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10-2018

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Carl A. Favata

Anabela Maia

Manisha Pant

Virginia Institute of Marine Science

Vaskar Napal

Virginia Institute of Marine Science

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RESEARCH ARTICLE

Fish assemblage change following the structural restoration of a degraded stream

Carl A. Favata  | Anabela Maia  | Manisha Pant  | Vaskar Nepal  | Robert E. Colombo 

Department of Biological Sciences, Eastern Illinois University, Charleston, Illinois, USA

Correspondence

Carl A. Favata, Department of Biological Sciences, Eastern Illinois University, Charleston, IL 61920, USA.
Email: carlfavata@gmail.com

Present Address

Anabela Maia, Department of Biology, Rhode Island College, Providence, Rhode Island, USA.

Manisha Pant, Virginia Institute of Marine Science, The College of William & Mary, Gloucester Point, Virginia, USA.

Vaskar Nepal, Virginia Institute of Marine Science, The College of William & Mary, Gloucester Point, Virginia, USA.

Funding information

Eastern Illinois University; U.S. Geological Survey; University of Illinois Urbana Champaign, Grant/Award Number: G11AP20077; Illinois Water Resources Center, Grant/Award Number: 2011-03502-07; Illinois Environmental Protection Agency, Grant/Award Number: 3191416

Abstract

Decades of anthropogenic pressure have harmed riverscapes throughout North America by degrading habitats and water quality and can result in the extirpation of sensitive aquatic taxa. Local stream restoration projects have increased in frequency, but monitoring is still infrequent. In 2010, Kickapoo Creek in East Central Illinois was subjected to a stream restoration project that included implementation of artificial riffles, riprap, scouring keys, and riparian vegetation. We monitored the restoration efforts for 6 years after the restoration through annual sampling efforts at restored and reference sites to determine changes in habitat and fish assemblage using standard habitat sampling and electrofishing techniques. We observed distinct temporal and spatial shifts in physico-chemical parameters along with changes in fish community structure. Although biotic integrity remained moderately low in reference assemblages, restored reaches showed 3-year delay in response to restoration, with biotic integrity positively linked to additional instream habitat and altered channel morphology. Larger substrate sizes, submerged terrestrial vegetation, and newly formed scour pools along with reduced siltation were found in the restored sites, in contrast to the reference sites. These changes resulted in increased species diversity, reduced number of opportunistic species and consequently an overall increase in health of fish communities. We also observed recruitment of habitat specialists and increase in species with reproductive strategies that rely on complex substrates. The results of this study highlight some of the complex dynamics driving reach-scale restoration projects. We demonstrate the usefulness of structural restoration as a management tool to increase biotic integrity through long-term alteration of critical habitat. The delay in the response of species to the restoration efforts emphasizes the need for long-term continuous temporal and spatial monitoring.

KEYWORDS

biotic integrity, fish assemblages, habitat, QHEI, restoration

1 | INTRODUCTION

Decades of anthropogenic pressure have devastated lotic ecosystems across the riverscapes of North America, resulting in degradation of instream habitat and contributing to sharp declines in biotic integrity (NRC, 1992). Namely, agricultural practices in the Midwest have prompted increased bank erosion and sedimentation, leading to a loss

of critical habitat for aquatic organisms (Berkman & Rabeni, 1987; Walser & Bart, 1999; Wood & Armitage, 1997). Local stream restoration projects are increasingly frequent (Lake, Bond, & Reich, 2007; Moerke & Lamberti, 2003), but little effort has been allocated to monitoring (Moerke & Lamberti, 2003; NRC, 1992; Palmer et al., 2005; Roni, 2005). In addition, monitoring is normally limited in scope and in duration due to lack of resources which could hinder the

perceived success of the restoration and subsequent management initiatives (Bond & Lake, 2003; Pretty et al., 2003; Roni, 2005).

Reach-scale restoration projects may effectively mitigate ecological damage (Lake et al., 2007; Moerke & Lamberti, 2003; Palmer et al., 2005) and are more effective when conducted in cooperation with other watershed-level efforts (Bond & Lake, 2003; Lake et al., 2007; Palmer, Ambrose, & Poff, 1997). However, watershed management has been shown to be the most efficient approach to recover biodiversity in impacted streams throughout the rural Midwest (Rhoads, Wilson, Urban, & Herricks, 1999). When implemented in support of larger watershed-level conservation, local restoration can take two main forms: directly by altering geomorphology through dredging, addition of substrates in the form of riffles, and scouring keys; or indirectly, by altering the riparian ecosystem through the addition of riparian vegetation, overhanging vegetation, and buffer areas (Baldigo, Warren, Ernst, & Mulvihill, 2008; Lake et al., 2007). Altogether, these can reduce rates of sedimentation and run-off through addition of riparian buffer strips and direct alteration of instream habitat (Berkman & Rabeni, 1987; Wood & Armitage, 1997; Rabeni, Doisy, & Zweig, 2005).

It is especially important to consider habitat fragmentation, biodiversity, and organismal life histories when determining project success (Lake et al., 2007). Stream fishes depend on a variety of habitats to survive in complex three-dimensional environments. Geomorphology dictates many stream characteristics that are likely drivers of biotic integrity (Schlosser, 1982). Aspects such as time since channelization, boulder abundance, overhanging vegetation, and frequency of erosion have all been shown to drive biotic integrity in Midwestern streams (Talmage, Perry, & Goldstein, 2002; Wang et al., 1998). Similarly, the presence of pools has been shown to drive species richness and diversity (Schlosser, 1982). Siltation has also been negatively linked with fish diversity (Talmage et al., 2002), likely due to a reduced abundance of species that rely on benthic invertebrates and algae as food sources (Berkman & Rabeni, 1987). There is also evidence that substrate diversity drives species diversity (Schlosser, 1982). Habitat use and productivity in fish are also linked to instream structures, which undoubtedly serve as crucial sources of refuge for many species. Coarse substrate and boulder cover provide diverse and stable habitats in degraded systems and may also promote productivity of aquatic macroinvertebrates (Fischenich, 2003; White, Gerken, Paukert, & Makinster, 2009). Course woody debris and vegetation can also benefit stream ecosystems by increasing channel depths and reducing siltation (Angermeier & Karr, 1984; Talmage et al., 2002). Thermal shading benefits fauna in warm water streams and can affect seasonal habitat use, distribution, and behaviour of a variety of stream fish (Peterson & Rabeni, 1996).

Restorations often target increases in the heterogeneity of these habitats (Whiteway, Biron, Zimmermann, Venter, & Grant, 2010), for example, woody debris, terrestrial vegetation, and boulder cover are often recommended for restoration projects (Talmage et al., 2002). However, few projects have examined the relationships between restoration of instream habitat, the parameters that it affects, and the resulting shifts in community-level biotic response (White et al., 2009).

This study aims to determine the long-term impacts on fish community structure following direct and indirect habitat restoration in Kickapoo Creek, East Central Illinois. Operating at the reach-scale, this project and subsequent monitoring is supported under the larger Embarras River watershed monitoring plan (Illinois Environmental Protection Agency; IEPA). Our main objectives are (a) to describe changes to fish assemblage structure at restored sites at the genus and community levels, (b) to describe responses of biotic integrity, and (c) to identify habitat drivers of fish assemblages. We expect to observe an increase in habitat heterogeneity in the restored sites but not in the reference sites, and we predict increased biotic integrity and recruitment of sensitive species.

2 | METHODS

2.1 | Study area

Kickapoo Creek (Latitude 39°27', Longitude 88°13') is a fourth-order, low gradient stream which originates south of Mattoon, Illinois, and flows east for nearly 66 stream km until meeting its confluence with the Embarras River (Figure 1). Draining approximately 265 km², this human-impacted system is subjected to multiple anthropogenic pressures within a relatively small basin. Land use within the Kickapoo Creek watershed consists primarily of agriculture, disconnected fragments of forest, grasslands, and urban stressors (e.g., road crossings, golf course, sewage treatment plant, and residential areas).

2.2 | Stream restoration of Kickapoo Creek

As part of the larger Embarras River watershed, identified by the IEPA as a region of concern, Kickapoo Creek was subjected to restoration and mitigation efforts. Following a chemical-induced fish kill in 2001, mitigation efforts from Illinois Department of Natural Resources and IEPA enabled the structural restoration of over 800 m of streambank and main channel habitat in September 2010. Prior to an instream restoration project, all study reaches shared similar habitat characteristics. They consisted of shifting sand-gravel substrates, elevated levels of bank erosion and sedimentation, and stream slope averaged 9.2 ft/mile with an average soil permeability of 1.4 in./hr (Pers. Obs.). To improve habitat heterogeneity and biotic integrity, the restoration included construction of two artificial rock 0.5-m-high v-shaped Newbury riffles (Newbury Hydraulics, Okanagan Centre British Columbia, Canada). These increased average water depths and simulated scour pool hydraulics within the restoration reach. Boulder riprap was installed along both streambanks, and 5-ft scouring keys were used to facilitate geomorphic stabilization and improve hydrologic conditions. Additionally, revegetation of streambanks further aided the recovery of riparian habitat and helped to reduce bank erosion. Restoration of riparian vegetation included a wide heavy crop filter strip (i.e., Winter Wheat *Triticum spp.*) and an assortment of native prairie grasses (e.g., Big Bluestem *Andropogon gerardii*, Switch Grass *Panicum virgatum*, and Indian Grass *Sorghastrum nutans*) planted in thin filter strips along each bank.

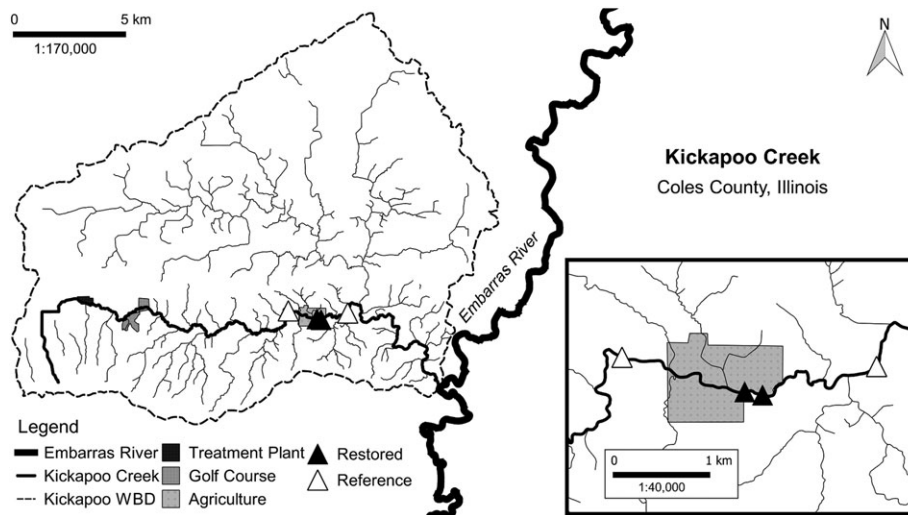


FIGURE 1 Locations of restored and reference sites monitored within the Kickapoo Creek watershed boundary (WBD) in East Central Illinois from 2010 to 2015

2.3 | Sampling sites

Beginning immediately after the restoration in summer 2010, habitat and fish communities were examined in three fixed 200-m sites—two located within the larger restoration reach and associated with each artificial riffle, and one site approximately 1.8-km upstream, which served as a reference. The restoration sites started at each riffle and extended 200-m downstream. Both sites were entirely within the 800-m restoration area. In 2012, an additional 200-m reach was added approximately 1.8-km downstream of the restoration site as an added reference.

2.4 | Habitat assessment

Stream habitat and integrity were monitored annually in the fall using the Qualitative Habitat Evaluation Index (QHEI; Rankin, 1989) by teams of two researchers, following extensive training and using the Illinois Division of Natural Resources training handbook. The following sub-metrics were measured: substrate type, origin and quality, instream cover and amount, channel morphology (sinuosity, development, channelization, stability, and modifications), riparian zone width, flood plain quality, bank erosion, pool maximal depth, morphology and current velocity, riffle depth, run depth, riffle/run substrate, and embeddedness. Each site was divided into 10 even transects spaced by 15 m, and depth and substrate were examined at 1.2-m (4 ft) intervals along the wetted width of the channel. Relative abundance of instream and riparian habitat was also estimated between each transect using a standard QHEI protocol. Water quality variables (dissolved oxygen, specific conductivity, water temperature, and pH) were collected instantaneously during each sampling event using an YSI Pro multimeter probe (YSI Inc., Ohio, USA). Additionally, continuous in situ nitrate, temperature, and dissolved oxygen levels were monitored by the U.S. Geological Survey (USGS) and recorded using two monitoring stations located within the restoration reach and near the upstream reference.

2.5 | Fish sampling

To account for variation in seasonal assemblage patterns and reproduction, communities were sampled annually in the early-mid fall at normal (base) water flow and gauge height, concurrently with habitat monitoring. This reduced sampling inefficiency, decreased fish mortality (lower temperature and higher [O₂]), and provided a consistent seasonal assemblage to measure. Block nets (mesh size, 5 mm) were employed during sampling at the upstream and downstream ends of four 200 m sites. Teams of six researchers conducted single-pass electrofishing surveys within each site using standardized protocols (Rabeni, Lyons, Mercado-silva, & Peterson, 2009). We sampled all available habitats within the stream channel and recorded time as a measure of sampling effort. Whenever feasible, fishes were weighed (g), measured (mm), identified to species, and released unharmed near each site. Fishes that were unable to be identified in the field were euthanized using a lethal dose of MS-222, fixed in a 10% formalin solution, and later stored in 75% ethanol before further identification using a taxonomic key (Pflieger, 1997).

Our initial electrofishing protocol (2010–2013) utilized an 8-m AC-electrified seine equipped with two electrodes operating at the terminal ends of a series of copper droppers, a tow barge, and a 2,000-W generator (Bayley, Larimore, & Dowling, 1989). Although the AC seine is a highly effective sampling gear in wadeable Midwestern streams, it may also lead to elevated rates of injury and mortality among stream fish (Bayley et al., 1989; Snyder, 2003). Fish community sampling resumed in 2014 using an advanced DC barge electrofishing unit equipped with three anodes, a 3500 W generator and Infinity electrofishing control box (Midwest Lake Management, Inc., Missouri, USA) used to modulate waveform and power goals (Miranda, 2009). In comparison, DC barge electrofishing has relatively low documented rates of mortality in warmwater fishes (Bardygula-Nonn, Nonn, & Savitz, 1995; Dolan & Miranda, 2004) and low interannual and spatial variation in community sampling (Meador & McIntyre, 2003). In addition, DC electrofishing induces

galvanotaxis, or forced swimming towards the anode, which may help mitigate decreased capture efficiency in deep scour pools. Further, a recent study by Favata et al. (unpublished data) quantified gear selectivity using AC electric seine and DC barge electrofishing in a wadeable Midwestern stream. They found no significant difference in community structure between gear types, and assemblage variation was better explained by spatial dissimilarity rather than sampling gear. By switching to pulsed DC barge electrofishing, we could maintain efficient power goals based on temperature and conductivity (Miranda, 2009) and utilized consistent waveform settings (25% duty cycle; 60-Hz pulse rate) to minimize rates of injury and mortality while sampling an equally robust and diverse assemblage of fish.

2.6 | Fish assemblage response

We calculated an index of biotic integrity specifically developed for this region of Illinois (IBI; Karr, Fausch, Angermeier, Yant, & Schlosser, 1986) to estimate changes in biological health of fish communities following restoration. We quantified differences in IBI scores using 95% confidence intervals to predict average biotic integrity within restored and reference sites (Baldigo et al., 2008). We also assessed fish community changes at multiple organizational levels following restoration. Fishes were assigned to genera to explain changes in taxonomic distribution. To better understand the dynamics driving recruitment, we employed four distinct guild approaches based on the following parameters: (a) functional group where fish are aggregated by taxonomy levels higher than genera that reflect ecosystem function (i.e., black bass, madtom, crappie, darter, herring, minnow, mosquitofish, shiner, silverside, sucker, sunfish, and topminnow); (b) feeding classes (Smith, 1971; Pflieger, 1997) to estimate changes in forage; (c) reproductive guilds (Balon, 1975) to examine specific changes in recruitment strategies; and (d) habitat guilds (Persinger, Orth, & Averett, 2011) to monitor the impacts of altered channel morphology and flow regime on fish communities.

2.7 | Multivariate analyses

As the most robust measure of distance in community ecology (Minchin, 1987), we employed nonmetric multidimensional scaling (NMDS; Faith, Minchin, & Belbin, 1987) using the R Package Vegan (Oksanen et al., 2015). Using a Bray–Curtis dissimilarity matrix of scaled assemblage data across two dimensions, we examined temporal trends in fish community structure within restored sites and compared data to two spatial reference sites. We tested variation in community structure as a factor of time (year post-restoration), treatment type (i.e., restored vs. reference), and time and treatment interaction term using permutational multivariate analysis of variance (perMANOVA; Anderson, 2001).

2.8 | Modelling changes in habitat

Linkages between driving habitat parameters and shifts in fish community structure at the genus and guild levels were analysed using permutational regression analysis with the envfit function within the R Package Vegan (Oksanen et al., 2015). We examined relationships of

25 habitat parameters, which were derived from the QHEI, with the NMDS community matrices. All models were run for 999 permutations. Significant habitat drivers were assessed at $\alpha = 0.01$.

3 | RESULTS

3.1 | Fish assemblage following restoration

During the 6-year study period, 79,013 fishes comprising 46 species, 27 genera, and nine taxonomic families were sampled. Species from five families, Cyprinidae (85.6%), Centrarchidae (5.6%), Percidae (3.8%), Catostomidae (2.3%), and Ictaluridae (1.5%), accounted for more than 98% of the total catch, with nominal contributions from Clupeidae, Poeciliidae, Fundulidae, and Atherinopsidae.

Following implementation of artificial riffles, riprap, scouring keys, and riparian vegetation, we observed distinct temporal and spatial shifts in community structure. Initially, assemblages in all sites were largely composed of tolerant cyprinids from the genera *Notropis* and *Cyprinella*; Sand Shiner, Silverjaw Minnow, and Spotfin Shiner accounted for 55% of all catch. However, 3 years post-restoration, there was a distinct shift in taxonomic distribution. Recruitment of sensitive taxa was detected in the restored reaches, as specified by the relative loadings of genera within the NMDS plot (Figure 2). We sampled an increased relative abundance of darter species from the genera *Etheostoma* and *Percina*, along with sensitive *Moxostoma* fishes in the restoration sites. Restored reaches were also characterized by increased recruitment of habitat-specialist centrarchids belonging to the *Lepomis*, *Micropterus*, and *Pomoxis* genera (Table 1). Results were supported by a perMANOVA, which indicated community structure

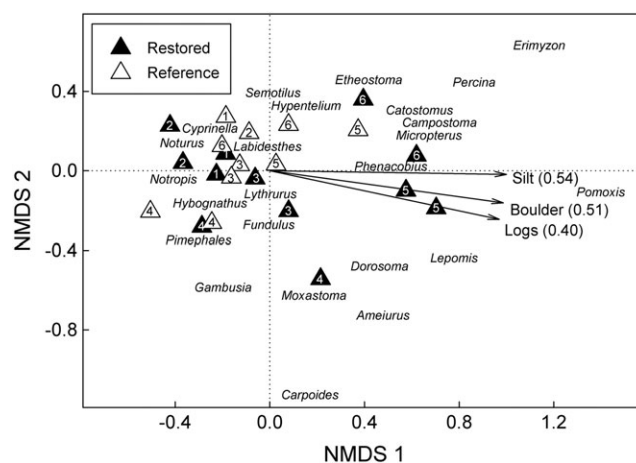


FIGURE 2 Nonmetric multidimensional scaling (NMDS) plot computed with a Bray–Curtis dissimilarity matrix examining temporal and spatial changes in community structure following an instream restoration project in Kickapoo Creek. Fish communities were sampled in restored and reference sites from 2010 to 2015 and numbers within the plot correspond to years post restoration (1–6). Relative loadings of taxonomic groups are represented by genera. Solid vectors represent significant ($\alpha < 0.01$) habitat parameters, with direction and magnitude related to the correlation to the community matrix. Habitat vectors are labelled by variable, with respective permutational R^2 values

TABLE 1 Catch rates and relative abundance of dominant taxa sampled in Kickapoo Creek from 2010 to 2015

Taxonomic group	N	Abundance (%)	Mean CPUE (Fish/hr * km) ± SD	
			Restored	Reference
Cyprinidae	63911	85.58	8382.25 ± 4038.33	14040.82 ± 5856.41
<i>Notropis</i> spp.	28468	38.12		
<i>Cyprinella</i> spp.	15131	20.26		
<i>Pimephales</i> spp.	9642	12.91		
Centrarchidae	4192	5.61	1024.82 ± 649.34	476.96 ± 358.47
<i>Lepomis</i> spp.	4050	5.42		
<i>Micropterus</i> spp.	125	0.17		
<i>Pomoxis</i> spp.	17	0.02		
Percidae	2808	3.76	266.73 ± 175.02	373.19 ± 287.02
<i>Etheostoma</i>	2795	3.74		
<i>Percina</i>	13	0.02		
Catostomidae	1678	2.25	266.73 ± 175.02	373.19 ± 287.02
<i>Moxostoma</i> spp.	560	0.75		
<i>Hypentelium</i> spp.	545	0.73		
<i>Catostomus</i> spp.	517	0.69		
Ictaluridae	1081	1.45	172.52 ± 154.34	192.16 ± 83.21
<i>Noturus</i> spp.	782	1.05		
<i>Ameiurus</i> spp.	299	0.40		

Note. The most prevalent genera for each taxonomic family are represented, whereas mean catch per unit effort (CPUE) data are summarized at the family level.

was significantly influenced by the habitat restoration ($F(1,21) = 5.63$, $R^2 = 0.11$, $p = 0.003$). Variation in data was also strongly driven over a temporal scale ($F(5,21) = 5.58$, $R^2 = 0.55$, $p = 0.001$), despite no significant interaction effects between the restoration treatment and temporal scale.

3.2 | Community response to restoration

Our study indicates that fish recruitment in Kickapoo Creek was significantly driven by stream restoration practices during the 6-year period. We further broke down this analysis by examining changes in recruitment that could be explained by ecological guilds. Fish assemblages aggregated by functional group yielded similar results to taxonomic analyses (effect of restoration— $F(1,21) = 4.89$, $R^2 = 0.12$, $p = 0.004$; effect of year— $F(1,21) = 3.87$, $R^2 = 0.46$, $p = 0.003$; with no significant interaction) and reflected sensitive taxa driving increases in diversity (Figure 2). Reference reaches were consistently dominated by tolerant shiner and minnow species, whereas restored sites had increased recruitment of suckers, madtom, sunfish, black bass, and darter taxa. Although the various types of taxa differed across the study, our analysis of trophic feeding guilds yielded mixed results. Regardless of treatment type, there was significant interannual variation driving the distribution of trophic classes ($F(5,21) = 6.51$, $R^2 = 0.66$, $p = 0.004$). Ultimately, we found no significant linkage between the restoration project and trophic guilds across all sites studied.

The monitoring of reproductive strategies and habitat use highlighted perhaps the most significant impacts on fish recruitment. Results of a perMANOVA revealed significant impacts of treatment type on reproductive guilds of fish ($F(1,21) = 7.34$, $R^2 = 0.12$,

$p = 0.001$). Further, there was a substantial amount of interannual variation within and between treatments ($F(5,21) = 7.46$, $R^2 = 0.58$, $p = 0.001$). However, we found no significant interaction term between treatment and temporal scale. At the start of our study, dominant fish taxa in all sites were classified as nonguarding open substrate spawners, showing little fidelity to specific site conditions (Figure 3). Soon after habitat alteration, fishes in restored reaches showed elevated diversity of reproductive strategies. Increased abundance of guarding nest-spawning taxa (e.g., sunfish, black bass, and crappie taxa) was a strong driver of this model. Relative abundance of nonguarding brood hidiers (e.g., darter taxa) was also a significant driver of assemblage diversity. This was perhaps due to the substantial changes in the morphological characteristics of the restored channel.

Fishes also displayed distinct shifts in habitat use throughout the study. We found significant linkages between treatment type and habitat guilds of stream fish in Kickapoo Creek ($F(1,21) = 4.73$, $R^2 = 0.06$, $p = 0.015$). However, habitat use in fishes was better explained by interannual variability ($F(5,21) = 9.83$, $R^2 = 0.62$, $p = 0.001$). Further, this analysis revealed a significant time * treatment interaction ($F(5,21) = 3.12$, $R^2 = 0.20$, $p = 0.011$). Throughout much of the study, habitat use among fishes was more likely to be similar across treatment types within a given year (Figure 4). As time progressed, restoration sites displayed increased recruitment of riffle-specialists which were previously uncommon to both restored and reference sites.

3.3 | Responses in biotic integrity

As a measure of assemblage health, we found a similar delayed response in biotic integrity following the restoration project. Initially, communities in both restored and reference sites had moderately

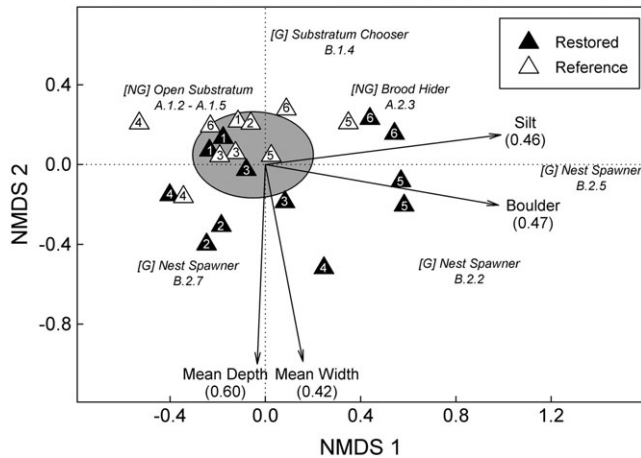


FIGURE 3 Nonmetric multidimensional scaling (NMDS) plot computed with a Bray–Curtis dissimilarity matrix examining temporal and spatial changes in fish reproductive strategies following an instream restoration project. Fish communities were sampled in restored and reference sites from 2010 to 2015 and numbers within the plot correspond to years post restoration (1–6). Relative loadings of fish guilds are represented in the plot. Solid vectors represent significant ($\alpha < 0.01$) habitat parameters, with direction and magnitude related to the correlation to the community matrix. Habitat vectors are labelled by variable, with respective permutational R^2 values. G: guarding; NG: nonguarding strategies

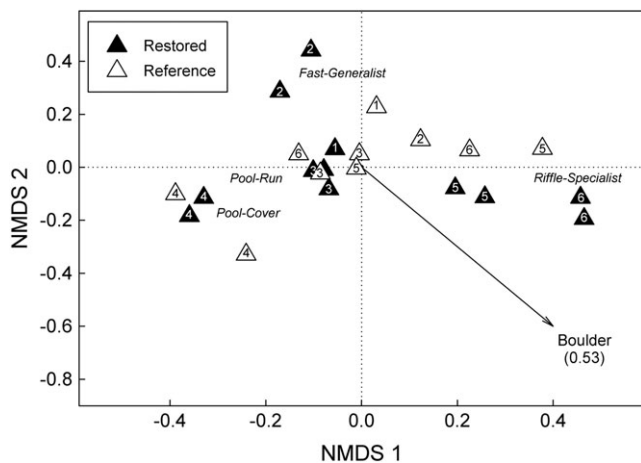


FIGURE 4 Nonmetric multidimensional scaling (NMDS) plot computed with a Bray–Curtis dissimilarity matrix examining temporal and spatial changes in fish habitat guilds following an instream restoration project. Fish communities were sampled in restored and reference sites from 2010 to 2015 and numbers within the plot correspond to years post restoration (1–6). Relative loadings of fish guilds are represented in the plot. Solid vectors represent significant ($\alpha < 0.01$) habitat parameters, with direction and magnitude related to the correlation to the community matrix. Habitat vectors are labelled by variable, with respective permutational R^2 values

low biotic integrity. Whereas IBI scores remained moderately low throughout the study in reference sites, fish communities began responding to the restoration with significant increases within 3 years (Figure 5). Assemblage health was only classified as moderate in the restoration sites, and only 6 years after monitoring. These trends were

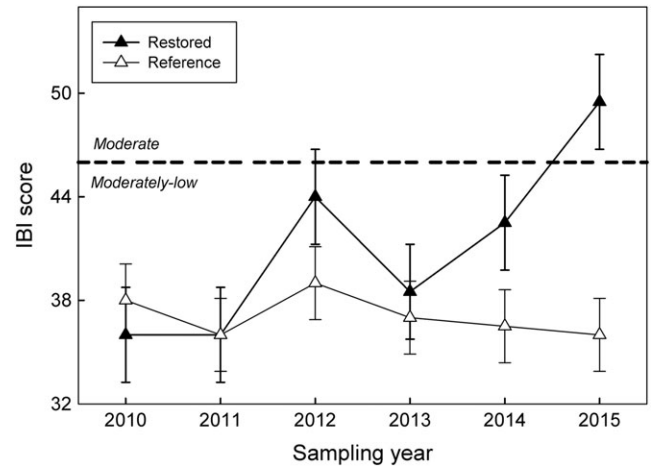


FIGURE 5 Index of biotic integrity (IBI) scores for fish assemblages in restored and reference sites sampled from 2010 to 2015. The horizontal dotted line separates the “moderately-low” classification from “moderate” biotic integrity. Error bars represent 95% confidence intervals for assemblages within each of the treatment types across all sampling years

supported by the clear separation of 95% confidence intervals between treatment types. From 2010 to 2015, average biotic integrity in the restored reaches increased 37.5%, whereas assemblage health in the reference sites decreased by 5%.

3.4 | Habitat drivers of community

Increased habitat diversity was linked to fish community shifts and increased biotic integrity in Kickapoo Creek. Multidimensional scaling at the genus level resulted in significant relationships between substrate and instream structure (Figure 2, Table 1). Addition of large woody debris (Logs; $R^2 = 0.40$, $p = 0.008$), boulders ($R^2 = 0.51$, $p = 0.006$), and silt substrate ($R^2 = 0.54$, $p = 0.002$) to the restoration sites drove temporal increases in the proportion of sensitive taxa. Specifically, habitat-specialists from the *Lepomis*, *Micropterus*, and *Pomoxis* genera benefited from altered channel morphology and increased instream structure provided by the restoration. Catostomids from the genera *Moxostoma*, *Hypentelium*, and *Catostomus* responded more strongly to the increases in vegetation and shifts in substrate. It was also clear that recruitment of fishes was in part due to changes in reproductive strategy. Restoration sites were characterized by the colonization of guarding nest-spawners and nonguarding brood hiders, both of which are heavily dependent on channel morphology and substrate. The proportion of boulders ($R^2 = 0.47$, $p = 0.002$) and silt ($R^2 = 0.46$, $p = 0.007$), along with the mean width ($R^2 = 0.42$, $p = 0.001$) and depth ($R^2 = 0.60$, $p = 0.001$) of the channel were the primary drivers affecting the reproductive strategies of fish in the restoration sites (Figure 3). This contrasts with the tolerant *Cyprinella* and *Notropis* genera that were more abundant prior to restoration. These genera are nonguarding taxa that spawn in open substratum and thus less dependent on habitat diversity for recruitment. The artificial Newbury riffles provided increased boulder abundance and promoted scour-pool hydraulics which increased the mean width and

depth in the restoration site, leaving deep silt-bottom pools. These also promoted the formation of smaller downstream riffles which correlated with increased abundance of darter taxa, which reproduce via brood-hiding strategy. This was evident when examining drivers of habitat guilds, where riffle-specialists keyed in on the increase in large substrate abundance (Boulders; $R^2 = 0.53$, $p = 0.002$) throughout the restoration sites (Figure 4).

4 | DISCUSSION

Our study indicates that reach-scale restoration of instream habitat heterogeneity had strong localized impacts on fish community structure in Kickapoo Creek, despite a long delay in the initial response. Reference fish communities were consistently similar during the long-term study and were largely composed of tolerant Cyprinids with low abundances of sensitive intolerant fishes (e.g., *Noturus spp.*, *Moxostoma spp.*, and *Etheostoma spp.*). Consequently, reference communities displayed moderately low IBI scores, which reflected degraded environmental conditions. Streams and rivers are largely affected by land use, and biotic integrity has been negatively linked with anthropogenic degradation in many systems (Casatti, Langeani, & Ferreira, 2006; Diana, Allan, & Infante, 2006; Lammert & Allan, 1999; Rabeni & Smale, 1995; Roth, Allan, & Erickson, 1996; Snyder, Young, Vilella, & Lemarié, 2003). In the Midwest, agriculture and urban land use are primary drivers of displacement of sensitive taxa (Smith, 1971) and may help explain diminished integrity within references reaches along Kickapoo Creek.

Our hypothesis that restoration of geomorphic stability and habitat heterogeneity would elicit long-term community level biotic response was supported. Initially, assemblages in all sites were largely composed of tolerant cyprinids. However, in a 3-year post-restoration, there was a distinct shift towards sensitive taxa and habitat-specialists, including darter species from the genera *Etheostoma* and *Percina* and *Moxostoma* fishes in the restoration sites. Restored reaches were also characterized by increased recruitment of habitat-specialist centrarchids belonging to the *Lepomis*, *Micropterus*, and *Pomoxis* genera. NMDS plots indicated delayed temporal response to instream habitat restoration. It has been reported in other systems that response to habitat alteration can be delayed up to 10 years (Fitzgerald, Kott, Lanno, & Dixon, 1998), and restoration guides recommend a minimum of 5 years of continuous sampling to accurately assess biotic response post-restoration (Roni, 2005). Comparatively, restored communities in Kickapoo Creek underwent substantial restructuring in a relatively short period, likely due to the size of the system and proximity to the Embarras River that allowed for recruitment.

Fish assemblages aggregated by functional group yielded similar results to taxonomic analyses and reflected sensitive taxa driving increases in diversity. Reference reaches were consistently dominated by tolerant shiner and minnow species while restored sites had increased recruitment of suckers, madtom, sunfish, black bass, and darter taxa. Artificial riffles, such as those used in Kickapoo Creek, were constructed for the threatened Neosho Madtom *Noturus placidus* in the Cottonwood River, Kansas (Fuselier & Edds, 1996).

These structures were quickly colonized by other intolerant benthic invertivores and riffle-specialists (e.g., *Etheostoma spp.* and *Percina spp.*), suggesting potential rapid recovery of sensitive taxa in the presence of high quality habitat. Centrarchids and other nongame fishes that inhabit slower waters have also responded positively to structural mitigation techniques within channelized portions of the Olentangy River near Columbus, Ohio (Edwards, Griswold, Tubb, Weber, & Woods, 1984). Like Kickapoo Creek, fishes in the Olentangy River benefitted from areas mitigated with artificial riffles and pools, and communities displayed beneficial increases in richness and abundance of previously displaced species.

Contrary to our expectations, we found no significant linkage between the restoration project and trophic guilds. This is likely driven by interannual variation in the macroinvertebrate community as it was in flux in response to changes in geomorphology (Ebrahimnezhad & Harper, 1997; Edwards et al., 1984; Harper, Ebrahimnezhad, Climent, & Cot, 1998).

Our study revealed that restoration affected recruitment of new reproductive strategies in the system. Right after restoration, the system was composed mostly of nonguarding open substrate spawners, characterized by little fidelity to specific site conditions. Soon after intervention, there was increased abundance of guarding nest-spawning taxa (e.g., sunfish, black bass, and crappie taxa). Relative abundance of nonguarding brood hiders (e.g., darter taxa) was also a significant driver of assemblage diversity. This was perhaps due to the substantial changes in the morphological characteristics of the restored channel. Increased siltation resulting from erosion and run-off is also known to disrupt sensitive species that require clean gravel for spawning (Berkman & Rabeni, 1987; Wood & Armitage, 1997).

Habitat use was perhaps one of the strongest predictors of restoration over time. Throughout much of the study, habitat use among fishes was more likely to be similar across treatment types within a given year. As time progressed, restoration sites displayed increased recruitment of riffle-specialists which were previously uncommon to both restored and reference sites. Sensitive benthic-dwelling species such as the Northern Hog Sucker *Hypentelium nigricans* and Brindled Madtom *Noturus miurus* were sampled more frequently post-restoration. These fishes tend to aggregate in the tail waters and pools below swift riffles to feed on aquatic macroinvertebrates (Pflieger, 1997). It was apparent that the formation of scour pools and overall increases in habitat heterogeneity led to recovery of Brindled Madtom populations. The recovery of this species was related to increases in overall habitat integrity, confirming the need for mitigation of degraded stream conditions in Kickapoo Creek.

Habitat restoration also promoted significant increases in biotic integrity. In this study, IBI scores in restored reaches exceeded that of reference areas within 6 year post-restoration. Similar marked increases in density, biomass, and diversity of fishes were documented in the North Branch Chicago River in Illinois, although biotic integrity remained considerably lower than rural references due to increased presence of tolerant species (Schwartz & Herricks, 2007). Our results agreed with other studies detailing the effects of instream habitat alteration on fish populations and assemblages (Angermeier & Karr,

1984; Edwards et al., 1984; Fuselier & Edds, 1996; Baldigo et al., 2008; White et al., 2009; Whiteway et al., 2010).

Increased habitat diversity was linked to fish community shifts and increased biotic integrity in Kickapoo Creek. Addition of large woody debris, boulders, and silt substrate to the restoration sites drove temporal increases in the proportion of sensitive taxa. Specifically, habitat-specialists from the *Lepomis*, *Micropterus*, and *Pomoxis* genera benefited from increased instream structure provided by the restoration. Construction of artificial riffles and riprap keys promoted the formation of deep scour pools and increased mean channel depths within the restoration reach. The proportion of boulders and silt, along with the mean width and depth of the channel, were the primary drivers affecting the reproductive strategies of fish in the restoration sites. Fishes in large rivers have responded positively to the increased stability and heterogeneity provided by riprap structures. Channel alteration in the Kansas River led to fine-scale increases in species richness and diversity associated with artificial riprap banks, although woody debris also provided substantial habitat for biota (White et al., 2009). Large woody debris have been previously related to habitat use of warm water stream fish (Angermeier & Karr, 1984; Talmage et al., 2002). In Jordan Creek, Illinois, artificial increases in woody debris promoted increased productivity of macroinvertebrates and provided sufficient refuge for fish to forage and seek cover from predation (Angermeier & Karr, 1984). Additionally, similar benefits of instream habitat restoration have been observed in coldwater systems. Coldwater fishes of the Northeast responded to natural channel design and improved habitat heterogeneity, resulting in an increased community richness and productivity of salmonid species (Baldigo et al., 2008). Moreover, a recent meta-analysis detailing the effects of instream structures on salmonid abundance suggests overall positive impacts from restoration across North America (Whiteway et al., 2010). Although highly dependent on scale and larger confounding factors, it is apparent that instream habitat alterations have the potential to positively impact lotic fish populations and communities in a variety of systems.

Positive relationships between riffles, boulder riprap, and fish biodiversity have been observed at the local scale in other lotic systems (Baldigo et al., 2008; Fuselier & Edds, 1996; White et al., 2009), suggesting coarse substrate may provide stable habitat to support increased biotic integrity in degraded systems. In addition, overhanging and submerged riparian vegetation provided further cover through shading and physical structure and may have also functioned to decrease stream temperatures (Baldigo et al., 2008). Moreover, the establishment of novel microhabitats from instream vegetation benefits several life stages of fish and acts as a buffer to anthropogenic pressure (Lau, Lauer, & Weinman, 2006).

This study shows that the implementation of artificial riffles, scouring keys, and coarse boulder substrate facilitated geomorphic stabilization of Kickapoo Creek. These habitat alterations lead to distinct changes in fish community structure and initial recovery of degraded biotic integrity. We demonstrate that artificial riffles and instream structures employed in channelized warm water streams can effectively mitigate degradation and may also support levels of fish biodiversity which exceeded reference sites sampled in Kickapoo Creek. Revegetation of riparian banks with native grasses provided

potential sources of refuge for juvenile and adult fishes during periods of moderate and high flows. Although woody debris and vegetation are often transient portions of the aquatic environment (Angermeier & Karr, 1984; Reich, Kershner, & Wildman, 2003), these natural structures supported long-term recovery of fishes in Kickapoo Creek and are crucially important to low-gradient streams (Pretty et al., 2003).

Through this long-term study, we emphasize the importance of continuous temporal and spatial reference monitoring to accurately assess the relationships between community structure and instream restoration. Because movement and dispersal of some stream fishes may be limited by connectivity, physical barriers, and home ranges (Gerking, 1959; Berra & Gunning, 1972; Mundahl & Ingersoll, 1983; Matheny & Rabeni, 1995), recolonization of degraded areas may depend largely on distance from source populations. Thus, long-term monitoring is necessary to cover the entire temporal scope of fish community recovery following perturbation. When practical, we recommend multiple or mixed restoration techniques during reach-scale restoration. Because various structures can positively impact habitat conditions and fish populations (Whiteway et al., 2010), it is beneficial to consider the implications of utilizing direct (i.e., altering instream habitat) and indirect (i.e., restoring riparian habitat) practices to improve community diversity. Overall, we demonstrate the ability to use structural restoration as an effective management tool to mitigate loss of biotic integrity through long-term alteration of critical habitat. Given the variation in results among projects, it is imperative to increase the frequency and spatial resolution of monitoring to mitigate further loss.

ACKNOWLEDGEMENTS

Trent Thomas of the Illinois Department of Natural Resources, as well as Tim Straub and Don Roseboom of the U.S. Geological Survey served extensive roles in implementation and monitoring of restoration efforts in Kickapoo Creek. Eastern Illinois University graduate researchers, especially John West, and undergraduate volunteers aided during field collections. The Cole Family and other landowners deserve recognition for their cooperation and collaborations. The Illinois Environmental Protection Agency (Agreement 3191416) provided funds for restoration and along with the Illinois Water Resources Center (Sub-award 2011-03502-07) funded the monitoring project. The monitoring project was also supported by the U.S. Geological Survey and the University of Illinois Urbana-Champaign (Agreement G11AP20077). Eastern Illinois University provided additional funding. The contents of this article are solely the responsibility of the authors and do not necessarily represent the official views of the Illinois Department of Natural Resources, Illinois Environmental Protection Agency, United States Geological Survey, the University of Illinois Urbana-Champaign, or Eastern Illinois University.

CONFLICT OF INTEREST

The authors declare no conflict of interest.

ORCID

Carl A. Favata  <http://orcid.org/0000-0002-4240-7322>

Anabela Maia  <http://orcid.org/0000-0002-8559-2652>
 Manisha Pant  <http://orcid.org/0000-0002-6197-0532>
 Vaskar Nepal  <http://orcid.org/0000-0001-9155-7768>
 Robert E. Colombo  <http://orcid.org/0000-0001-9000-8986>

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How to cite this article: Favata CA, Maia A, Pant M, Nepal V, Colombo RE. Fish assemblage change following the structural restoration of a degraded stream. *River Res Applic.* 2018;34:927–936. <https://doi.org/10.1002/rra.3336>