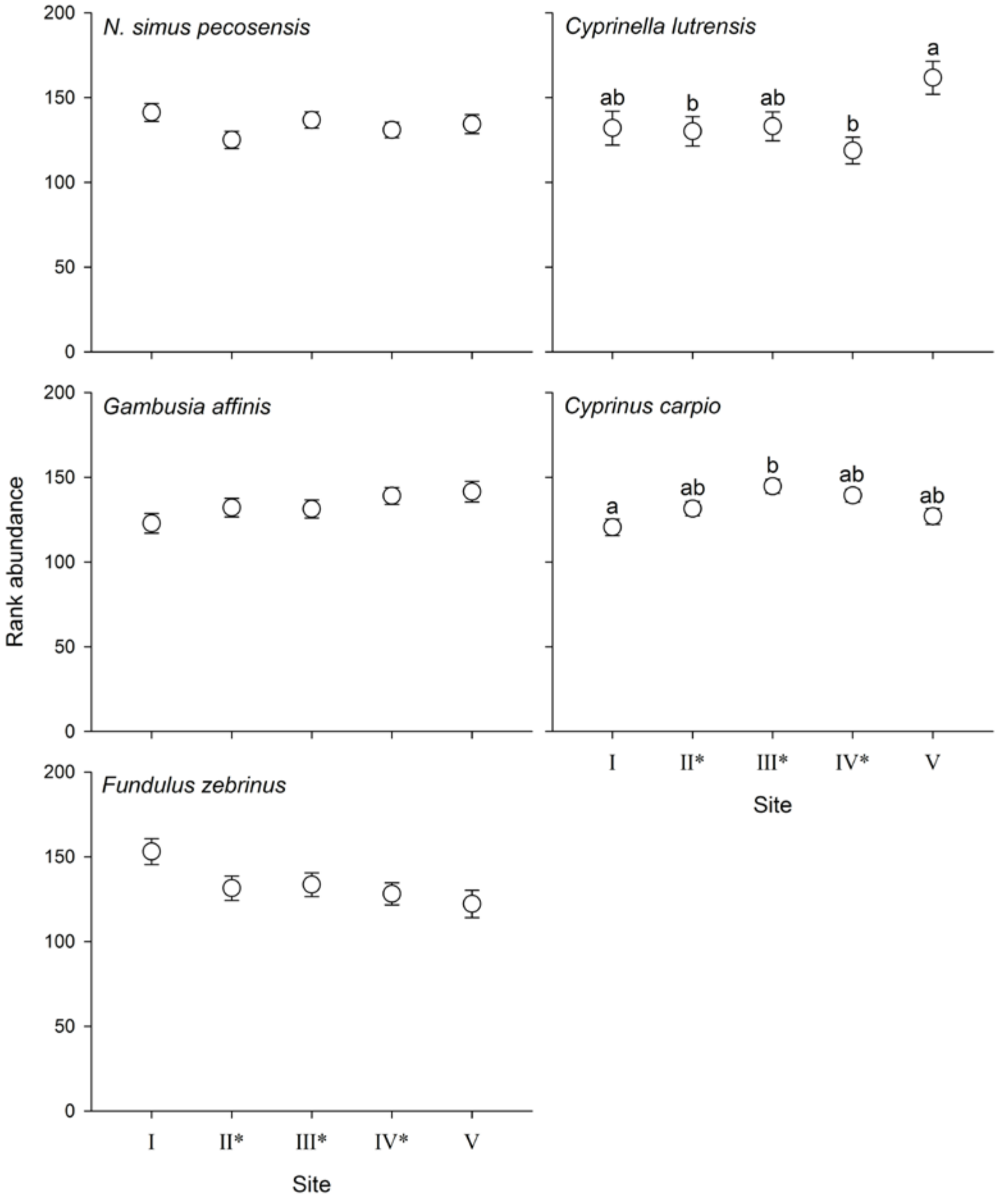
N. simus pecosensis
Cyprinella lutrensis
Gambusia affinis
Cyprinus carpio
Fundulus zebrinus
Rank abundance
200
150
100
50
0
ab
b
ab
b
a
a
ab
b
ab
ab
I
II*
III*
IV*
V
Site

Figure 5.

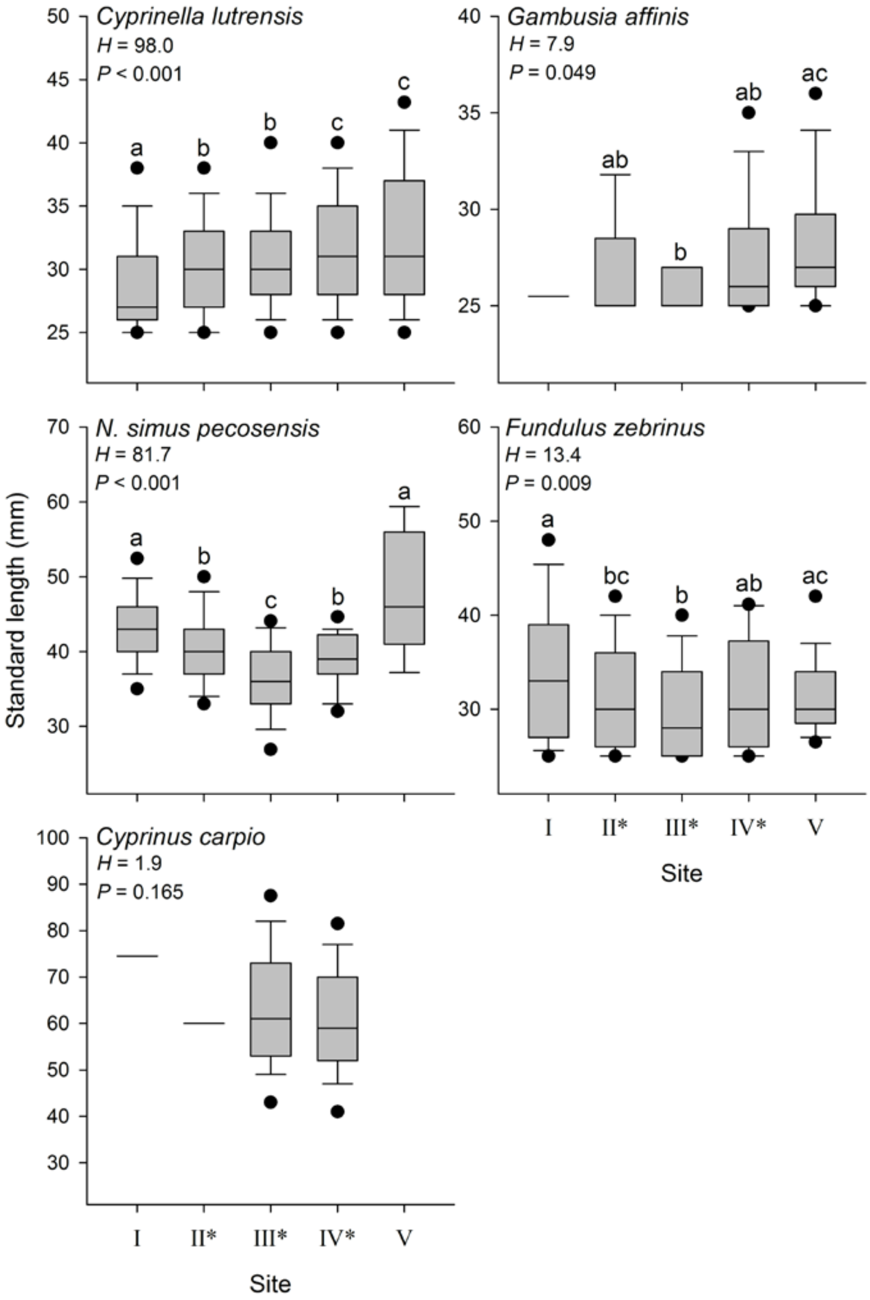
Cyprinella lutrensis
H = 98.0
P < 0.001
a
b
b
c
c
Gambusia affinis
H = 7.9
P = 0.049
ab
b
ab
ac
N. simus pecosensis
H = 81.7
P < 0.001
a
b
c
b
a
Fundulus zebrinus
H = 13.4
P = 0.009
a
bc
b
ab
ac
Cyprinus carpio
H = 1.9
P = 0.165
Standard length (mm)
I
II*
III*
IV*
V
Site

Figure 6.

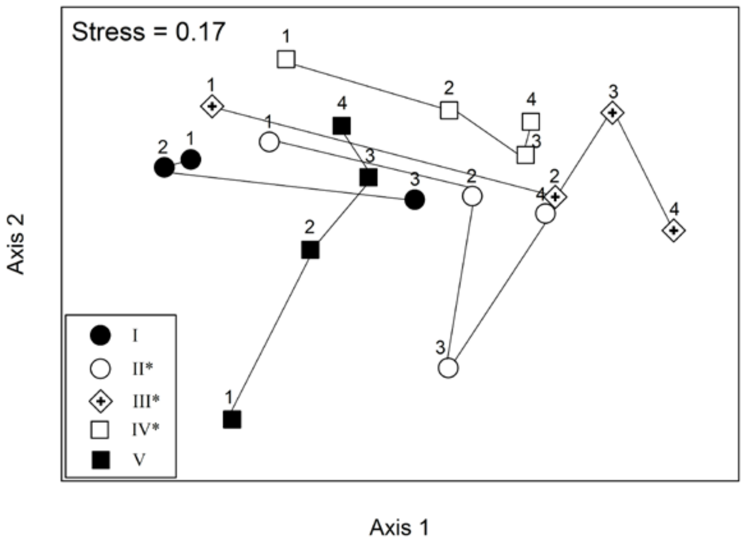

Figure 7.

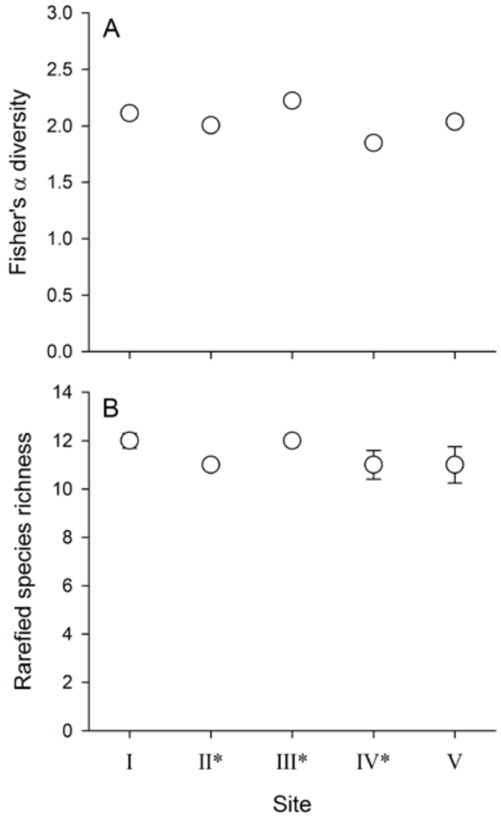
A
B
Fisher's α diversity
Rarefied species richness
3.0
2.5
2.0
1.5
1.0
0.5
0.0
14
12
10
8
6
4
2
0
I
II*
III*
IV*
V
Site

Figure 8.

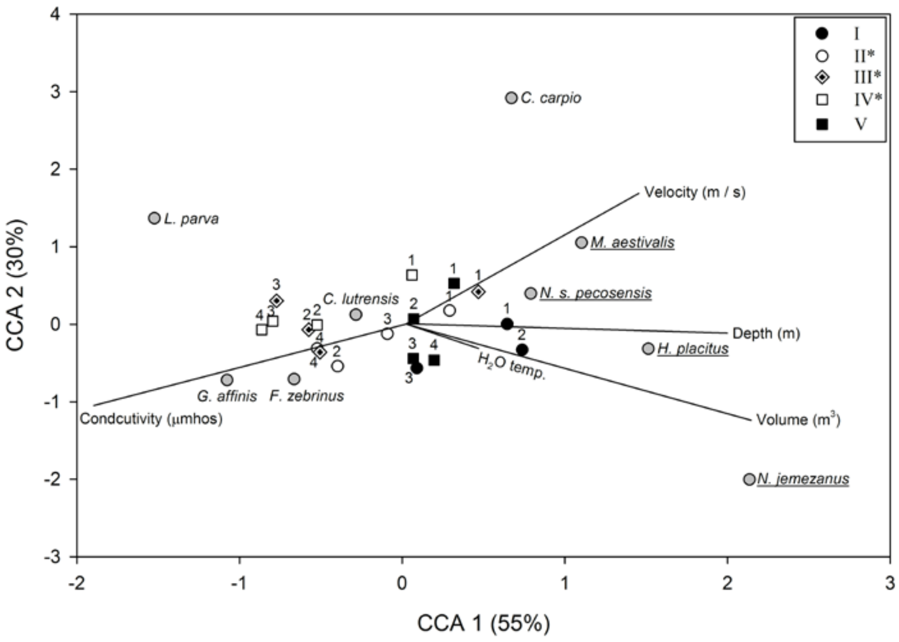
● I
○ II*
◈ III*
□ IV*
■ V
C. carpio
L. parva
Velocity (m / s)
M. aestivalis
N. s. pecosensis
C. lutrensis
Depth (m)
H. placitus
H_2O temp.
Volume (m^3)
G. affinis
F. zebrinus
Condcutivity (μmhos)
N. jemezanus
1
2
3
4
CCA 2 (30%)
CCA 1 (55%)
4
3
2
1
0
-1
-2
-3
-2
-1
0
1
2
3

Figure 9.

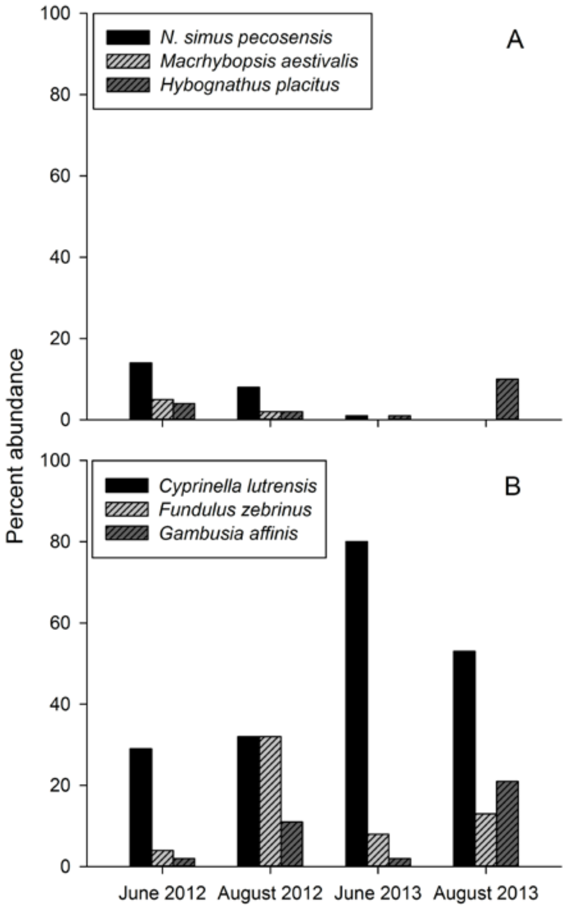
100
80
60
40
20
0
Percent abundance
N. simus pecosensis
Macrhybopsis aestivalis
Hybognathus placitus
A
Cyprinella lutrensis
Fundulus zebrinus
Gambusia affinis
B
June 2012
August 2012
June 2013
August 2013

Figure 10.

APPENDIX

Appendix I. Total density and percent abundance of fish collected by site and trip.

	Braided	Unchannelized	Restored	Channelized	Ditched
June 2012					
Total density (fish / 100 m^2)	869	1,985	5,751	10,734	1,329
N. s. pecosensis	15.4%	18.8%	27.6%	7.5%	0.0%
N. jemezanus	7.1%	0.0%	0.2%	0.0%	0.0%
M. aestivalis	4.2%	0.0%	14.3%	0.4%	16.2%
H. placitus	9.0%	21.6%	4.3%	0.9%	0.0%
N. girardi	3.5%	0.2%	0.2%	0.2%	0.0%
C. carpio	1.3%	1.0%	27.0%	60.0%	0.0%
C. lutrensis	29.7%	48.3%	21.0%	23.1%	82.6%
C. pecosensis	0.0%	0.0%	0.7%	0.5%	0.0%
P. promelas	0.0%	0.2%	0.2%	1.0%	1.1%
G. affinis	0.0%	0.3%	1.1%	3.5%	0.0%
F. zebrinus	28.9%	9.5%	3.5%	1.8%	0.0%
L. parva	0.0%	0.0%	0.0%	1.1%	0.0%
I. punctatus	0.9%	0.0%	0.0%	0.0%	0.0%
August 2012					
Total density (fish / 100 m^2)	1,670	1,788	192	1,044	899
N. s. pecosensis	13.4%	0.0%	0.0%	14.0%	8.9%
N. jemezanus	23.5%	0.2%	0.0%	0.0%	0.0%
M. aestivalis	4.6%	0.0%	2.3%	0.0%	1.2%
H. placitus	5.8%	0.0%	0.0%	0.0%	0.2%
N. girardi	7.6%	0.0%	0.0%	0.0%	0.0%
N. stramineus	3.5%	0.0%	0.0%	0.0%	0.0%
C. lutrensis	9.4%	32.1%	30.7%	22.9%	85.1%
C. pecosensis	0.0%	0.7%	0.0%	0.0%	0.0%
C. carpoides	4.8%	0.0%	0.0%	0.0%	0.2%
P. promelas	0.0%	0.0%	3.8%	0.0%	0.0%
G. affinis	0.0%	3.5%	24.8%	46.3%	2.4%
F. zebrinus	27.3%	63.4%	35.9%	13.0%	0.5%
L. parva	0.0%	0.0%	2.6%	3.9%	0.0%
M. chrysops	0.0%	0.0%	0.0%	0.0%	1.4%

Appendix I cont.

	Braided	Unchannelized	Restored	Channelized	Ditched
June 2013					
Total density (fish / 100 m^2)	772	357	933	1,138	1,486
N. s. pecosensis	0.0%	0.0%	0.0%	0.0%	4.1%
N. jemezanus	1.8%	0.0%	0.0%	0.0%	1.8%
H. placitus	3.6%	1.5%	0.0%	0.0%	0.0%
N. girardi	6.6%	0.0%	0.0%	0.0%	0.5%
N. stramineus	1.7%	0.0%	0.0%	0.0%	0.0%
C. lutrensis	66.7%	97.1%	80.4%	83.6%	78.3%
C. pecosensis	0.0%	0.0%	0.6%	0.0%	0.0%
C. carpoides	0.9%	0.0%	0.0%	0.0%	1.4%
P. promelas	0.0%	0.0%	0.0%	0.0%	2.7%
G. affinis	6.1%	1.5%	0.0%	3.4%	1.8%
F. zebrinus	12.7%	0.0%	5.2%	7.2%	9.5%
L. parva	0.0%	0.0%	13.7%	5.8%	0.0%
August 2013					
Total density (fish / 100 m^2)		360	195	1,405	3,721
N. s. pecosensis	NA	0.0%	0.0%	0.0%	0.1%
N. jemezanus	NA	0.0%	0.0%	0.0%	0.4%
H. placitus	NA	1.2%	0.0%	0.0%	14.8%
C. lutrensis	NA	7.2%	19.2%	23.4%	70.5%
C. pecosensis	NA	16.8%	0.0%	1.6%	0.1%
P. promelas	NA	0.0%	0.0%	0.0%	0.1%
G. affinis	NA	11.5%	0.0%	45.5%	13.0%
F. zebrinus	NA	62.6%	80.8%	21.8%	1.0%
L. parva	NA	0.8%	0.0%	7.8%	0.0%

CHAPTER 2: HOW CHANNELIZATION, RESTORATION AND DEWATERING AFFECT SLACKWATER FAUNAS ALONG A SAND-BED RIVER

This chapter was co-authored by Christopher W. Hoagstrom and Brian D. S. Graeb. It is formatted for The Prairie Naturalist

ABSTRACT Slackwater habitats function as nursery areas for early-life-stage fishes and are critical for their growth and survival. Channelization can reduce slackwater availability by reducing channel complexity. Because of this, river channel restoration efforts have become more common and have been utilized to increase slackwater availability. We compared slackwater habitat conditions and early-juvenile fish and meiofauna assemblage characteristics of a recently restored reach of the Pecos River, New Mexico with unchannelized and channelized reaches. We also assessed the relationship between flow regime and estimated hatch dates of common fish species including red shiner (*Cyrpinella lutrensis*), plains killifish (*Fundulus zebrinus*) and western mosquitofish (*Gambusia affinis*). Slackwater availability and extent were greatest in unchannelized reaches. Plains killifish were most abundant in unchannelized reaches. Western mosquitofish were most abundant in channelized reaches. Fish assemblages of the channelized reach differed from all other reaches. The common fish species spawned and hatched during periods of lower discharge. Meiofauna assemblages of the more perennial channelized reach differed from those of the unchannelized reach and restored reach. Abundances of total meiofauna, Ostracoda, Nematoda and Turbellaria were highest in the more perennial channelized reach. Rotifera were most abundant in the more perennial unchannelized reach. We suggest that future restoration efforts should focus on maintaining sufficient base flows in addition to physical channel restoration. Maintenance of base flows contributes to the provision of more extensive and abundant slackwater nursery areas within the restored river channel conducive to the growth and survival of early-juvenile fishes.

KEY WORDS Channelization, restoration, meiofauna, early-juvenile, slackwater, nursery

Riverine ecosystems contain a diverse mosaic of aquatic habitat types (Thorp et al. 2006). Within this mosaic, slackwater habitats (areas of little to no velocity) provide areas of refuge for early-life-stage fishes (Humphries et al. 2006, Pease et al. 2006). These habitats function as ideal nursery areas for early-life-stage fishes because early-life-stage fishes have limited powers of mobility (Mann and Bass 1997) and limited energy reserves (Wieser 1991), which renders them unable to exist and develop in higher velocity fluvial habitats (Schiemer et al. 2001). Additionally, slackwater habitats are often characterized by warm temperatures, low turbidity levels, and high nutrient concentrations (Humphries et al. 1999, Hoagstrom and Turner 2013). This combination of conditions makes them optimal for the growth of early-life-stage fishes and for high levels of primary production, which leads to the provision of abundant amounts of appropriately sized (200-500 μm) prey items (i.e. zooplankton) (Humphries et al. 1999, Nunn et al. 2007), which is critical for growth and survival of developing fish (Werner and Blaxter 1980).

Flow regime and channel morphology play a central role in the formation of slackwater habitats. In unregulated reaches of rivers, elevated stream flow events structure habitat via transportation and deposition of sediments within the river channel (Hoagstrom and Turner 2013). During intermediate flows, channel braiding is increased and abundant sand bars formed during high flow periods create diverse flow environments that provide numerous slackwater areas (Moore and Thorp 2008). Subsequent low flow periods then serve to increase temperatures and concentrate prey items within existing slackwater habitats (Humphries et al. 1999). Channel morphology functions synergistically with flow regime in the creation of slackwater nursery areas. Unmodified river channels are often wide and complex and provide a variety of areas for slackwaters to form at different levels of discharge (Price et al. 2013, Vietz et al. 2013). Hence, unaltered river channels in combination with a natural flow regime likely provide ideal conditions for the production of potential prey items and recruitment of fishes (Humphries et al. 1999, Moore and Thorp 2008, Hoagstrom and Turner 2013).

Altered flow regimes and channelization can reduce the provision and persistence of slackwater habitats (Morris et al. 1968, Vietz et al. 2013). Flow regulation reduces sediment load, which narrows and incises the river channel (Kondolf 1997). This leads to a reduction of sand bars and other mesoforms which direct flows and create slackwater areas (Moore and Thorp 2008, Hoagstrom and Turner 2013). As a consequence, regulated flow regimes reduce total slackwater area as well as the area of individual slackwater habitat patches (Vietz et al. 2013). Channelization reduces the complexity of channel morphology (Shields et al. 1994, Lau et al. 2006). This reduces slackwater availability during base flows (Hoagstrom and Turner 2013). Further, because channelization confines the river within its banks, the potential for overbank flooding and the formation of slackwater habitat within the floodplain is reduced or eliminated at higher discharges (Jurajda 1995, Vietz et al. 2013). Reduction of slackwater nursery area negatively impacts recruitment of fishes (Jurajda 1995) and can reduce the diversity and abundance of potential prey items (Ning et al. 2010).

In the Pecos River, NM, a sand-bed river on the plains, altered flow regimes and channelization have been shown to negatively impact fish species occupying the river via displacement of eggs and larvae and through the reduction of favorable habitat conditions (Hoagstrom and Brooks 2005, Dudley and Platania 2007, Hoagstrom et al. 2008a, 2008b, Hoagstrom et al. 2010). Based on these observations, channel restoration has been suggested as a method that could potentially benefit fish assemblages by providing a more complex river channel similar to unchannelized reaches upstream (Hoagstrom and Brooks 1999, Hoagstrom et al. 2008b). A more complex channel will likely result in greater availability and extent of slackwater nursery areas conducive to recruitment of fishes and the production of potential prey items.

The main goal of this study was to assess if a recent channel restoration effort improved conditions for recruitment of fishes of the Pecos River, NM. An additional focus of the study nested within this goal was to take a novel approach in assessing the availability of potential prey items within sand bed rivers on the plains. In slackwater nursery areas of an Australian sand bed river, King et al. (2004) found that epibenthic meiofauna are an abundant prey source for early-life-stage fishes that often outnumber pelagic zooplankton considerably. The study by King et al. (2004) represented a paradigm shift in assessing prey availability for early-life-stage fishes in sand bed rivers, as the majority of prior studies have focused almost exclusively on pelagic zooplankton as a prey source (e.g. Ferrari et al. 1989, Thorp et al. 1994). We chose to evaluate the availability of epibenthic meiofauna as a potential prey source within the Pecos River due to the shallow ephemeral nature of many slackwaters within the river, which likely limits habitat for pelagic prey sources. To our knowledge this is the first evaluation of this prey source within rivers on the plains.

Our objectives were focused on comparisons among river reaches (unchannelized, restored, and channelized). Specifically, our objectives were: (1) compare the availability and extent of slackwater habitat among reaches; (2) compare abundance and assemblage composition of early-juvenile fishes and potential prey items (i.e. meiofauna) among reaches; and (3) assess relationships between flow regime and recruitment (presence and abundance) of early-juvenile fishes. Our hypotheses were: (1) the availability and extent of slackwater habitat will be greater in the restored reach and unchannelized reaches than in channelized reaches; (2) the abundance of early-juvenile fishes will be greater in the restored reach and unchannelized reaches than in channelized reaches and that assemblages of both fishes and meiofauna will be most similar between the restored reach and unchannelized reaches; and (3) recruitment of fishes will coincide with low flow periods, which create slackwater nurseries characterized by warm temperatures and abundant prey items.

STUDY AREA

The study area is located approximately 14.5 km northeast of Roswell New Mexico within the middle tract of the Bitter Lake National Wildlife Refuge. The river upstream of the middle tract of the refuge is unchannelized. The river downstream of the middle tract of the refuge is deeply incised due primarily to mechanical channelization and the stabilization of its banks by non-native salt cedar (*Tamarix* spp.).

Restoration efforts took place at the point where the river transitions from an unchannelized river to a channelized river within the middle tract of the Bitter Lake NWR. Here, the river was diverted away from the point of channelization into an historic oxbow. Non-native salt cedar was removed and banks were lowered and destabilized

(USBOR 2009). This was done to lengthen the river, create a wider more dynamic river channel, and to reestablish connection with the floodplain. This effort was carried out in accordance with the goals of the restoration project. One of which was to create habitat conditions more favorable for fishes and other aquatic species occupying the river (USBOR 2009).

Five study sites were used to assess the effects of the restoration effort on slackwater habitat conditions, recruitment of early-juvenile fishes, and the provision of potential prey items (i.e. meiofauna). Three sites were located within the Bitter Lake NWR (Fig. 1). These include the recently restored site, an unchannelized site approximately 2 km upstream from the restored site and a channelized site approximately 2 km downstream from the restored site. Remote sites were also included in the study because the proximity of sites within the Bitter Lake NWR could mask biological and habitat differences. These included an unchannelized site 63 km north of the restored site hereafter referred to as the “braided” site and a channelized site 60 km south of the restored site located within the William S. Huey Wildlife Area hereafter referred to as the “ditched” site (Fig. 1). Both remote sites are more perennial in nature than the refuge sites.

METHODS

Field work took place over a two-year time period (2012-2013). Sampling was conducted in both June and August of each year. We attempted to sample during stable base flow conditions, when flows were produced mainly from groundwater seepage into the channel. We avoided periods of flooding and intermittence as much as possible to facilitate comparisons among sampling trips (Kwak and Peterson 2007). However, flooding prevented sampling at the braided site in August 2013. Sampling was completed within one to two weeks each trip to minimize temporal differences among sites. All data were collected from a sampling area of two meanders (two riffle-pool sequences) representative of the variety of slackwater mesohabitats present within each site. Because of the disruptive nature of the sampling methods that were employed, sampling proceeded from downstream to upstream within the two meander sampling area, one slackwater mesohabitat at a time

All slackwater mesohabitats within the two meander sampling area were sampled for early-juvenile fishes and meiofauna. Fish were sampled using a 3.0-m seine with 1.6-mm mesh. All fish collected were identified in the field if possible. Because some smaller individuals were difficult to identify in the field, these specimens were preserved in 10% formalin and were later transferred to 70% ETOH and identified in the laboratory. Meiofauna were collected with an epibenthic corer made of a 12-cm tall, 4.4-cm diameter section of PVC pipe with an additional piece of PVC pipe glued around the outside of the corer 1 cm from the bottom (King 2004). Three representative subsamples were taken from each slackwater mesohabitat within the sampling area by pushing the corer 1cm into the sediment, sealing the top with a PVC cap and sliding a paint scraper with a rubberized surface underneath the corer (King 2004). Each sample was emptied into an 80-µm mesh plankton net, filtered in the river and preserved with 95% ETOH. All meiofauna from core samples were identified in the laboratory. After fish and meiofauna were collected, the dimensions of each slackwater mesohabitat sampled were recorded including maximum depth, maximum width and maximum length and temperature and conductivity were measured using a thermometer and YSI meter. All slackwater mesohabitats sampled were also photographed, given a brief fluvial-geomorphological description and named using nomenclature modified from Polivka (1999) and King (2004).

Meiofauna samples were processed in the lab following King (2004). Each sample was poured in a bucket and a combination of stirring and swirling with the addition of water was used to suspend meiofauna and organic matter. The suspended material was then decanted into a 74-µm mesh sieve. This process was repeated until no organic matter or invertebrates could be seen in the decanted water. The meiofauna and other organic material in the sieve was then washed into a jar and preserved with 95% ETOH. The remaining sediment was also preserved with 95% ETOH and later checked to ensure no meiofauna remained.

Enumeration and identification of meiofauna followed Nunn et al. (2007). The content of each jar was poured into a beaker and the volume of the beaker was increased to 100 ml with 95% ETOH. The beaker was stirred and three 500-µl subsamples were removed from the beaker using a calibrated pipette. Each subsample was examined individually in a Sedgwick-Rafter counting cell, where all meiofauna were counted and identified to the lowest practical taxonomic level with the aid of a compound microscope.

Analyses

Only fish ≤ 25 mm SL were included in analyses because the main focus of this study was early-juvenile fishes, their potential prey items and habitat/flow regime conditions conducive to their recruitment. This length cutoff was chosen based on the literature describing the life history of fishes that occupy the river (Krumholz 1948, Yildrim and Peters 2006, Minckley and Klaassen 1969), which indicated that the majority of fish ≤ 25 mm SL are young-of-year individuals that are primarily confined to slackwater nursery areas.

We included two covariates (mesohabitat volume (m^3) and conductivity (µmhos)) in our analyses assessing spatial differences in early-juvenile fish and meiofauna abundance, which was rank-transformed due to non-normality. These covariates were included in the analyses because they have been shown to affect both plains stream fish assemblages (Ostrand and Wilde 2004, Hoagstrom 2009) and meiofauna assemblages (Pillay and Perissinotto 2009). We used a Spearman's ρ test to assess correlation between our covariates and found that Spearman's ρ values ranged from -0.30 to 0.18, indicating that the covariates were not highly correlated and could be retained in analyses.

Nursery extent (i.e. area (m^2)), which was also rank-transformed prior to analysis, was compared among sites using analysis of variance (ANOVA). If a significant effect of site was observed, a Tukey's HSD (Honestly Significant Difference) test was used to test which sites differed significantly from each other with regard to nursery extent. Nursery availability was assessed qualitatively by comparing the mean number of nursery's available per meander among sites.

Rank abundance of common early-juvenile fishes and meiofauna was compared among sites using separate ANCOVA's. If a significant effect of site was observed, Tukey's HSD tests were used to test which sites differed significantly from each other with regard to specific taxa.

Assemblage composition of both early-juvenile fish and meiofauna was compared among sites by trip using non-metric multidimensional scaling (NMS) with Bray-Curtis dissimilarity used as the resemblance measure. We removed species/taxonomic groups that comprised less than 5% of the total assemblage for both early-juvenile fish and meiofauna. This was done to focus on those species/taxonomic groups that were most likely driving differences in assemblage structure among sites (Marchant 2002). For each ordination, we considered stress values above 0.25 to be uninterpretable (e.g. Clark and Warwick 2001). Hence, if stress below 0.25 was not achieved in two dimensions, a three dimensional ordination was utilized. Separate permutational multivariate analyses of variance (PERMANOVA) were then conducted (one for early-juvenile fish and one for meiofauna) to assess whether spatial patterns depicted in the ordinations were significant. We were unable to test for differences among sampling trips with PERMANOVA due to a lack of degrees of freedom. We used 9,999 permutations for both PERMANOVA analyses.

Spawning phenologies of common species were estimated using length data for each species combined with length-at-hatch and growth rate data obtained from the literature (Krumholz 1948, Saksena 1962, Yildrim and Peters 2006). Estimates of hatching initiation and duration were then plotted against daily discharge values to assess relationships between hatching and discharge for each species. Species specific length-at-hatch and growth rates were used for all species with the exception of plains killifish, where the length-at-hatch and growth rate of mummichog (*Fundulus heteroclitus*) was used as a surrogate measure (Marteinsdottir and Able 1992, Kneib 1993).

All analyses used to compare slackwater nursery characteristics and abundance of both fish and meiofauna were conducted in JMP 11 (SAS Institute 1989-2007). All community analyses were conducted in program PAST (Version 3.04) (Hammer et al. 2001).

RESULTS

Habitat conditions

A total of 120 slackwater habitats was sampled. Slackwater extent, defined as ranked area varied among sites ($F = 7.9$, $P \leq 0.001$), with the greatest amount of slackwater area found in the braided site (Fig. 2A). Slackwater availability, defined as the number of slackwaters per meander, was greatest in the braided site and lowest in the ditched site (Fig. 2B). An overall trend of greater slackwater extent and availability in unchannelized sites compared to all other sites was observed, with the restored site showing a trend of lower extent and availability than nearly all other sites.

Fish

A total of 7,257 early-juvenile fishes (≤ 25 mm SL) was collected. The fish assemblage was dominated by western mosquitofish (*Gambusia affinis*), red shiner (*Cyprinella lutrensis*), plains killifish (*Fundulus zebrinus*) and Pecos pupfish (*Cyprinodon pecosensis*). These species comprised 95% of all fish sampled (Appendix I).

Site scale differences in rank abundance of early-juvenile fish were observed for two common species (i.e. species comprising ≥ 5% of the total assemblage), the western mosquitofish and plains killifish (Table 1; Fig. 3). The abundance of western mosquitofish was lower in the unchannelized sites than in the channelized sites and the abundance of plains killifish was higher in the unchannelized sites than in the ditched site. While no site scale differences in abundance were observed for Pecos pupfish or red shiner (Table 1; Fig. 3), the abundance of red shiner showed a negative relationship with higher levels of conductivity (µmhos) (Table 1).

Fish assemblages were similar among sites and exhibited considerable overlap with the exception of the channelized site, which separated the most from all other sites in the ordination (Fig. 4A). While no collective pattern in assemblage shifts among sampling trips was discernable, assemblage shifts were present within each site, with assemblages of the channelized site changing the least through time (Fig. 4A). While a lack of degrees of freedom did not allow for comparisons of assemblage structure among sampling trips with PERMANOVA, site-scale differences observed in the ordination were confirmed with PERMANOVA ($F = 2.9$ $P = 0.001$) and post-hoc pairwise testing for specific site differences indicated that assemblages of the channelized site differed from all other sites.

Hatch dates were estimated for 2012 and 2013 (Fig. 5) using length data from three of the most common species sampled (i.e. red shiner, western mosquitofish and plains killifish). The majority of individuals of all species likely spawned and hatched during periods of low discharge. Western mosquitofish exhibited the most protracted spawning of the three species.

Meiofauna

A total of 4,735 meiofauna was collected. The most abundant taxonomic groups collected were Ostracoda (24%), Rotifera (24%), and Nematoda (23%). Other major contributing taxa included Turbellaria (6%), early-instar Chironomidae (6%), and early-instar Ceratopogonidae (5%) (Appendix II).

Rank abundance of the total meiofauna assemblage differed among sites (Table 2) and was highest in the ditched site (Fig. 6). Separate ANCOVA’s comparing rank abundance among sites for each of the common taxa (i.e. those taxa comprising ≥ 5% of the total assemblage) revealed spatial patterns in abundance for Ostracoda, Nematoda, Rotifera, and Turbellaria (Table 2; Fig. 6). Ostracoda abundance was lowest in both unchannelized sites, higher in the restored site and channelized site, and highest in the ditched site. Nematoda abundance was higher in the ditched site than in all refuge sites. Nematoda abundance also exhibited a positive relationship with high levels of conductivity (µmhos) (Table 2). Rotifera abundance was higher in the braided site in comparison to all other sites with the exception of the ditched site. Turbellaria abundance was higher in the ditched site than in all other sites. No site scale differences in abundance were found for Chironomidae and Ceratopogonidae, but both exhibited a negative relationship with high levels of conductivity (Table 2).

Like fish assemblages, meiofauna assemblages showed similarity among sites (Fig. 4B). The most noticeable difference in meiofauna assemblage composition was that assemblages of the ditched site separated from all other sites. Again, no collective pattern in assemblage shifts among sampling trips was discernable, although assemblages of all sites showed shifts in assemblage structure through time. A PERMANOVA test confirmed site scale differences in assemblage composition (F = 2.3, P = 0.009) and pairwise comparisons of sites indicated that assemblages of the ditched site differed from those of the unchannelized site and the restored site.

DISCUSSION

The effects of channelization: biotic change?

Although slackwater area was only greater in the braided site in comparison to the channelized sites, the trend observed was that more extensive nursery area was available in unchannelized sites. In addition, slackwater frequency tended to be higher in the unchannelized sites with the exception of the channelized refuge site. These findings supported our hypotheses, which are also supported by other studies conducted in rivers on the plains (e.g. Morris et al. 1968, O'Neill and Thorp 2011) and in other lowland rivers (e.g. Jurajda 1995, Price et al. 2013), which suggest that increased river channel complexity results in the provision of more refuge or slackwater areas and that channelization decreases channel complexity and slackwater extent and availability. Hence, the different channel morphologies of our study sites were likely responsible for the patterns we saw in area and availability of slackwater nurseries.

Differences in the abundance of early-juvenile fishes between unchannelized and channelized river reaches were apparent in our study. This finding is consistent with other studies which show that channelization can negatively impact the recruitment of a variety of fish species. For example, Jurajda (1995) showed that channelization negatively impacted the recruitment of a suite of cyprinid species in the River Morava, Czech Republic, and Copp (1990) showed that in comparison to other unregulated rivers, channelized reaches of the River Great Ouse, East Anglia (U.K.) exhibited a reduction in recruitment of cyprinids. Similar patterns were evident in our study for plains killifish, which is more characteristic of shallower higher velocity habitats than the other species we collected (Rahel and Thel 2004). These habitats are more prevalent in unchannelized reaches of the river (Hoagstrom et al. 2008a) and our findings suggest higher levels of recruitment in these reaches for this species in comparison to channelized reaches.

Differences in the abundance of meiofauna between unchannelized and channelized reaches were noticeable for several of the most abundant taxa including Rotifera, Ostracoda and Nematoda. Rotifera were most prevalent in the braided reach. Assemblages of Rotifera have been shown to be more diverse and to occur in greater abundance in coarse sediments (Ricci and Balsamo 2000). The braided reach has the most coarse sediment and greatest variety of sediment sizes of all reaches that we sampled (personal observation), whereas the more channelized reaches have finer sediments and the substrate is more armored (Hoagstrom et al. 2005). This could be one of the factors explaining the patterns we observed in Rotifer abundance. Rotifera are the most favorable food source available for early-juvenile fishes that we sampled (Theilacker and McMaster 1971, King 2004, Nunn et al. 2007) and the abundance of Rotifera in the braided reach is likely an additional indication of quality habitat.

Ostracoda were more prevalent in channelized reaches compared to unchannelized reaches, with the greatest abundance occurring in the ditched site. Thorp and Covich (2009) show that lotic habitats often contain less speciose assemblages of Ostracoda than lentic habitats. This may explain the high abundance of Ostracoda in the ditched site, as this site was the most lentic in nature and contained the most standing water of all sites sampled (personal observation). Studies have shown that early-life stage fishes prey upon Ostracoda (Tito de Morias and Bodiou 1984, Roca et al. 1993). However, Aarnio and Bonsdorff (1997) showed that high percentages of Ostracoda are able to survive gut passage in juvenile fishes. Hence, despite their abundance within channelized reaches, Ostracoda may not be as favorable of a prey item as softer bodied prey for early-juvenile fishes due to their ability to pass through the gut undigested.

Nematoda abundance was high in the ditched site. Bott and Kaplan 1989 showed that Nematodes of a Piedmont stream exhibited higher densities in fine sediments than in coarse sediments. This may explain why more nematodes occurred in these reaches where more fine sediments are present (Hoagstrom et al. 2005). Spieth et al. (2010)

showed that Nematoda can be an important prey source for some early-life-stage fishes. Based on this information, Nematoda are potentially a more important prey item in channelized reaches than are Ostracoda due to their soft-bodied nature which may make them more digestible than Ostracoda.

Channel restoration vs. flow regime

Our results suggest that channel restoration did not improve nursery conditions (Fig. 2). In turn, the restoration effort appeared to have little effect on the recruitment of early-juvenile fishes and the production of potential prey items. None of the fish species we collected were more abundant in the restored site than in the channelized sites (Fig. 3) and prey production was similar to or lower than in channelized sites (Fig. 6). However, we do not conclude that channel restoration was a failure. We think that our inability to detect both abiotic and biotic differences in the restored site was likely due to confounding factors, the foremost of which was that discharge during our study period was extremely low in comparison to historical conditions representative of a period of twenty years of pre-dam data (Fig. 7). In addition, extended periods of stream-flow intermittence also occurred between sampling events (Fig. 7).

While all of the common fish species we collected are considered tolerant to harsh physico-chemical conditions that accompany periods of low discharge and intermittence (e.g. Ostrand and Wilde 2004), it is likely that red shiner, which are less tolerant of high levels of conductivity than Pecos pupfish, plains killifish and western mosquitofish (Hoagstrom and Brooks 1999, Ostrand and Wilde 2004), may have been negatively impacted by low flow conditions that occurred during our study period. Although site scale differences in abundance of this species weren’t significant, the trend observed was that they were less prevalent in the refuge sites compared to the braided site and the ditched site. In addition, there was a significant negative relationship between red shiner abundance and higher levels of conductivity (Table 1).

Meiofauna are generally more tolerant of low flows and intermittence than fishes. However, studies have shown that intermittence and drought can negatively impact meiofauna assemblages (Ricci and Balsamo 2000, Pillay and Perissinotto 2009). This is supported by our findings, which indicated that for the majority of common taxa collected, abundance was higher in the more perennial sites (i.e. braided site and ditched site) in comparison to the refuge sites, and although differences in the abundance of Rotifera and Ostracoda were observed among the refuge sites, low flows may have masked differences for Nematoda and Turbellaria that may have been detected had adequate base flows been present.

A missing guild

Pelagic-broadcast spawning minnows are a declining guild of fishes, which occupy the Pecos River within our study area. We collected only three early-juvenile individuals of this guild during our entire study period. This included two plains minnows (*Hybognathus placitus*) collected from the ditched site during the final sampling trip and a single Rio Grande shiner (*Notropis jemezanus*) collected from the channelized site during the third sampling trip. The absence of this guild from all study sites is likely due to the harsh conditions that occurred during our study. Periods of low discharge and intermittence reduce spawning and recruitment of pelagic-broadcast spawning species (e.g. Hoagstrom et al. 2008b, Durham and Wilde 2009, Durham and Wilde 2014), which often require peaks in discharge to cue spawning and adequate base flows for recruitment

(Durham and Wilde 2008, 2009). Both conditions were absent during our study. Of particular concern is the fact that no early-juveniles of this declining guild were collected from the braided site, an area that has been a stronghold for these species (Hoagstrom and Brooks 2005, Hoagstrom et al. 2008a, 2008b).

MANAGEMENT IMPLICATIONS

We found that epibenthic meiofauna were present in nursery areas and that their densities fell within the range proposed to be required for the growth and survival of early-life-stage fishes (i.e. 100-1000 individuals L^{-1}) (Bone et al. 1995). For example, the density of rotifers (772.85 ± 247.74 individuals L^{-1}), a favorable prey item for early-life-stage fish, fell within this range. In contrast we found few pelagic zooplankton (Appendix II). This was likely due to the shallow nature of the slackwater habitats that we sampled (Table 3). Based on these findings, we conclude that epibenthic meiofauna are an overlooked prey source that are likely important to recruitment of fish on the plains and future diet studies assessing the importance of this prey source would be valuable.

The results of our study also have broad implications for the prioritization and planning of restoration efforts worldwide as they relate to the conservation of sensitive fish species. Studies have shown that climate change as well as human induced dewatering of rivers and other anthropogenic disturbances will likely result in increasing rates of extinction of riverine fishes in the future (Xenopoulos et al. 2005, Tedesco et al. 2013). This is especially true in arid and Mediterranean regions including the Southwestern USA, Mexico, Southern Europe and Australia (Tedesco et al. 2013). Hence, in these regions, physical channel restoration projects, which have been conducted in great numbers with both limited success and limited evaluation (e.g.

Kondolf et al. 2007, Jähnig et al. 2010), will likely confer little benefit to declining riverine fishes without flow regime restoration and the maintenance of minimum base flows.

More specifically, our findings are relevant for conservation actions in many plains streams, which are impacted by both channelization and dewatering (Huggins and Moss 1975, Cross and Moss 1987, Dodds et al. 2004). Indeed, for pelagic-broadcast spawning fishes as well as less tolerant demersal-adhesive spawning fishes of rivers on the plains, conditions of intermittence and little to no recruitment go hand-in-hand (Durham and Wilde 2009, Falke et al. 2010) and may confound any benefits of physical habitat restoration.

Thus, for fishes of the Pecos River, NM, fishes of plains rivers in general and riverine fishes in more arid regions worldwide, we suggest that a more holistic approach may be necessary that includes both physical channel restoration and flow-regime restoration, which mimics the natural flow regime and provides maintenance of minimum base flows (Dudley and Platania 2007, Hoagstrom et al. 2008b). It is likely that such an approach will be critical not only for the conservation of declining pelagic-broadcast spawning minnows within the Pecos River, NM (sensu Hoagstrom et al. 2008b) and small-bodied fishes throughout the plains, but for all riverine fishes impacted by channelization and dewatering due to both human disturbances and climate change.

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LITERATURE CITED

Aarnio, K., and E. Bonsdorff. 1997. Passing the gut of juvenile flounder, *Platichthys flesus*: differential survival of zoobenthic prey species. Marine Biology 129:11-14.

Bone, Q., N. B. Marshall, and J. H. S. Blaxter. 1995. Biology of fishes, 2nd edition. Blackie Academic and Professional Publishers, 1995.

Bott, T.L., and L. A. Kaplan. 1989. Densities of benthic protozoa and nematodes in a Piedmont stream. Journal of the North American Benthological Society 8:187-196.

Clarke, K., and R. M. Warwick. 2001. Change in marine communities: an approach to statistical analysis and interpretation.

Copp, G. H. 1990. Effect of regulation on 0+ fish recruitment in the great ouse, a lowland river. Regulated Rivers: Research & Management 5(3):251-263.

Cross, F. B., and R. E. Moss. 1987. Historic changes in fish communities and aquatic habitats in plains streams of Kansas. Community and evolutionary ecology of North American stream fishes. University of Oklahoma Press, Norman (1987):155-165.

Dodds, W. K., K. Gido, M.R. Whiles, K.M. Fritz, and W.J. Matthews. 2004. Life on the Edge: The ecology of Great Plains Prairie Streams. Bioscience 54(3):205-216.

Dudley, R. K., and S. P. Platania. 2007. Flow regulation and fragmentation imperil pelagic-spawning riverine fishes. Ecological Applications 17:2074-2086.

Durham, B. W., and G. R. Wilde. 2008. Asynchronus and synchronous spawning by smalleye shiner *Notropis buccula* from the Brazos River, Texas. Ecology of Freshwater Fish 17:528-541.

Durham, B. W., and G. R. Wilde. 2009. Effects of streamflow and intermittency on the reproductive success of two broadcast-spawning Cyprinid fishes. Copeia 2009(1):21-28.

Durham, B. W., and G. R. Wilde. 2014. Understanding complex reproductive ecology in fishes: the importance of individual and population-scale information. Aquatic Ecology 48(1):91-106.

Falke, J. A., K. R. Bestgen, and K. D. Fausch. 2010. Streamflow reductions and habitat drying affect growth, survival, and recruitment of brassy minnow across a Great Plains riverscape. Transactions of the American Fisheries Society 139:1566-1583.

Ferrari, I., A. Farabegoli, and R. Mazzoni. 1989. Abundance and diversity of planktonic rotifers in the Po River. Hydrobiologia 186/187:201-208.

Hammer, Ø., D. A. T. Harper, and P. D. Ryan. 2001. PAST: Paleontological statistics software package for education and data analysis. Palaeontologia Electronica 4(1): 9pp. http://palaeo-electronica.org/2001_1/past/issue1_01.htm

Hoagstrom, C. W. 2009. Causes and impacts of salinization in the Lower Pecos River. Great Plains Research 19:27-44.

Hoagstrom, C. W., and J. E. Brooks. 1999. Distribution, status, and abundance of the Pecos Pupfish, *Cyprinodon Pecosensis*. Tech Rpt. No. 2. New Mexico Department of Game and Fish, Santa Fe, NM. 76 pp.

Hoagstrom, C. W., and J. E. Brooks. 2005. Distribution and status of Arkansas River shiner *Notropis girardi* and Rio Grande shiner *Notropis jemezanus*, Pecos River, New Mexico. Texas Journal of Science 57(1):35-58.

Hoagstrom, C. W., J. E. Brooks, and S. R. Davenport. 2008a. Recent habitat association and the historical decline of Notropis simus pecosensis. River Research and Applications 24:789-803.

Hoagstrom, C. W., J. E. Brooks, and S. R. Davenport. 2008b. Spatiotemporal population trends of *Notropis simus pecosensis* in relation to habitat conditions and the annual flow regime of the Pecos River, 1992-2005. Copeia 2008:5-15.

Hoagstrom, C. W., and T. F. Turner. 2013. Recruitment ecology of pelagic-broadcast spawning minnows: paradigms from the ocean advance science and conservation of an imperiled freshwater fauna. Fish and Fisheries doi: 10.1111/faf.12054.

Hoagstrom, C. W., N. D. Zymonas, S. R. Davenport, D. L. Propst, and J. E. Brooks. 2010. Rapid species replacements between fishes of the North American plains: a case history from the Pecos River. Aquatic Invasions 5(2):141-153.

Huggins, D. G., and R. E. Moss. 1975. Fish population structure in altered and unaltered areas of a small Kansas stream. Transactions of the Kansas Academy of Science 77(1):18-30.

Humphries, P., R. A. Cook, A. J. Richardson, and L. G. Serafini. 2006. Creating a disturbance: manipulating slackwaters in a lowland river. River Research and Applications 22:524-542.

Humphries, P., A. J. King, and J. D. Koehn. 1999. Fish, flows and flood plains: links between freshwater fishes and their environment in the Murray-Darling River system, Australia. Environmental Biology of Fishes 56:129-151.

Jähnig, S. C., K. Brabec, A. Buffagni, S. Erba, A. W. Lorenz, T. Ofenböck, P. F. M. Verdonschot, and D. Hering. 2010. A comparative analysis of restoration measures and their effects on hydromorphology and benthic invertebrates in 26 central and southern European rivers. Journal of Applied Ecology 47:671-680.

JMP©, Version 11 SAS institute Inc., Cary, NC, 1989-2007.

Jurajda, P. 1995. Effect of channelization and regulation on fish recruitment in a flood plain river. Regulated Rivers: Research and Management 10:207-215.

King, A. J. 2004. Density and distribution of potential prey for larval fish in the main channel of a floodplain river: pelagic versus epibenthic meiofauna. River Research and Applications 20:883-897.

Kneib, R. T. 1993. Growth and mortality in successive cohorts of fish larvae within an estaurine nursery. Marine Ecology Progress Series 94(2):115-127.

Kondolf, G. M. 1997. Hungry water: effects of dams and gravel mining on river channels. Environmental Management 21:533-551.

Kondolf, G. M., S. Anderson, R. Lave, L. Pagano, A. Merenlender, and E. S. Bernhardt. 2007. Two decades of river restoration in California: what can we learn? Restoration Ecology 15(3):516-523.

Krumholz, L. A. 1948. Reproduction in the Western Mosquitofish, *Gambusia affinis affinis* (Baird & Girard), and its use in mosquito control. Ecological Monographs 18(1):1-43.

Kwak, T. J., and J. T. Peterson. 2007. Community indices, parameters, and comparisons. In: Guy CS, Brown ML. Analysis and interpretation of freshwater fisheries data. American Fisheries Society, Bethesda, Maryland. Pages 677-763.

Lau, J. K., T. E. Lauer, and M. L. Weinman. 2006. Impacts of channelization on stream habitats and associated fish assemblages in East Central Indiana. American Midland Naturalist 156(2):319-330.

Mann, R. H. K., and J. A. B. Bass. 1997. The critical water velocities of larval roach (*Rutilus rutilus*) and dace (*Leuciscus leuciscus*) and implications for river management. Regulated Rivers: Research and Management 13:295-301.

Marchant, R. 2002. Do rare species have any place in multivariate analysis for bioassessment? Journal of the North American Benthological Society 21(2):311-313.

Marteinsdottir, G., and K. W. Able. 1992. Influence of egg size on embryos and larvae of *Fundulus heteroclitus* (L.). Journal of Fish Biology 41:883-896.

Minckley, C.O., and H. E. Klaassen. 1969. Life history of the plains killifish *Fundulus kansae* (Garman), in the Smoky Hill River, Kansas. Transactions of the American Fisheries Society 98(3):460-465.

Moore, S. L., and J. H. Thorp. 2008. Coping with hydrogeomorphic variations in a prairie river: resiliency in young-of-the-year fishes. River Research and Applications 24(3):267-278.

Morris, L. A., R. N. Langemeier, T. R. Russell, and A. Witt Jr. 1968. Effects of main stem impoundments and channelization upon the limnology of the Missouri

River, Nebraska. Nebraska Game and Parks Commission Staff Research Publications Paper 10.

Ning, N. S. P., D. L. Nielsen, W. L. Paul, T. J. Hillman, and P. J. Suter. 2010. Microinvertebrate dynamics in riverine slackwaters and mid-channel habitats in relation to physico-chemical parameters and food availability. River Research and Applications 26:279-296.

Nunn, A. D., J. P. Harvey, and I. G. Cowx. 2007. The food and feeding relationships of larval and 0+ year juvenile fishes in lowland rivers and connected waterbodies. I. Ontogenetic shifts and interspecific diet similarity. Journal of Fish Biology 70:726-742.

O’Neill, B. J., and J. H. Thorp. 2011. Flow refugia for the zoobenthos of a sand-bed river: the role of physical-habitat complexity. Journal of the North American Benthological Society 30(2):546-558.

Ostrand, K. G., and G. R. Wilde. 2004. Changes in prairie stream fish assemblages restricted to isolated streambed pools. Transactions of the American Fisheries Society 133:1329-1338.

Pease, A. A., J. J. Davis, M. S. Edwards, and T. F. Turner. 2006. Habitat and resource use by larval and juvenile fishes in an arid-land river (Rio Grande, New Mexico). Freshwater Biology 51(3):475-486.

Pillay, D., and R. Perissinotto. 2009. Community structure of epibenthic meiofauna in the St. Lucia Estaurine Lake (South Africa) during a drought phase. Estaurine, Coastal and Shelf Science 81(1):94-104.

Polivka, K. M. 1999. The microhabitat distribution of the Arkansas River shiner, *Notropis girardi*: a habitat-mosaic approach. Environmental Biology of Fishes 55:265-278.

Price, A. E., P. Humphries, B. Gawne, and M. C. Thoms. 2013. Effects of discharge regulation on slackwater characteristics at multiple scales in a lowland river. Canadian Journal of Fishes and Aquatic Sciences 70(2):253-262.

Rahel, F. J., and L. A. Thel. (2004, March 18). Plains killifish (*Fundulus zebrinus*): a technical conservation assessment. USDA Forest Service, Rocky Mountain Region. Available: http://www.fs.fed.us/r2/projects/scp/assessments/plainskillifish.pdf.

Ricci, C., and M. Balsamo. 2000. The biology and ecology of lotic rotifers and gastrotrichs. Freshwater Biology 44:15-28.

Roca, J. R., A. Baltanas, and F. Uiblein. 1993. Adaptive responses in *Cypridopsis vidua* (Crustacea: Ostracoda) to food and shelter offered by a macrophyte (*Chara fragilis*). Hydrobiologia 262:127-131.

Saksena, V. P. 1962. The post-hatching stages of the red shiner, *Notropis lutrensis*. Copeia 1962(3):539-544.

Schiemer, F., H. Keckeis, W. Reckendorfer, and G. Winkler. 2001. The “inshore retention concept” and its significance for large rivers. Large Rivers 12(2-4):509-516.

Shields Jr., F. D., S. S. Knight, and C. M. Cooper. 1994. Effects of channel incision on base flow stream habitats and fishes. Environmental Management 18(1):43-57.

Spieth, H. R., T. Möller, Ch. Ptatscheck, A. Kazemi-Dinan, and W. Traunspurger. 2010. Meiobenthos provides a food resource for young cyprinids. Journal of Fish Biology 78:138-149.

Tedesco, P. A., T. Oberdorff, J-F. Cornu, O. Beauchard, S. Brosse, H. H. Dürr, G. Grenouillet, F. Leprieur, C. Tissueil, R. Zaiss, and B. Hugueny. 2013. A scenario for impacts of water availability loss due to climate change on riverine fish extinction rates. Journal of Applied Ecology 50:1105-1115.

Theilacker, G. H., and M. F. McMaster. 1971. Mass culture of the rotifer *Brachionus plicatilis* and its evaluation as a food for larval anchovies. International Journal on Life in Oceans and Coastal Waters 10(2):183-188.

Thorp, J. H., A. R. Black, K. H. Haag, and J. D. Wehr. 1994. Zooplankton assemblages in the Ohio River: Seasonal, tributary, and navigation dam effects. Canadian Journal of Fisheries and Aquatic Sciences 51:1634-1643.

Thorp, J. H., and A. P. Covich. 2009. Ecology and classification of North American freshwater invertebrates. Academic Press, 2009.

Thorp, J. H., M. C. Thoms, and M. D. Delong. 2006. The riverine ecosystem synthesis: biocomplexity in river networks across space and time. River Research and Applications 22(2): 123-147.

Tito de Morais, L., and J. Y. Bodiou. 1984. Predation on meiofauna by juvenile fish in a Western Mediterranean flatfish nursery ground. Marine Biology 82:209-215.

USBOR (U.S. Bureau of Reclamation). 2009. Pecos River channel restoration at the Bitter Lake National Wildlife Refuge, Chaves County, New Mexico, Environmental Assessment. U.S. Bureau of Reclamation, Albuquerque Area Office, Albuquerque, New Mexico.

Vietz, G. J., M. J. Sammonds, and M. J. Stewardson. 2013. Impacts of flow regulation on slackwaters in river channels. Water Resources Research 49:1797-1811.

Werner, R. G., and J. H. S. Blaxter. 1980. Growth and survival of larval herring (*Clupea harengus*) in relation to prey density. Canadian Journal of Fisheries and Aquatic Sciences 37(7):1063-1069.

Wieser, W. 1991. Limitations of energy acquisition and energy use in small poikilotherms: evolutionary implications. Functional Ecology 5:234-240.

Xenopoulos, M. A., D. M. Lodge, J. Alcamo, M. Märker, K. Schulze, and D. P. Van Vuurens. 2005. Scenarios of freshwater fish extinctions from climate change and water withdrawal. Global Change Biology 11:1557-1564.

Yildrim, A., and E. J. Peters. 2006. Life history characteristics of red shiner, *Cyprinella lutrensis*, in the Lower Platte River, Nebraska, USA. Journal of Freshwater Ecology 21(2):307-314.

TABLES

Table 1. Results of Analysis of Covariance (ANCOVA) or Analysis of Variance (ANOVA) testing for site scale differences in rank abundance of common fish species.

Source	df	*F*	*P*	Source	df	*F*	*P*
Red shiner				Plains killifish			
Site	4	2.8	0.029	Site	4	5.3	≤ 0.001
Conductivity (µmhos)	1	7.0	0.009	Volume (m^3)	1	0.3	0.568
Volume (m^3)	1	0.3	0.586	Western mosquitofish			
Pecos pupfish				Site	4	10.5	≤ 0.001
Site	4	2.2	0.078				
Conductivity (µmhos)	1	0.6	0.456				
Volume (m^3)	1	0.0	0.929				

Table 2. Results of Analysis of Covariance (ANCOVA) testing for site scale differences in rank abundance of common meiofauna taxa.

Source	df	*F*	*P*	Source	df	*F*	*P*
Ceratopogonidae				Ostracoda			
Site	4	2.2	0.074	Site	4	32.3	≤ 0.001
Conductivity (µmhos)	1	10.7	0.001	Conductivity (µmhos)	1	2.4	0.125
Volume (m^3)	1	1.6	0.214	Volume (m^3)	1	0.0	0.846
Chironomidae				Rotifera			
Site	4	1.6	0.187	Site	4	8.9	≤ 0.001
Conductivity (µmhos)	1	4.2	0.042	Conductivity (µmhos)	1	0.1	0.806
Volume (m^3)	1	0.9	0.343	Volume (m^3)	1	0.1	0.755
Nematoda				Turbellaria			
Site	4	4.5	0.002	Site	4	7.9	≤ 0.001
Conductivity (µmhos)	1	12.8	≤ 0.001	Conductivity (µmhos)	1	1.0	0.309
Volume (m^3)	1	0.3	0.610	Volume (m^3)	1	3.6	0.061
				Total meiofauna			
				Site	4	12.9	≤ 0.001
				Conductivity (µmhos)	1	0.3	0.573

Table 3. Slackwater mesohabitat characteristics by site. Total number of slackwaters, range of depths, mean ± SE depth, range of conductivity and mean ± SE conductivity are shown.

Site	Total Slackwaters	Depth (cm) min – max	Mean depth (cm)	Conductivity (µhmos) min - max	Mean conductivity (µmhos)
Braided	24	8.0 – 90.0	27.0 ± 4.1	2790 - 19431	5001 ± 682
Unchannelized	25	2.0 – 71.0	18.3 ± 4.2	3687 - 34824	13682 ± 1992
Restored	22	1.5 – 54.0	14.5 ± 2.9	6734 - 77000	20090 ± 3638
Channelized	31	2.5 – 90.0	21.3 ± 3.9	5420 - 27950	15652 ± 1142
Ditched	18	2.0 – 53.0	17.8 ± 3.0	11630 - 18340	14362 ± 472

FIGURES

Figure 1. Map of the Pecos River, New Mexico. Stars and Roman numerals indicate sampling sites. Roman numeral I represents the braided site, II* represents the unchannelized site, III* represents the restored site, IV* represents the channelized site and V represents the ditched site. Distances between the unchannelized, restored and channelized sites within the Bitter Lake National Wildlife Refuge (i.e. sites II*, III* and IV*) were approximately 2 km and are encompassed by the refuge site star.

Figure 2. Slackwater nursery habitat characteristics. Panel A: open circles indicate means and upper and lower whiskers indicate standard errors. Means not sharing the same letter are significantly different (Tukey's HSD tests: $P < 0.05$). Panel B: open circles indicate means and upper and lower whiskers indicate standard errors. The total number of slackwaters sampled is shown in the upper right corner of both panels.

Figure 3. Rank abundance of early-juvenile fish species which comprised at least 5% of the total assemblage sampled. Open circles indicate ANCOVA adjusted means for red shiner, Pecos pupfish and plains killifish. Open circles indicate the mean for western mosquitofish. Upper and lower whiskers indicate standard errors for all species. Means not sharing the same letter are significantly different (Tukey's HSD tests: $P < 0.05$). Means without letters did not exhibit any significant differences (ANCOVA site effect: $P > 0.05$).

Figure 4. Non-metric multidimensional scaling ordination of early-juvenile fish (A) and meiofauna (B) assemblages by site. Centroids represent sampling trips for each site. Numbers indicate the sampling trip represented by each centroid, with 1 = June 2012, 2 = August 2012, 3 = June 2013 and 4 = August 2013.

Figure 5. Spawning phenology of the three most common fish species collected for 2012 (upper panel) and 2013 (lower panel). Symbols represent hatch date estimates for each species with lines representing the upper and lower standard deviation of each estimate. The solid line represents daily discharge values (m^3 sec^{-1}).

Figure 6. Rank abundance of total meiofauna and those taxa which comprised at least 5% of the total meiofauna assemblage sampled. Open circles indicate ANCOVA adjusted means and upper and lower whiskers indicate standard errors. Means not sharing the same letter are significantly different (Tukey's HSD tests: $P < 0.05$). Means without letters did not exhibit any significant differences (ANCOVA site effect: $P > 0.05$).

Figure 7. Mean daily discharge from the USGS Near Artesia NM gage (08396500). Panels A and B indicate mean daily discharge for our study period (2012-2013 black hydrograph) and historical mean daily discharge (1930-1931 gray hydrograph). Panel A is scaled to show historical peaks in discharge. Panel B is scaled to better illustrate discharge during the study period. The data from 1930-1931 was chosen as representative of mean daily discharge from a 20 year pre-dam period. In panel B, spaces between vertical hashed lines on the x-axis indicate sampling periods for 2012 and 2013.

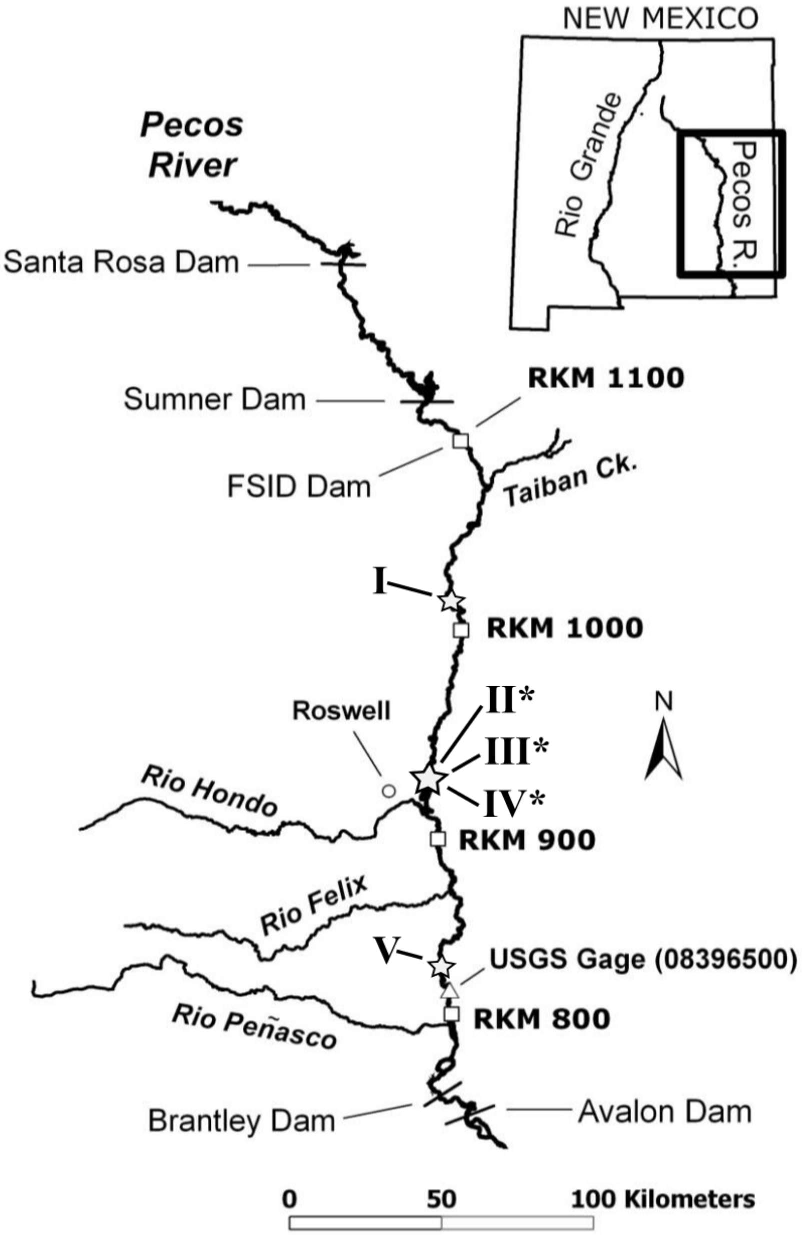
NEW MEXICO
Rio Grande
Pecos R.
Pecos River
Santa Rosa Dam
Sumner Dam
RKM 1100
FSID Dam
Taiban Ck.
I
RKM 1000
II*
III*
IV*
Roswell
N
Rio Hondo
RKM 900
Rio Felix
V
USGS Gage (08396500)
RKM 800
Rio Peñasco
Brantley Dam
Avalon Dam
0
50
100 Kilometers

Figure 1.

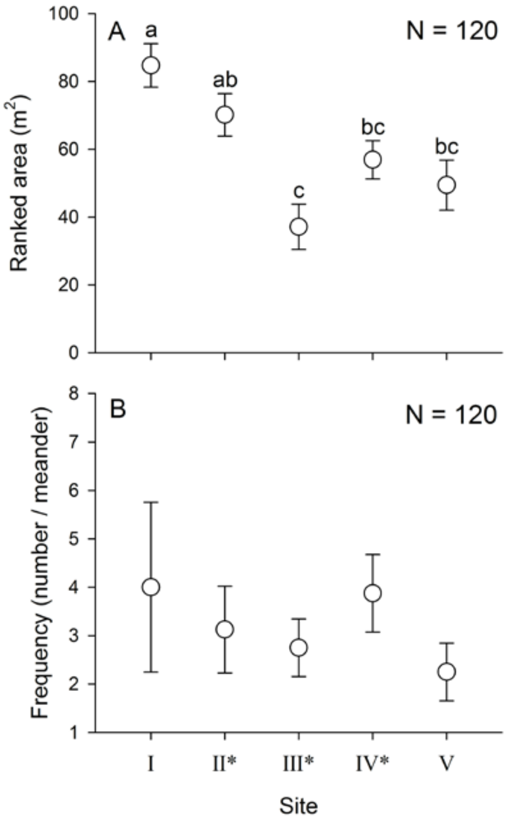
A
N = 120
Ranked area (m²)
100
80
60
40
20
0
a
ab
c
bc
bc
B
N = 120
Frequency (number / meander)
8
7
6
5
4
3
2
1
I
II*
III*
IV*
V
Site

Figure 2.

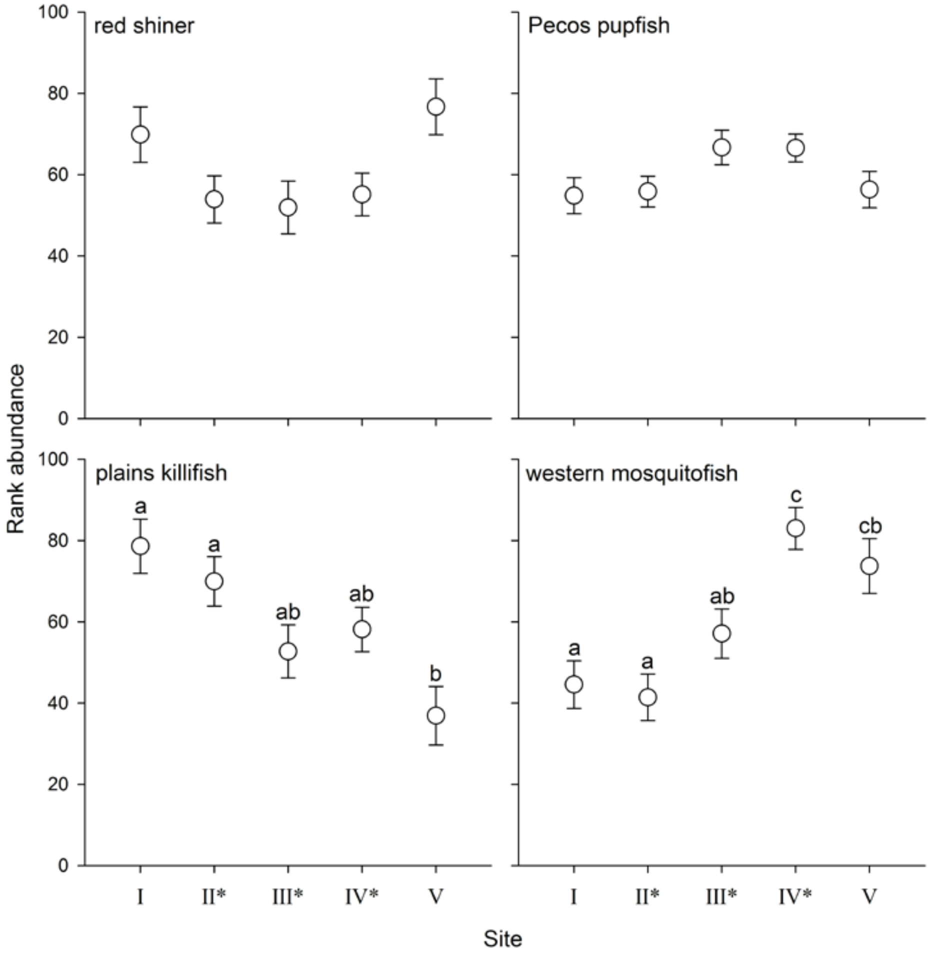
red shiner
Pecos pupfish
plains killifish
western mosquitofish
Rank abundance
0
20
40
60
80
100
a
a
ab
ab
b
a
a
ab
c
cb
I
II*
III*
IV*
V
Site

Figure 3.

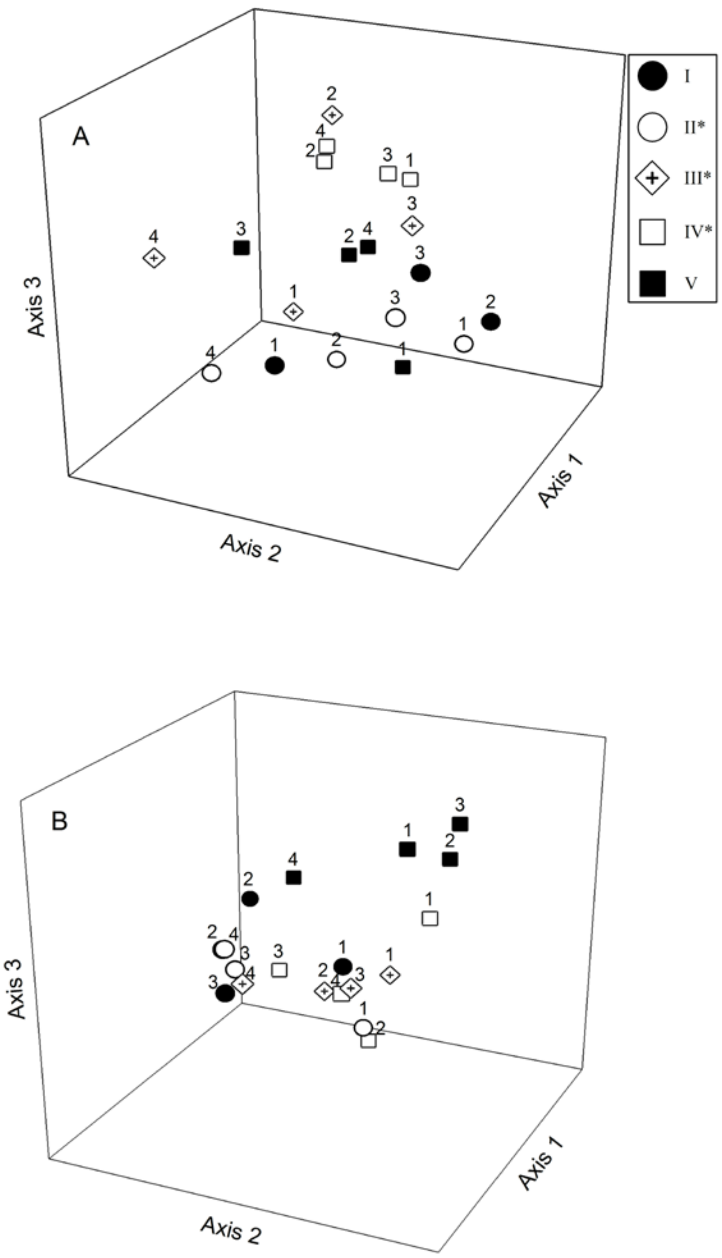

Figure 4.

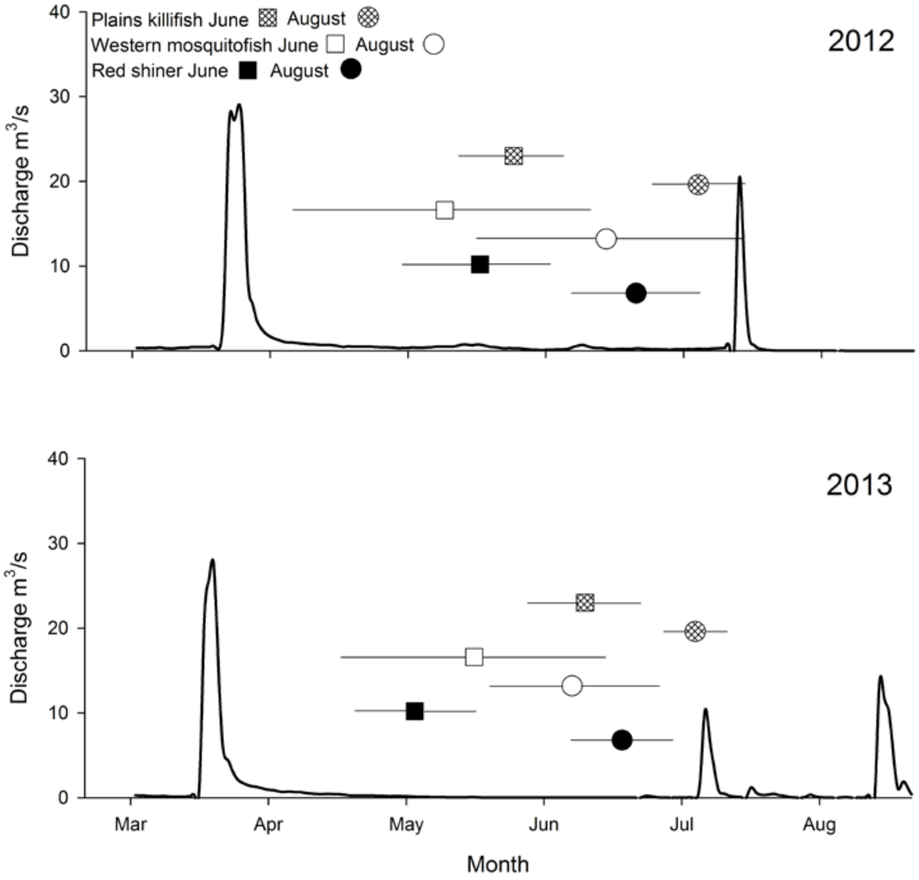
Plains killifish June ▩ August ◎
Western mosquitofish June ☐ August ○
Red shiner June ■ August ●
2012
2013
Discharge m^3/s
0
10
20
30
40
Mar
Apr
May
Jun
Jul
Aug
Month

Figure 5.

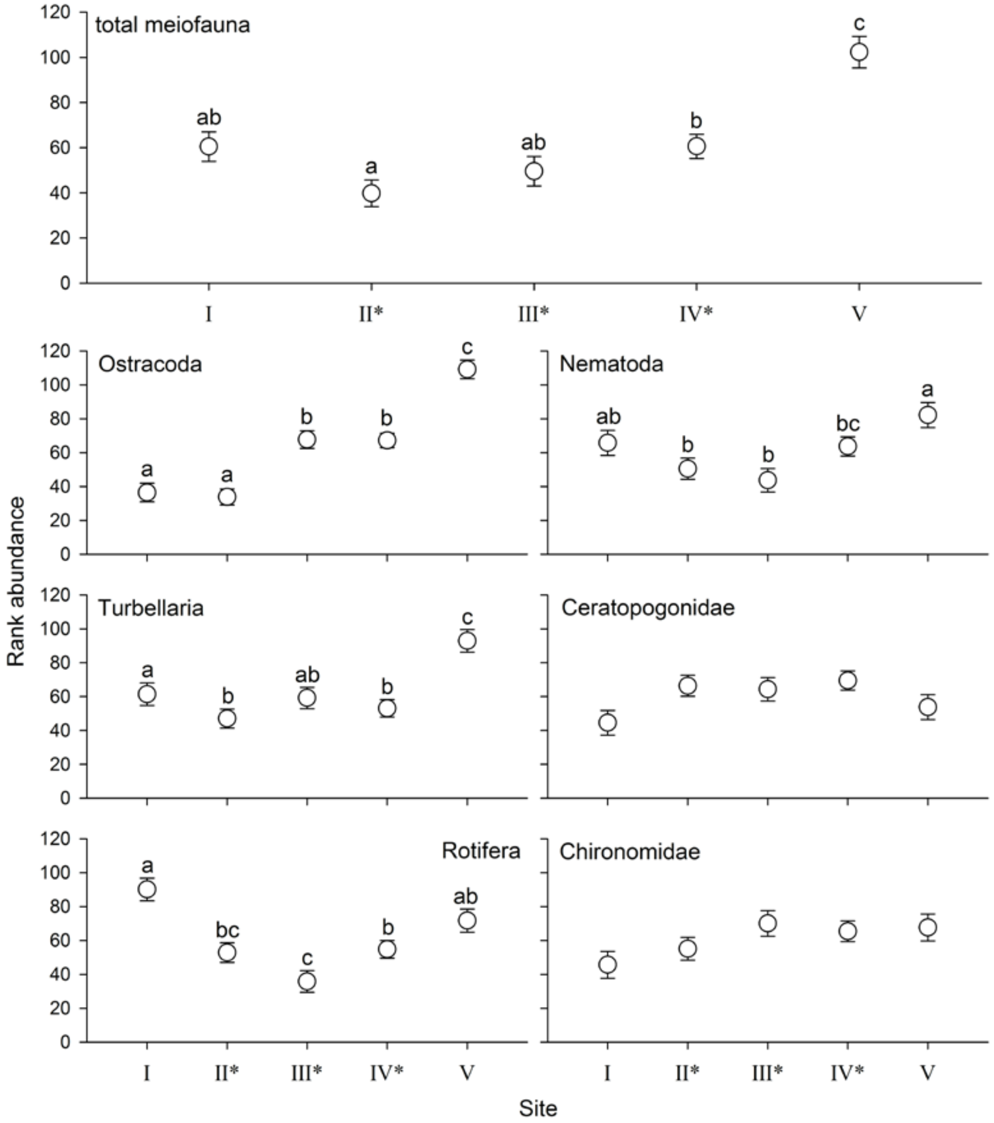
total meiofauna
Ostracoda
Nematoda
Turbellaria
Ceratopogonidae
Rotifera
Chironomidae
Rank abundance
Site
0
20
40
60
80
100
120
I
II*
III*
IV*
V
a
ab
b
bc
c

Figure 6.

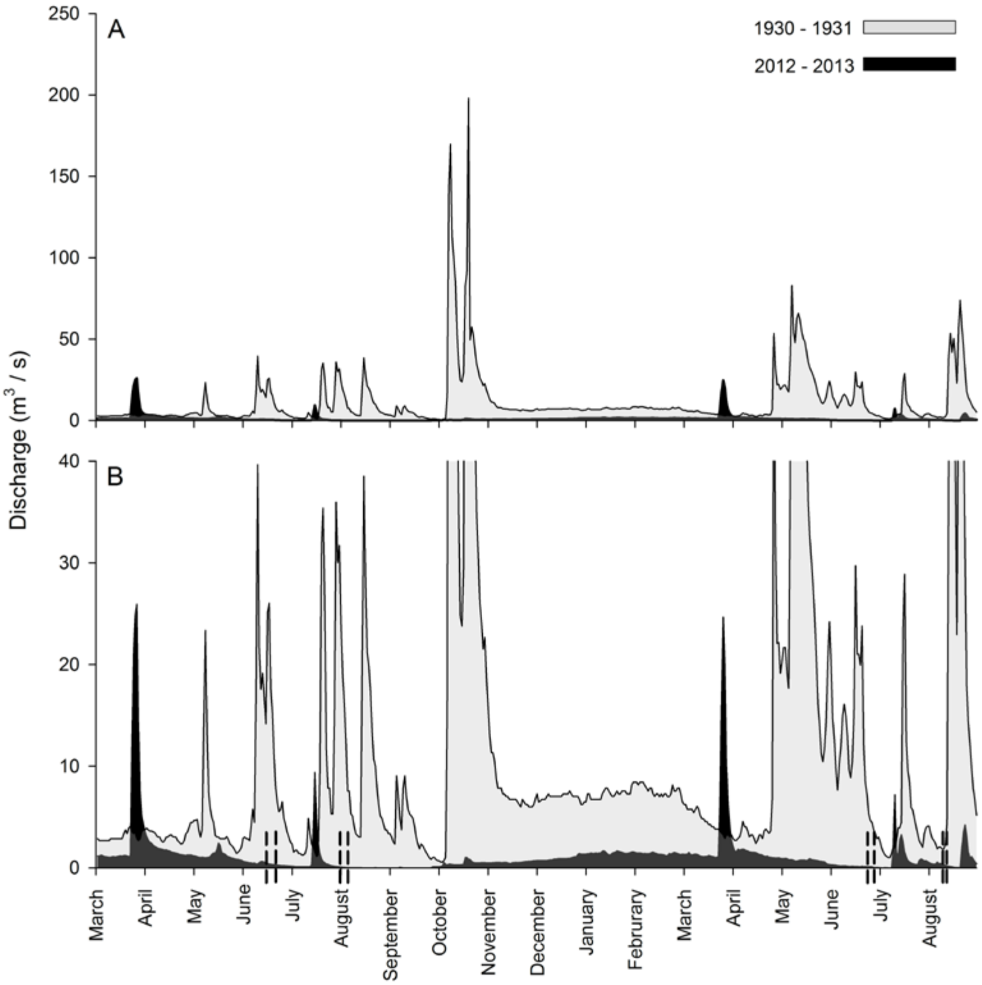
250
200
150
100
50
0
A
1930 - 1931
2012 - 2013
Discharge (m³ / s)
40
30
20
10
0
B
March
April
May
June
July
August
September
October
November
December
January
Februrary
March
April
May
June
July
August

Figure 7.

APPENDIX

Appendix I. Total density and percent abundance of early-juvenile fish collected by site and trip.

	Braided	Unchannelized	Restored	Channelized	Dtiched
June 2012					
Total density (fish / 100 m^2)	512	3210	1059	8883	1320
red shiner	10.9%	50.6%	36.9%	6.7%	96.3%
plains killifish	65.8%	49.4%	9.7%	8.0%	1.2%
western mosquitofish	23.3%	0.0%	49.8%	82.5%	2.5%
rainwater killifish	0.0%	0.0%	3.6%	2.5%	0.0%
Pecos pupfish	0.0%	0.0%	0.0%	0.2%	0.0%
August 2012					
Total density (fish / 100 m^2)	8359	1124	5887	4900	3717
river carpsucker	0.4%	0.0%	0.0%	0.0%	0.0%
white sucker	0.4%	0.0%	0.0%	0.0%	0.0%
Pecos pupfish	0.0%	0.0%	54.4%	0.8%	0.0%
red shiner	62.7%	68.9%	0.0%	2.8%	19.1%
plains killifish	28.1%	17.1%	0.0%	1.9%	0.0%
western mosquitofish	3.1%	12.6%	45.6%	82.9%	38.6%
rainwater killifish	0.0%	0.0%	0.0%	0.6%	0.0%
sand shiner	5.2%	0.0%	0.0%	0.0%	0.0%
fathead minnow	0.1%	1.5%	0.0%	11.1%	42.3%
June 2013					
Total density (fish / 100 m^2)	3671	2090	6555	7320	903
Pecos pupfish	0.0%	6.8%	53.8%	25.1%	1.0%
red shiner	1.3%	0.0%	0.5%	0.0%	6.6%
plains killifish	83.2%	88.0%	41.5%	22.5%	2.4%
western mosquitofish	15.4%	5.1%	3.5%	51.3%	89.9%
rainwater killifish	0.0%	0.0%	0.6%	0.8%	0.0%
Rio Grande shiner	0.0%	0.0%	0.0%	0.3%	0.0%
August 2013					
Total density (fish / 100 m^2)	NA	220	355	6996	2779
Mexican tetra	NA	0.0%	0.0%	0.0%	0.5%
Pecos pupfish	NA	0.0%	69.2%	5.3%	0.0%
red shiner	NA	0.0%	7.7%	0.9%	35.2%
plains killifish	NA	50.7%	15.4%	3.1%	0.0%
western mosquitofish	NA	49.3%	0.0%	85.0%	63.3%
plains minnow	NA	0.0%	0.0%	0.0%	1.1%
rainwater killifish	NA	0.0%	7.7%	4.7%	0.0%
fathead minnow	NA	0.0%	0.0%	0.9%	0.0%

Appendix II. Total density and percent abundance of meiofauna collected by site and trip.

	Braided	Unchannelized	Restored	Channelized	Ditched
June 2012					
Total density (Indv. / L^{-1})	15,346	15,102	15,954	32,274	21,679
P. Rotifera	11.1%	16.9%	0.8%	0.8%	3.9%
P. Nematoda	28.6%	12.9%	38.9%	12.1%	14.6%
P. Platyhelminthes					
C. Turbellaria	14.3%	2.4%	10.7%	6.0%	0.6%
P. Annelida					
C. Oligochaeta	4.8%	0.0%	0.0%	0.4%	0.0%
P. Arthropoda					
sub C. Acari	0.0%	2.4%	0.0%	0.0%	0.0%
C. Crustacea					
sub O. Cladocera					
F. Daphniidae	0.0%	0.0%	0.0%	0.0%	1.1%
F. Illyocryptidae	0.8%	0.0%	0.0%	0.0%	0.0%
sub C. Ostracoda	7.9%	12.9%	16.8%	30.6%	65.7%
sub C. Ostracoda nauplii	0.0%	0.0%	0.8%	3.0%	0.6%
sub C. Copepoda nauplii	3.2%	0.0%	0.0%	14.0%	6.7%
O. Harpacticoida	0.8%	0.0%	0.0%	15.5%	1.7%
O. Cyclopoida	3.2%	0.8%	0.0%	0.0%	0.0%
C. Insecta					
O. Coleoptera					
F. Curculionidae	0.0%	0.0%	0.0%	0.4%	0.0%
O. Diptera pupa	0.0%	0.8%	0.0%	0.0%	0.0%
O. Diptera					
F. Ceratopogonidae	10.3%	31.5%	17.6%	12.1%	1.1%
F. Chironomidae	15.1%	18.5%	14.5%	5.3%	3.9%
F. Chironomidae pupa	0.0%	0.8%	0.0%	0.0%	0.0%
August 2012					
Total density (Indv. / L^{-1})	59,312	3,410	13,397	30,691	42,383
P. Rotifera	74.3%	14.3%	2.7%	5.6%	2.0%
P. Nematoda	9.7%	53.6%	53.6%	71.4%	9.2%
P. Platyhelminthes					
C. Turbellaria	5.1%	0.0%	2.7%	4.0%	14.4%
P. Annelida					
C. Oligochaeta	4.5%	0.0%	0.0%	0.0%	0.3%
P. Arthropoda					
sub C. Acari	0.4%	10.7%	0.9%	0.0%	0.6%
C. Crustacea					
sub O. Cladocera					
sub C. Ostracoda	0.6%	0.0%	9.1%	4.0%	52.3%
sub C. Ostracoda nauplii	0.0%	0.0%	0.0%	0.0%	0.3%
sub C. Copepoda nauplii	0.8%	0.0%	7.3%	1.2%	8.9%
O. Harpacticoida	0.0%	0.0%	5.5%	1.6%	5.2%
O. Cyclopoida	0.6%	0.0%	0.0%	0.0%	0.0%
C. Insecta					
O. Ephemeroptera	0.2%	0.0%	0.0%	0.0%	0.0%
O. Diptera					
F. Ceratopogonidae	1.4%	0.0%	3.6%	3.6%	1.4%
F. Chironomidae	2.3%	17.9%	14.5%	8.7%	5.5%
F. Chironomidae pupa	0.0%	3.6%	0.0%	0.0%	0.0%

Appendix II cont.

	Braided	Unchannelized	Restored	Channelized	Ditched
June 2013					
Total density (Indv. / L^{-1})	9,356	5,555	4,873	24,706	32,990
P. Rotifera	30.2%	27.2%	3.0%	32.5%	1.2%
P. Nematoda	16.1%	38.6%	26.0%	32.5%	12.0%
P. Platyhelminthes					
C. Turbellaria	4.2%	7.0%	17.0%	3.6%	6.8%
P. Annelida					
C. Oligochaeta	3.1%	0.0%	1.0%	0.2%	0.1%
P. Arthropoda					
sub C. Acari	2.1%	2.6%	6.0%	0.0%	0.1%
C. Crustacea					
sub O. Cladocera					
sub C. Ostracoda	4.7%	6.1%	39.0%	6.5%	78.7%
sub C. Ostracoda nauplii	0.5%	0.0%	0.0%	1.4%	0.0%
sub C. Copepoda nauplii	12.0%	4.4%	0.0%	10.1%	0.0%
O. Harpacticoida	15.1%	1.8%	1.0%	8.9%	0.6%
O. Cyclopoida	1.6%	0.9%	1.0%	1.4%	0.0%
C. Insecta					
O. Coleoptera	0.5%	0.9%	0.0%	0.4%	0.0%
O. Diptera					
F. Ceratopogonidae	3.1%	5.3%	0.0%	1.2%	0.0%
F. Chironomidae	6.8%	5.3%	6.0%	1.4%	0.4%
August 2013					
Total density (Indv. / L^{-1})	NA	2,826	2,388	5,555	40,348
P. Rotifera	NA	12.1%	2.0%	3.5%	65.2%
P. Nematoda	NA	63.8%	59.2%	22.8%	12.8%
P. Platyhelminthes					
C. Turbellaria	NA	5.2%	4.1%	10.5%	3.0%
P. Annelida					
C. Oligochaeta	NA	0.0%	0.0%	0.9%	0.4%
P. Arthropoda					
sub C. Acari	NA	0.0%	2.0%	3.5%	0.4%
C. Crustacea					
sub O. Cladocera					
sub C. Ostracoda	NA	0.0%	8.2%	28.9%	15.3%
sub C. Copepoda nauplii	NA	0.0%	2.0%	1.8%	0.4%
O. Harpacticoida	NA	3.4%	0.0%	3.5%	0.2%
O. Cyclopoida	NA	0.0%	0.0%	0.9%	0.0%
C. Insecta					
O. Diptera					
F. Ceratopogonidae	NA	1.7%	0.0%	1.8%	0.6%
F. Chironomidae	NA	13.8%	22.4%	21.9%	1.7%