Fish Community Response to Habitat Alteration: Impacts of Sand Dredging in the Kansas River

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Abstract

Instream dredging, a common practice in rivers worldwide, can affect fish and fish habitat. We investigated the magnitude of these alterations and their influence on the fish community of the Kansas River, a large sand-bed river. Fishes were collected monthly from June 2010 to June 2011 in Edwardsville and Lawrence, Kansas, from 12 reaches of three types: 3 actively dredged, 2 historically dredged (i.e., not been dredged for at least 1 month before sampling), and 7 nondredged control reaches. The reaches, each 1 km long, were sampled with bottom trawls, seines, and electrofishing. Water depths and velocities at all 12 reaches were measured with an acoustic doppler current profiler and interpolated in ArcGIS. Actively dredged reaches had proportionally more deepwater habitat (>3 m) and low velocity (<0.15 m/s) near the riverbed than control reaches. However, the mean proportion of shallow-water habitat (<0.5 m), high velocities near the riverbed (>0.30 m/s), low-velocity habitat (<0.25 m/s), and high-velocity habitat (>0.75 m/s) were similar among all reach types. A canonical correspondence analysis was used to characterize relationships among habitat variables, the three reach types, and CPUE. Mean velocity and depth explained a significant amount of variation in species CPUE; however, reach type was not a significant factor for any of the gear types for any season. Our results show that dredging in Great Plains rivers can increase depths, but alterations to fish community structure was not evident, probably because many of these fishes are adapted to a range of habitat conditions and are highly mobile.

Anthropogenic influences have degraded large rivers and have resulted in species loss and population declines of fish and other aquatic biota (Cross and Moss 1987; Richter et al. 1997; Hughes et al. 2005; Gido et al. 2010) through direct and indirect habitat alterations (Cross and Moss 1987; Meador and Layher 1998; Holčík 2003; Galat et al. 2005; Gerken and Paukert 2009; Falke et al. 2010). Instream sand and gravel dredging is an invasive process that can influence fish and habitat both directly

and indirectly (Kondolf et al. 2002). However, few studies, particularly in sand bed systems, on the biotic response to dredging have been published in the formal literature (Kondolf et al. 2002; Rempel and Church 2009; but see Paukert et al. 2008).

Dredging is common throughout much of the world (Kondolf 1997; Kondolf et al. 2002; Rinaldi et al. 2005; Padmalal et al. 2008; Liu 2009). In 2007 there were 685 dredges operating in the USA, the majority occurring in the south and midwest

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(U.S. Geological Survey, unpublished data). Sediment mined from stream channels is valued for its high quality and low cost, especially where in close proximity to developing areas where construction demand for sediment is high (Kondolf 1997; Meador and Layher 1998; Kondolf et al. 2002; Langer 2003). However, instream sand and gravel dredging directly alters riverine fish habitat through the removal of sediment, which increases turbidity and creates deep pools (Kondolf 1997; Brown et al. 1998; Harvey and Lisle 1998; Meador and Layher 1998; Kondolf et al. 2002; Paukert et al. 2008). The rate of sediment removed from dredging often exceeds the rate it is replenished causing large holes to form at the dredge site. At the leading edge of the hole (the "nickpoint"), the river gradient is increased, resulting in an increase in nickpoint water velocities and erosion upstream (Kondolf 1997; Doyle and Harbor 2003). Water velocities within the dredge hole are decreased (Kondolf 1997; Kondolf et al. 2002; Paukert et al. 2008), allowing suspended sediments to fall out of the water column. The sediment-starved water leaving the dredge hole erodes the tail end of the hole as water velocities begin to increase (Kondolf 1997). The erosion caused by dredging can incise beds, erode banks, reduce the number of sandbars and islands, and undermine bridges and other structures (Kondolf 1997; Meador and Layher 1998; Kondolf et al. 2002; Langer 2003; Rinaldi et al. 2005), all which have potential to impact aquatic biota.

Dredging has the potential to directly impact aquatic biota through entrainment into the dredges (Harvey and Lisle 1998; Hoover et al. 2009, 2011). Although the risk of entrainment is influenced by the proximity of individuals to the sediment intake of a dredge, juvenile paddlefish *Polyodon spathula*, pallid sturgeon *Scaphirhynchus albus*, and lake sturgeon *Acipenser fulvescens* had low risk of entrainment when their distance from the sediment intake of a dredge was greater than 1.25 m (Hoover et al. 2009, 2011).

At large spatial scales (i.e., large geographic distances) indirect impacts from dredging (i.e., habitat alteration) are probably more influential to the fish community than direct impacts through entrainment. Dredging can reduce riffle habitat, leading to reduced predation refugia for small-bodied fishes, thus altering the abundance of fishes in areas impacted by dredging (Brown et al. 1998). Similarly, Hayer and Irwin (2008) documented that the detection probabilities were lower in dredged areas for fish species in the Mobile River basin that prefer riffle habitat and (or) coarse substrate, whereas species that preferred fine substrate were more abundant in dredged areas. Cross et al. (1982) observed that fewer fish were captured at dredge sites and that the species present were more tolerant of lentic conditions and silt substrate (e.g., centrarchids, freshwater drum *Aplodinotus grunniens*, and common carp *Cyprinus carpio*) than other species in the river. Dredging can locally create a reservoir-like habitat, which may be beneficial to species preferring low-velocity and lentic habitats (Kondolf et al. 2002; Paukert et al. 2008). Paukert et al. (2008) concluded that proportionally more lentic species were present in dredge holes than in reference reaches. However, the authors also posited that the high velocity habitat directly upstream of the nickpoint may be beneficial to lotic species (e.g, shovelnose sturgeon *Scaphirhynchus platorynchus* and blue sucker *Cycleptus elongatus*) and that dredging operations may provide multiple habitat types. Thus, the spatial scale at which dredging operations are evaluated may affect the conclusions drawn. Conversely, Harvey (1986) and Rempel and Church (2009) concluded that dredging had no significant effects on fish and macroinvertebrates; thus, the response of fishes to dredging is unclear. Additionally, the majority of these studies were conducted in small, gravel-bed rivers and may not portray the influence of dredging on fish communities in larger, sand-bed rivers, such as those of the U.S. Great Plains.

Sand dredging has occurred in the Kansas River since the early 1900s (Cross et al. 1982) and may have contributed to the decline in abundance and extirpation of species native to the system. Although regulations have been adopted to limit the effects of sand dredging, the five dredges currently active on the river remove approximately 1.7 million tons of sediment each year (U.S. Army Corps of Engineers, unpublished data), which may be affecting native species through habitat alterations. The first objective of this study was to determine if instream dredging was associated with changes in fish habitat in the Kansas River. We hypothesized that dredged reaches would have greater depths and lower velocities, in part due to the expansion of the dredge hole, as in gravel bed streams (Kondolf 1997). The second objective was to determine if abiotic factors (e.g., depth and velocity) affected the fish communities differently in control (no dredging in the last 2 years) versus dredged reaches. We hypothesized that the influence of dredging on fish communities at large spatial scales (≥ 1 km) would be minimal and that abiotic factors would be more influential on fish communities because the size of the dredge holes was small relative to the river and that suitable habitats (e.g., feeding and resting habitat) would be available elsewhere in the reach, even if dredging operations were degrading a species' habitat locally. The third objective was to determine if fish abundances differed at the spatial scale of the dredge hole. We hypothesized that the area directly above the nickpoint of the dredge hole would be dominated by lotic species that may benefit from the fast water velocities in this area. Lentic species would be the dominant fishes within the dredge hole, where water velocities are slower, and that both lotic and lentic species would be present below the dredge hole, where the habitat transitions from a pool to a run.

METHODS

Study area.—The Kansas River is a 274 km 7th-order river, which begins at the confluence of the Smokey Hill and Republican rivers and flows into the Missouri River at Kansas City, Kansas. The river has a mean depth of 1.5 m and a mean width of 164 m (Eitzmann and Paukert 2010a). The dominant substrate is

sand, although gravel patches occur sporadically. No large dams occur on the main-stem Kansas River, but dams have been built on the river's major tributaries (Quist et al. 1999). Additionally, Bowersock Dam (river kilometer [rkm] 85) and Johnson County weir (rkm 18) are two low-head barriers on the Kansas River that create small, reservoir like habitats above each structure (Eitzmann and Paukert 2010a).

Two study locations on the Kansas River in Kansas, Edwardsville (rkm 24–35) and Lawrence (rkm 74–82), were selected based on the presence of active dredges and areas that had been recently dredged, but were not being dredged at the time of the study. Fish and habitat data were collected monthly from these locations from June to October 2010 and March to June 2011. No fish and habitat data were collected from November 2010 to February 2011 due to ice cover. The two locations contained three active dredges: two in Edwardsville (rkm 25 and rkm 33) and one in Lawrence that was operated at rkm 81 from June to August 2010 and at rkm 75 from September 2010 to June 2011. These operations extracted similar amounts of sediment annually (U.S. Army Corps of Engineers, unpublished data). From the two locations we selected 12 study reaches (each 1 km long), and each reach was categorized as one of three dredging-activity types: (1) seven were control reaches (no records of any dredging), three were actively dredged reaches (dredged at the time of study), and two were historically dredged reaches (not dredged at the time of our study but had been dredged between 2008 and 2 months prior to study). The Edwardsville site was comprised of two actively dredged, one historically dredged, and four control reaches, and the Lawrence site was comprised of one actively dredged, one historically dredged, and three control reaches (Figure 1). Control reaches were located near dredged reaches to minimize differences in other factors influential on the fish fauna (e.g., urbanization and channel form). This design minimized the possibility of the results being confounded by longitudinal differences in the fish community (Eitzmann and Paukert 2010a).

Collection of habitat variables.—A Sontek/YSI M9 River-Surveyor acoustic Doppler current profiler (ADCP) was used to measure water depths, mean velocity of the water column, and velocities within 1 cm of the riverbed monthly and within 2 d of fish sampling. However, equipment failure prevented habitat measurements in September 2010 for both locations and in April 2011 for Lawrence; the corresponding fish data were removed from analysis because habitat variables could not be included in the community analyses for these months. Continuous measurements of these variables were made along a series of zigzag transects at about 200-m intervals (about five per reach) in the upstream direction. The bottom shear velocity equation (Son-Tek/YSI March 2011, equation 1) was used to calculate near-bed velocities (u_*):

$$u = \frac{u_*}{9.5\left(\frac{z}{k_s}\right)^{\frac{1}{6}}}, \qquad (1)$$
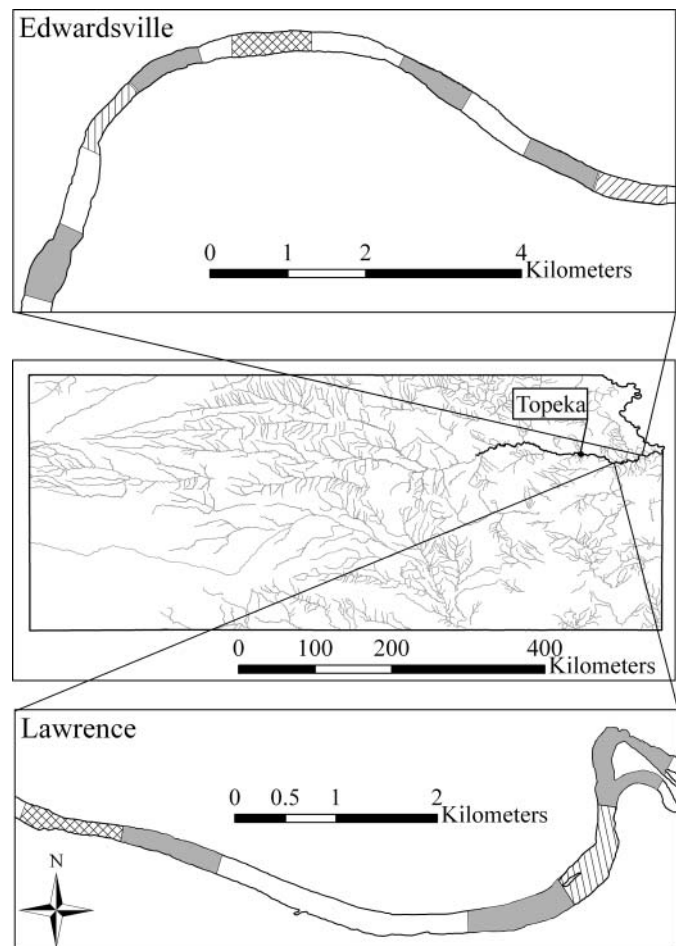

FIGURE 1. Reaches on the Kansas River near Edwardsville and Lawrence Kansas where fish and habitat sampling was conducted. The Topeka, Kansas, location where local-scale collections occurred is also shown. Gray shading represents control reaches, hatching represents actively dredged reaches, and historically dredged reaches are represented by cross hatching (unfilled areas were not studied). Flow is from west to east.

where z is the distance from the bed in meters, u is the velocity (m/s) at z, and k_s is the bed roughness height. The ADCP recorded variables z and u and k_s was calculated following the methods of van Rijn (1984) and Gaeuman and Jacobson (2006), using data collected by the ADCP and sediment diameters obtained from Simons, Li and Associates (1984). Additionally, the competent bottom velocities (V_c, equation 2; U.S. Bureau of Reclamation 1977) required to transport sediment diameters of the 50th (0.99 mm) and 90th percentiles (3.87 mm; Simons, Li and Associates 1984) were calculated to help identify areas of deposition and scour.

$$V_c = 0.155d^{1/2}, \qquad (2)$$

where d is the sediment diameter (mm).

All data collected with the ADCP were kriged in ArcGIS 9.3.1 using ordinary kriging via a spherical semivariogram

model with five neighbors. This was done to avoid biasing the data towards areas with slower water velocities because the ADCP must be moving at the same speed or less than the water velocity during data collection. Mean water depths, velocities, and near-bed velocities were obtained from the kriged data for each reach. The CV of water depths, velocities, and near bed velocities were also calculated for each reach.

Reach scale collections.—Boat electrofishing (pulsed DC) was used following the methods presented by Guy et al. (2009) and Miranda (2009) to collect large-bodied fishes within each 1-km reach. A stratified random design was used to determine the bank side (left or right) of the starting location of an electrofishing site in the upper, mid, and lower portion of each reach. Each electrofishing sample lasted 300 s with the boat moving downstream (Eitzmann et al. 2007), and CPUE was calculated as fish collected per hour. Benthic fishes were collected at three sites within each reach using a Gerken siamese trawl with a 2.5-m headrope, 2.9-m footrope, 38-mm outer mesh, and a 4-mm innermesh fish separator at the cod end. Trawling sites in each reach were selected using the stratified random design previously mentioned and were field-verified to ensure they were clear of obstructions (e.g, submerged trees) and were at least 2 m deep; sites that did not meet these criteria were moved to the nearest location that met the criteria. The trawl was attached to the bow of the boat with a 30.5-m rope and towed downstream for 50–140 m, depending on the length of obstruction-free habitat of appropriate depth (Guy et al. 2009); CPUE was calculated as fish/m trawled. Small-bodied and juvenile fishes were collected using 25-m seine (4.5 × 1.2 m, 6.4-mm mesh) hauls; CPUE for seine samples was calculated as fish/m^2. Three seining sites within each reach were selected based on suitable habitat (i.e., gradually sloped banks in depths <1.2 m that were free of large rocks or large woody debris). Fish were not collected with seining during high flows (July 2010) nor via trawling during low flows (March–June 2011), and equipment failure prevented electrofishing and trawling in Lawrence for June 2010.

Fish collected via trawling and electrofishing were identified to species, measured, weighed if over 150 mm, and released. Large sample sizes in seine hauls (i.e., >100 fish) prevented identification of fish in the field; therefore, specimens were preserved in 10% formalin and later identified and measured in the laboratory. Samples containing more than 500 individuals were sorted to species; predominant species were split in half and subsampled, and rare species were sampled in their entirety. Large fish collected via seining were processed in the field following the protocol for fish collected from electrofishing or trawling.

Local scale collections.—To determine fish distribution in relation to dredge holes, additional local-scale fish collections were conducted on 23 July, 7 August, and 28 August 2011 within dredge holes on the Kansas River in Lawrence (rkm 75) and Topeka, Kansas (rkm 122). Edwardsville dredge holes were excluded from these collections because one of the Edwardsville

dredges became inoperable at the time of collections, causing the other to operate sporadically. A dredge hole was defined as the region from the nickpoint at the upstream end of the hole to the downstream location, where depths were similar to those upstream of the nickpoint. One trawl haul was conducted above the dredge hole (starting one dredge-hole length upstream and ending at the nickpoint), within the dredge hole, and below the dredge hole (starting at the downstream end of the dredge hole and ending one dredge-hole length away). Fish were sampled with a bottom trawl for hauls lengths similar to that of the dredge hole (50–100 m) in each location. All fish were processed with the protocol discussed previously.

Statistical analysis.—Differences in water depths, velocities, and near bed velocities between reach types were assessed by determining the proportion of areas in each reach type (active dredge, historical dredge, and control) that were shallow water (<0.5 m), deep water (>3 m), slow water (<0.25 m/s), fast water (>0.75 m/s), had near-bed velocity incapable of moving sediment sizes at or above the 50th percentile (<0.15 m/s), and had near-bed velocity capable of moving sediment sizes up to the 90th percentile (>0.30 m/s). An ANCOVA on arcsine-square-root-transformed proportions tested whether mean proportion of aquatic habitat in the above categories differed by reach type with season (spring: March–May; summer: June–July; fall: August–October) and location as covariates.

A canonical correspondence analysis (CCA) was used to determine how fish abundance was related to mean depth, mean velocity, mean near-bed velocity, maximum depth, maximum velocity, maximum near-bed velocity, CV of depth, CV of velocity, CV of near-bed velocity, location, season, reach type, and an interaction of season and reach type. All analyses were conducted using program R, version 2.11.1, using the package "vegan." Because standardizing effort among gear types is not practical (Quist et al. 2009), each gear was analyzed with a separate CCA. To account for ontogenetic shifts in life history, species were separated into adult and juvenile life stages based on total length categories from the literature (Table 1). However, small-bodied species (i.e., small cyprinids and percids) were not divided into life stages due to their rapid growth and early age of maturity. To reduce the influence of predominant species on the analyses, CPUE data were log-transformed, $\log_e(x + 1)$ (Ter Braak 1995; Anderson and Willis 2003). In addition, rare species (less than three observations) were removed from the analysis due to their influence in a transformed data set (Ter Braak 1995). Lastly, samples where no fish were observed (one electrofishing sample and three trawl samples) were removed along with their corresponding habitat variables. Variables that were highly collinear (variance inflation factors > 10) were then iteratively removed from and added to the model to find a set of variables that explained the largest amount of variation that were not autocorrelated. A Monte Carlo simulation was then run with 1,000 permutations to determine which CCA axes and variables explained a significant ($\alpha = 0.05$) proportion of the variation in fish community structure.

TABLE 1. Total length at maturity for species collected in the Kansas River, 2010–2011. Species which were not distinguished as adult or juvenile are denoted with a na.

Species	Species code	Length at maturity (mm)
Bighead carp *Hypophthalmichthys nobilis*	BCP	620[a]
Bigmouth buffalo *Ictiobus cyprinellus*	BBF	380[b]
Black buffalo *Ictiobus niger*	BLF	309[c]
Black bullhead *Ameiurus melas*	BBH	400[b]
Black crappie *Pomoxis nigromaculatus*	BKC	180[b]
Blue catfish *Ictalurus furcatus*	BCF	500[b]
Blue sucker *Cycleptus elongatus*	BSU	500[b]
Bluegill *Lepomis macrochirus*	BGS	80[d]
Bluntnose minnow *Pimephales notatus*	BNM	na
Bullhead minnow *Pimephales vigilax*	BHM	na
Central stoneroller *Campostoma anomalum*	CSR	na
Channel catfish *Ictalurus punctatus*	CCF	250[b]
Common carp *Cyprinus carpio*	CRP	300[b]
Creek chub *Semotilus atromaculatus*	CCB	na
Emerald shiner *Notropis atherinoides*	ESH	na
Fathead minnow *Pimephales promelas*	FHM	na
Flathead catfish *Pylodictis olivaris*	FHC	380[b]
Freshwater drum *Aplodinotus grunniens*	FDR	275[b]
Gizzard shad *Dorosoma cepedianum*	GZS	200[b]
Goldeye *Hiodon alosoides*	GOE	350[b]
Grass carp *Ctenopharyngodon idella*	GCP	510[e]
Green sunfish *Lepomis cyanellus*	GSF	64[f]
Johnny darter *Etheostoma nigrum*	JHD	na
Largemouth bass *Micropterus salmoides*	LMB	250[b]
Logperch *Percina caprodes*	LGP	na
Longear sunfish *Lepomis megalotis*	LES	40[e]
Longnose gar *Lepisosteus osseus*	LNG	700[b]
Orangespotted sunfish *Lepomis humilis*	OSF	45[b]
Orangethroat darter *Etheostoma spectabile*	OTD	na
Quillback *Carpiodes cyprinus*	QUI	275[b]
Red shiner *Cyprinella lutrensis*	RSH	na
River carpsucker *Carpiodes carpio*	RCS	275[b]
Rosyface shiner *Notorpis rubellus*	RYS	na
Sand shiner *Notropis stramineus*	SSH	na
Shoal chub *Macrhybopsis hyostoma*	SCB	na
Shorthead redhorse *Moxostoma macrolepidotum*	SRH	225[b]
Shortnose gar *Lepisosteus platostomus*	SNG	375[b]
Shovelnose sturgeon *Scaphirhynchus platorynchus*	SST	540[b]
Silver carp *Hypophthalmichthys molitrix*	SCP	530[g]
Slender madtom *Noturus exilis*	SMT	na
Slenderhead darter *Percina phoxocephala*	SHD	na
Smallmouth bass *Micropterus dolomieu*	SMB	250[b]
Smallmouth buffalo *Ictiobus bubalus*	SBF	375[b]
Stonecat *Noturus flavus*	STC	na
Striped bass *Morone saxatilis*	STB	610[b]
Striped bass × white bass *Morone chrysops*	WIP	225
Suckermouth minnow *Phenacobius mirabilis*	SMM	na
Walleye *Sander vitreus*	WAL	300[b]
Western mosquitofish *Gambusia affinis*	WMF	na
White bass *Morone chrysops*	WBS	225[b]
White crappie *Pomoxis annularis*	WCP	180[b]

[a]Schrank and Guy 2002; [b]Pfieger 1997; [c]Houston 1990; [d]Hoxmeier et al. 2009; [e]Bonner 2011; [f]Trautman 1981; [g]Williamson and Garvey 2005.

Fish observed during the local-scale collections were analyzed two ways depending on frequency of occurrence of the species collected. A repeated-measures multiple analysis of variance (MANOVA) tested whether mean CPUE (log transformed) of frequently observed species (i.e., observed in over half the samples) differed among samples above, within, and below dredge holes, using location as the repeated measure. When species were collected in <50% of the samples, logistic regression was used to determine whether the proportion of sites with a particular species present differed above, within, and below a dredge hole.

RESULTS

Proportional Analysis of Habitat

Mean velocities ranged from 0.18 m/s in an actively dredged reach in Edwardsville to 0.66 m/s in an actively dredged reach in Lawrence. The maximum water velocity was 2.68 m/s in a control reach in Edwardsville (Table 2). The maximum near-bed velocity was 1.06 m/s in a control reach in Edwardsville and mean near-bed velocities ranged from 0.04 m/s in an actively dredged reach in Edwardsville to 0.16 m/s in a control reach in Lawrence. Mean depths ranged from 0.73 m in an Edwardsville control reach to 2.64 m in an actively dredged reach in Lawrence. The maximum depth observed was an 11.71 m scour hole near an armored bank in a control reach in Lawrence.

There were significant differences between locations for the proportion of fast-water and low near-bed velocity habitat and among seasons for the mean proportion of habitats, except for low near-bed velocities. Additionally, there was a significant month $\times$ location interaction for the mean proportion of deep-water habitats but no significant interactions included reach type (Table 3). The mean proportion of shallow water habitat (<0.5 m) in a reach ranged from 0.26 to 0.33 and was similar among reach types ($F_{2,67} = 0.56$, $P = 0.58$). However, the mean proportion of deep-water habitat (>3 m) was greater in actively dredged reaches (mean = 0.16, SE = 0.04) than in control (mean = 0.04, SE = 0.01) and historically dredged reaches (mean = 0.07, SE = 0.04; $F_{2,67} = 4.46$, $P = 0.01$; Figure 2). The mean proportion of low-velocity water (<0.25 m/s) in reaches was similar among all reach types ($F_{2,67} = 0.22$, $P = 0.80$), as was the mean proportion of high-velocity water (>0.75 m/s; $F_{2,67} = 0.62$, $P = 0.54$; Figure 2). The proportion of habitat with near-bed velocities incapable of moving sediment sizes at or above the 50th percentile (<0.15 m/s) ranged from 0.75 to 0.87, actively dredged reaches having a higher mean proportion (mean = 0.87, SE = 0.03) than control (mean = 0.75, SE = 0.03) and historically dredged reaches (mean = 0.77, SE = 0.04; $F_{2,67} = 3.09$, $P = 0.05$; Figure 2). The proportion of habitat with near-bed velocities capable of moving sediment sizes up to the 90th percentile (>0.30 m/s) was similar among reach types ($F_{2,67} = 0.58$, $P = 0.56$; Figure 2).

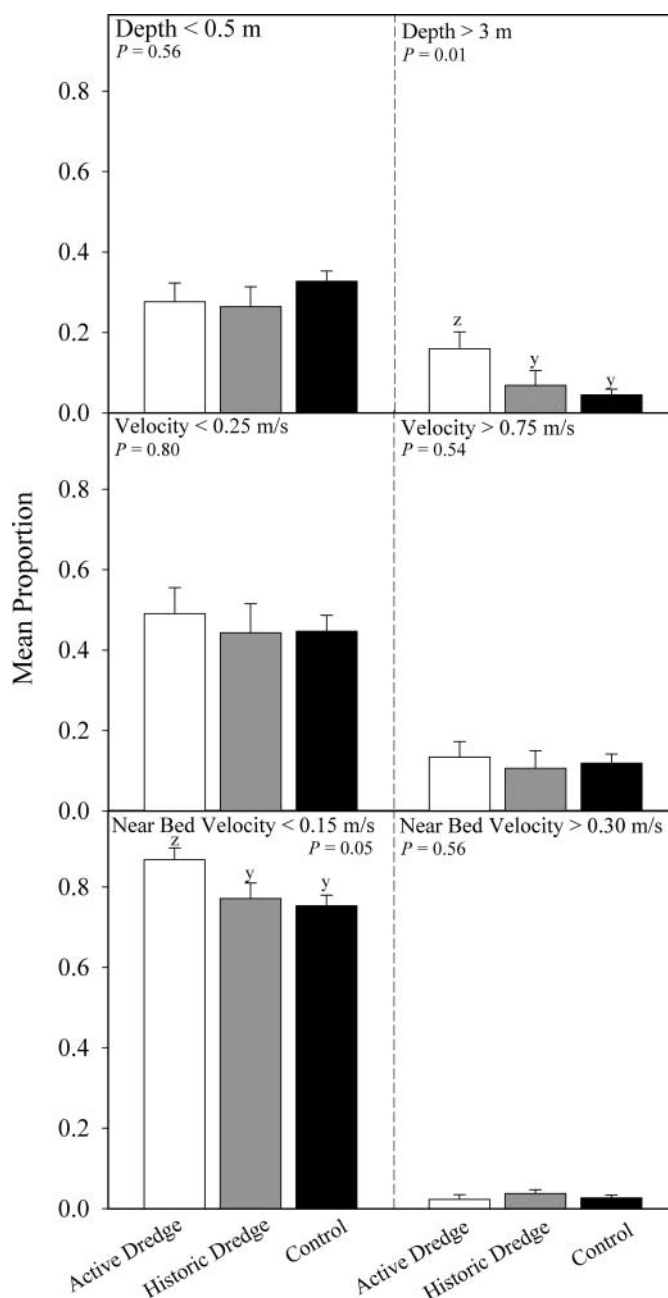FIGURE 2. Mean proportion of shallow water (<0.5 m), deep water (>3 m), slow water (<0.25 m/s), fast water (>0.75 m/s), near-bed velocities incapable of moving sediment sizes at or above the 50th percentile (<0.15 m/s), and near-bed velocities capable of moving sediment sizes up to the 90th percentile (>0.30 m/) habitat in actively dredged, historically dredged, and control reaches (see Figure 1). Disparate letters above error bars (1 SE) denote significant difference; bars without letters did not differ.

Reach-Scale Collections

A total of 23,879 fish (51 species) were collected over the course of the study including three of these species listed as in need of conservation in the state of Kansas. All three species,

TABLE 2. Metrics describing habitat variables included in the canonical correspondence analysis models for control (Con.), actively dredged (A.D.), and historically dredged reach types (H.D.) in each Kansas River location, June 2010 to June 2011. River kilometers (rkm) are from the confluence of the Kansas River with the Missouri River and increases upstream. The dredge in reach 12 moved to reach 9 in September or 2010; thus, mean values are presented for both reaches pre and post movement.

Reach (rkm)	Reach type	Mean (maximum; CV)		
		Near bed velocity (m/s)	Velocity (m/s)	Depth (m)
Edwardsville				
1 (rkm 25.2)	A.D.	0.04 (0.57; 0.83)	0.18 (0.86; 0.65)	1.63 (7.91; 0.96)
2 (rkm 26.1)	Con.	0.06 (0.39; 0.61)	0.22 (0.90; 0.66)	1.33 (4.28; 0.65)
3 (rkm 28.1)	Con.	0.06 (0.30; 0.89)	0.24 (2.68; 2.08)	1.24 (7.18; 0.81)
4 (rkm 30.3)	H.D.	0.10 (0.67; 0.61)	0.33 (1.32; 0.56)	1.15 (6.82; 1.35)
5 (rkm 31.7)	Con.	0.12 (0.57; 0.62)	0.39 (1.60; 0.55)	1.19 (6.32; 0.62)
6 (rkm 32.7)	A.D.	0.12 (0.81; 0.73)	0.43 (1.80; 0.72)	1.39 (8.51; 0.71)
7 (rkm 34.8)	Con.	0.14 (1.06; 0.76)	0.37 (1.71; 0.84)	0.73 (6.10; 0.90)
Lawrence				
8 (rkm 74.4)	Con.	0.11 (0.98; 0.63)	0.41 (2.05; 0.78)	1.13 (11.71; 0.95)
9 (rkm 75.5)	A.D.	0.11 (0.75; 0.96)	0.21 (1.08; 1.05)	0.91 (7.11; 1.07)
9 (rkm 75.5)	H.D.	0.14 (0.48; 0.46)	0.59 (1.40; 0.46)	1.80 (7.72; 0.60)
10 (rkm 76.5)	Con.	0.16 (0.95; 0.56)	0.47 (1.29; 0.62)	0.99 (4.30; 0.65)
11 (rkm 80.1)	Con.	0.12 (0.60; 0.56)	0.41 (1.24; 0.59)	1.22 (5.14; 0.63)
12 (rkm 81.1)	A.D.	0.09 (0.42; 0.55)	0.66 (1.45; 0.53)	2.64 (6.93; 0.56)
12 (rkm 81.1)	H.D.	0.08 (0.50; 0.82)	0.22 (1.16; 0.83)	0.87 (5.57; 0.91)

blue sucker, shoal chub, and johnny darter were observed in each reach type.

The CCA for fishes collected with electrofishing (Figure 3) explained 30.8% of the variation in the CPUE, and the first five axes were significant ($P < 0.05$; Table 4). Location, season, and mean velocity were significant explanatory variables; however, reach type ($P = 0.53$) and the season × reach type interaction ($P = 0.71$) were not (Table 4). Adult silver carp, blue catfish, and adult smallmouth buffalo were associated with high mean water column velocities; whereas white bass, adult black buffalo, and juvenile silver carp were associated with low mean water column velocities (Figure 3). The CCA model for fishes collected with

seining explained 46.4% of the variation in CPUE, and the first five axes were significant. Reach type ($P = 0.54$) and the season × reach type interaction ($P = 0.62$) did not explain a significant amount of variation in the model, but season, mean near bed velocity, and mean depth were significant variables. A number of fishes, such as juveniles of walleye, goldeye, and largemouth bass, were associated with lower near bed velocities (Figure 3). The CCA model for benthic fishes collected with trawling explained 40.0% of the variation, and the first three axes were significant. Location and mean bottom velocity were the only significant variables in the model, and reach type had no significant influence on CPUE. Shoal chub and shovelnose

TABLE 3. P-values and F-values (in parentheses) from ANCOVA comparisons of the proportion of low-velocity (<0.25 m/s), fast-velocity (>0.75 m/s), low near-bed velocity (<0.15 m/s), fast near-bed velocity (>0.30 m/s), shallow-water (<0.5 m), and deep-water (>3 m) habitats by reach type (actively dredged, historically dredged, and control reaches), locations (Edwardsville or Lawrence), and season (month) in the Kansas River from June 2010 to June 2011.

| Effect	df	Mean velocity		Near-bed velocity		Depth	
		Low	Fast	Low	Fast	Shallow	Deep
Type	2, 67	0.80 (0.22)	0.54 (0.62)	0.05 (3.09)	0.56 (0.59)	0.56 (0.58)	0.01 (4.46)
Location	1, 67	0.08 (3.07)	0.05 (3.99)	0.02 (5.69)	0.28 (1.18)	0.68 (0.17)	0.52 (0.42)
Type × location	2, 67	0.79 (0.24)	0.90 (0.10)	0.61 (0.49)	0.22 (1.53)	0.87 (0.14)	0.91 (0.09)
Season	2, 67	<0.01 (9.53)	<0.01 (9.21)	0.15 (1.93)	0.05 (3.06)	<0.01 (6.79)	<0.01 (9.33)
Type × season	4, 67	0.98 (0.11)	0.88 (0.29)	0.98 (0.11)	1.00 (0.04)	0.60 (0.70)	0.67 (0.59)
Location × season	2, 67	0.49 (0.72)	0.24 (1.46)	0.72 (0.33)	0.61 (0.49)	0.98 (0.02)	0.01 (4.52)
Type × location × season	4, 67	1.00 (0.03)	0.97 (0.14)	0.98 (0.09)	0.77 (0.45)	0.93 (0.21)	0.77 (0.45)
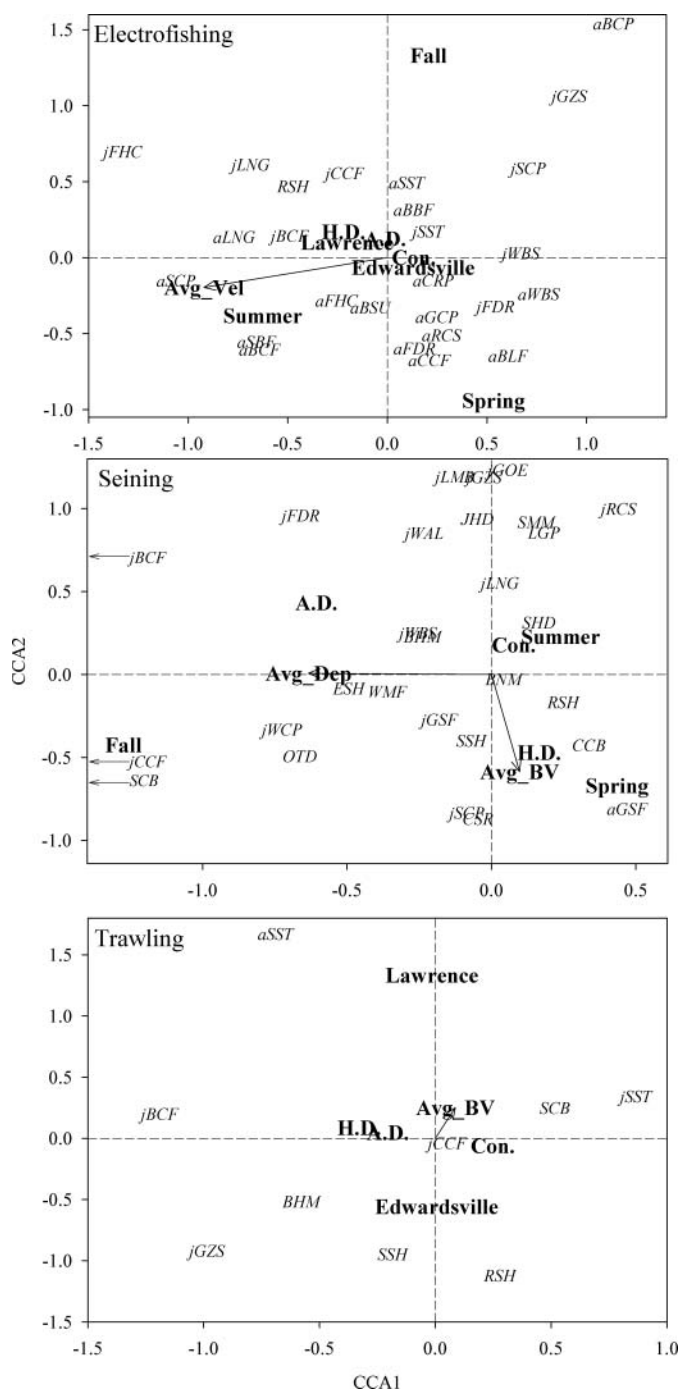
FIGURE 3. Canonical correspondence analysis of the $\log_e(x+1)$-transformed CPUE of large-bodied fishes collected with electrofishing, small-bodied and juvenile fishes collected with seining, and benthic fishes collected with trawling for the first two axes. Species are represented by italic lettering, a lower case a in front of the species code (Table 1) represents adults, and a lower case j represents juveniles; species not classified as adult or juvenile have no letter before their code. Only the significant ($P < 0.05$) habitat variables listed in Table 4 and reach type are shown. Variables included are dredged reaches (A.D.), historically dredged reaches (H.D.), control reaches (Con.), season (spring, summer, and fall) location (Lawrence or Edwardsville), mean near-bed velocity (Avg_BV), mean velocity (Avg_Vel), and mean depth (Avg_Dep).

sturgeon were associated with higher mean near-bed velocities, whereas juvenile gizzard shad, bullhead minnow, red shiners, and sand shiners were associated with lower mean near-bed velocities (Figure 3).

Local-Scale Collections

A total of 495 individuals (11 species) were collected with 18 bottom trawl hauls as part of the local scale collections; however, 91% (450) of these fish were juvenile blue catfish and channel catfish. Thus, only blue catfish and channel catfish catch rates were analyzed with a repeated-measures MANOVA. The other nine species (bullhead minnow, common carp, flathead catfish, freshwater drum, river carpsuckers, red shiners, sand shiners, shovelnose sturgeon, and suckermouth minnow; Table 5) were analyzed using presence–absence data and logistic regression models. Mean CPUE did not differ for sites above, in, or below the dredge hole for blue catfish ($F_{2,15} = 0.63$, $P = 0.55$) or channel catfish ($F_{2,15} = 1.91$, $P = 0.18$). The proportion of trawls that collected flathead catfish or freshwater drum ranged from 0.3 to 0.5 but did not differ among position from the dredge hole ($\chi^2 = 0.35$, $P = 0.84$). Similarly, the proportions of other species infrequently collected did not differ among position from the dredge hole (i.e., samples above, in, or below the dredge hole; Table 5).

DISCUSSION

Deep-water habitat represented a greater proportion of actively dredged reaches than did control and historically dredged reaches, which was probably due to an expansion of the dredge hole (Cross et al. 1982) and not a direct result of a deep hole created by sediment removal because control reaches also had deep holes. This has also been documented in gravel-bed systems where dredged areas have greater depths due to headcutting and erosion of the dredge hole downstream (Kondolf 1997; Kondolf et al. 2002; Rinaldi et al. 2005). However, the proportion of shallow-water habitat ranged from 0.02 to 0.81, regardless of reach type, indicating that dredging did not influence the amount of shallow-water habitat, at least at the 1-km scale. The lack of differences in the proportion of slow-water and fast-water velocities among reach types may pertain to the localized influence of dredging on water velocities. Although water velocities increased at the nickpoint and tail end of the dredge hole and were reduced within the dredge hole (Kondolf 1997), these changes were restricted to the extent of the dredge hole, minimizing the influence dredging had on velocities at the 1-km reach scale. The proportion of low velocities near the riverbed was greatest in actively dredged reaches, indicating that these reaches had more deposition than historically dredged and control reaches. Low water velocities within a dredge hole allow sediment to be deposited and the removal of sediment through dredging prevents the accumulation of deposited sediments (Kondolf 1997), allowing greater depths to be maintained. This supports our observation of a higher proportion of deep water in actively dredged reaches and

TABLE 4. The first five axes of canonical correspondence analysis (CCA) models and the variables included in the models to associate fishes and in-stream habitat in the Kansas River, June 2010 to June 2011 for each gear type. The F-values (parentheses) and P-values were derived from a Monte Carlo simulation with 1,000 permutations. Mean and maximum velocities were not included in the CCA model for trawling.

Variable/axis	Electrofishing	Seine	Trawl
Reach type	0.55 (0.98)	0.55 (1.82)	0.89 (0.48)
Season	<0.01 (3.40)	0.01 (5.37)	0.12 (2.09)
Reach type × season	0.76 (0.91)	0.66 (1.41)	0.98 (0.35)
Location	0.03 (1.91)	0.14 (2.78)	0.02 (2.87)
Maximum near-bed velocity	0.62 (0.94)	0.11 (2.84)	0.72 (0.59)
Mean near bed velocity	0.45 (0.96)	0.02 (4.89)	0.04 (3.13)
CV of near bed velocity	0.61 (0.91)	0.10 (2.72)	0.91 (0.25)
Maximum velocity	0.82 (0.69)	0.57 (1.51)	
Mean velocity	0.01 (4.26)	0.43 (1.75)	
CV of velocity	0.91 (0.58)	0.71 (0.57)	0.27 (1.51)
Maximum depth	0.08 (1.51)	0.08 (2.66)	0.27 (1.61)
Mean depth	0.43 (1.03)	0.02 (5.37)	0.35 (1.20)
CV of depth	0.75 (0.68)	0.26 (1.65)	0.89 (0.42)
CCA1	<0.01 (5.91)	<0.01 (14.30)	<0.01 (6.92)
CCA2	<0.01 (4.71)	< 0.01 (9.26)	0.02 (4.02)
CCA3	<0.01 (3.28)	< 0.01 (5.97)	0.03 (3.32)
CCA4	<0.01 (2.41)	<0.01 (5.38)	0.20 (1.14)
CCA5	0.02 (1.84)	0.02 (4.24)	0.38 (1.07)

prior inference of dredging-induced bed incision on the Kansas River (Simons, Li and Associates 1984), despite a greater proportion of slow near-bed velocities conducive to sediment deposition.

The lack of differences in habitat between historically dredged reaches and control reaches may suggest rapid recovery for reaches where dredging operations have ceased. The high mobility of sand allows for a quick recovery of sand-bed systems from degradation (Doyle and Harbor 2003), which we observed shortly after our study was concluded; i.e., the upstream dredge in Edwardsville halted operation, and the dredge hole completely filled with sediment within a month. Similarly, Martin and Hess (1986) reported that a dredge hole on the Chattahoochee River, Georgia, filled with sediment 2 weeks after the dredging operation ceased, and Rempel and Church (2009) reported that gravel bars removed through dredging were restored following a high-flow event on the Fraser River, British Columbia. Therefore, the elimination of dredging may allow quick recovery of sediment at the reach scale. However, the sediment that fills in abandoned dredge holes is most likely derived from headcutting and bank erosion upriver (Kanehl and Lyons 1992; Kondolf 1997), causing degradation upriver to continue after dredging operations have ceased and the dredged reach has recovered.

TABLE 5. Results of the logistic regression analysis for species collected infrequently at the local-scale study, June to August 2011. Proportions are out of the total number of trawl hauls above, within, and below dredge holes. Sample size (N) = 6 and df = 14 for all analyses.

Species	Proportion			χ^2	P
	Above	Within	Below		
Bullhead minnow	0.00	0.17	0.33	0.43	0.80
Common carp	0.00	0.00	0.17	0.01	1.00
Freshwater drum	0.00	0.33	0.50	0.35	0.84
Flathead catfish	0.00	0.33	0.50	0.35	0.84
River carpsucker	0.17	0.17	0.17	<0.01	1.00
Red shiner	0.17	0.17	0.00	<0.01	1.00
Suckermouth minnow	0.00	0.00	0.17	0.01	1.00
Sand shiner	0.17	0.00	0.00	0.01	1.00
Shovelnose sturgeon	0.33	0.17	0.00	0.43	0.80

The majority of habitat variables measured did not differ between locations, yet location explained a significant amount of variation in CPUEs for fishes collected with electrofishing and trawling. This is likely because the Edwardsville location is near Kansas City and the confluence with the Missouri River, both which may affect the regional pool of species contributing to this location. Additionally, Bowersock Dam was located just upriver of the Lawrence location, which may have influenced the fish community in those reaches, similar to results reported by Eitzmann and Paukert (2010a).

There was evidence for significant relationships between CPUEs and near-bed velocities and water-column velocities. For all gear types, species preferring lotic environments (e.g., blue catfish, shovelnose sturgeon, and shoal chub; Pflieger 1997; Eitzmann and Paukert 2010a) tended to be associated with high mean water-column and near-bed velocities. Species preferring lentic environments or habitat generalists (e.g., white bass, walleye, and centrarchids; Pflieger 1997; Eitzmann and Paukert 2010a) tended to be associated with low mean water-column and near-bed velocities. However, there were a few exceptions. Shovelnose sturgeon collected with electrofishing were not strongly associated with mean water-column velocity, probably because this species may be more strongly influenced by near-bed velocities (Quist et al. 1999), as was observed in our trawling samples. Additionally, adult green sunfish were associated with areas of high near-bed velocity and shallow depths, particularly during spring, when they were observed most frequently. There was also evidence for an ontogenetic shift in silver carp; adults were associated with high mean velocities and juveniles were associated with low mean velocities. This observation concurs with DeGrandchamp et al. (2008), who found adult silver carp preferred channel boarders over backwater habitats, and with Schultz et al. (2007), who found that juvenile silver carp entered backwater habitats more often than adults.

Dredging did not explain a significant amount of variation in CPUE for any of the CCA models, and although fish communities differed by season, there was no significant interaction between reach type and season, indicating dredging activities had little influence on the fish communities, regardless of season. Although the proportion of low near-bed velocities differed among reach types, mean near-bed velocities were similar among all reach types, explaining the lack of influence dredging operations had on CPUEs. Even at the local spatial scale within the dredge reach, the relative abundances or presence–absence of species did not differ, based on their location relative to the dredge hole. This suggests that the species observed may not be using different habitats within and near the dredge hole preferentially, potentially because the structural differences in these areas were functionally similar to these species.

We found little direct effect of sand dredging on the fish community of a Great Plains sand-bed river. However, our study was limited in scope to a few sand dredging sites within one river, which at the time of this study was subjected to moderate dredging activity compared with dredging intensity before the early 1990s. It is quite possible that higher historical dredging intensities, in combination with installation of major tributary dams, served to dramatically alter the fish community of the Kansas River to the point that species sensitive to dredging-related habitat alterations are no longer present. However, testing this hypothesis is challenging because of the difficulty in finding comparable paired systems affected and unaffected by dredging and damming.

Given these constraints, expanding this study to more dredging and control sites at larger spatial scales within one system may allow for inferences across larger spatial scales. We intentionally selected control reaches within 10 km of dredging sites to minimize the effects of changes in channel form and longitudinal changes in fish fauna (Eitzmann and Paukert 2010) that would have confounded the results. Additionally, hierarchical comparisons between the reach and local scale collections may be limited because the local collections did not include a seasonal aspect and sample locations were not identical. However, seasonal differences in abundances of benthic species at the local scale may be minimal because season had no significant influence on CPUEs of benthic species at the 1-km reach scale.

Our observations were similar to Rempel and Church (2009), who found no effect of gravel-bar mining on salmonids and macroinvertebrates and only short-term effects on habitat, which they attributed in part to sustainable harvest rates of sediment. Likewise, Harvey (1986) concluded that the influence of dredging on benthic organisms was highly localized and that fish and invertebrates were influenced more by natural abiotic variations than by dredging activities. Conversely, Paukert et al. (2008) observed more centrarchids within dredge holes in the Kansas River than in areas outside of dredge holes and concluded that the lentic-like habitat of dredge holes may be beneficial to centrarchid species; however, centrarchids were a small proportion of the fish community, and revisiting these dredged areas in this study did not reveal an influence of centrarchids on fish communities at the 1-km reach scale. Additionally, Cross et al. (1982) concluded that dredged sites differed from control sites on the Kansas River, dredged sites having a higher abundance of species adapted to large pools and silt substrates than control sites. Further, a number of the species driving the differences seen by Cross et al. (1982) were beginning to invade the system or were in decline at the time of the study and have since become either abundant throughout the river (bullhead minnow) or have declined and are now potentially extirpated (silver chub and plains minnow; Cross and Moss 1987; Haslouer et al. 2005; Gido et al. 2010). The discrepancy between the results of our study and those of Cross et al. (1982) may pertain to the current predominance of those species that have since increased in abundance and from our inability to detect the species that have declined sharply.

High mobility of a number of species in the Kansas River may further explain the lack of fish community differences we noted among reach types. For instance, blue suckers, shovelnose

sturgeon, plains minnow, and shoal chub may move 100 km within a year (Dudley and Platania 2007; Neely et al. 2009; Perkin and Gido 2011; Wildhaber et al. 2011). Thus, even if dredged areas may be less suitable habitats for these species, the suitability of small-scale habitat patches may be negligible to highly mobile species, where the river serves as a corridor between suitable resource patches (Junk et al. 1989). However, Great Plains fishes tolerate a wide range of abiotic conditions, which fluctuate widely throughout the seasons (Bramblett et al. 2005; Eitzmann and Paukert 2010a) and may be similar to the abiotic conditions resulting from dredging operations (Paukert et al. 2008). Dredging has been linked with bed degradation and channel narrowing (Kondolf 1997; Rinaldi et al. 2005) and may affect fish communities in ways similar to channelization (Brown et al. 1998). The fish fauna of the Kansas River basin are considered to be homogenized, with life histories adapted to a wide range of habitat conditions (Eberle 2007; Eitzmann and Paukert 2010a). The homogenized fish community currently present within the Kansas River may also explain the lack response by the fish community to dredging activities, despite increases in the availability of deep water and low near-bed velocity habitat associated with actively dredged reaches.

The spatial scale of the study may not have been sufficiently large to detect differences in species abundances and the composition of the fish communities. Dredging removes a large amount of sediment from rivers, similar to dams (Kondolf 1997; Kondolf et al. 2002), and this has resulted in the reduction of sandbars and islands, which in turn may lead to decreased fish species diversity (Wyżga et al. 2009; Eitzmann and Paukert 2010b). While our study found that in-stream sand dredging in a Great Plains sand bed river did alter the main-stem river habitat, the modern fish community, which is tolerant to extreme conditions, did not differ between dredged and control reaches.

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REFERENCES

Anderson, M. J., and T. J. Willis. 2003. Canonical analysis of principal coordinates: a useful method of constrained ordination for ecology. Ecology 84:511–525.

Bonner, T. H. 2011. Texas freshwater fishes. Texas State University, Biology Department, San Marcos. Available: www.bio.txstate.edu/~tbonner/txfishes/index.htm. (April 2012).

Bramblett, R. G., T. R. Johnson, A. V. Zale, and D. G. Heggem. 2005. Development and evaluation of a fish assemblage index of biotic integrity for northwestern Great Plains streams. Transactions of the American Fisheries Society 134:624–640.

Brown, A. V., M. M. Lyttle, and K. B. Brown. 1998. Impacts of gravel mining on gravel bed streams. Transactions of the American Fisheries Society 127:979–994.

Cross, F. B., F. J. DeNoyelles, S. C. Leon, S. W. Campbell, S. L. Dewey, B. D. Heacock, and D. Weirick. 1982. Report on the impacts of commercial dredging on the fishery of the lower Kansas River. U.S. Army Corps of Engineers, Kansas City District, Report DACW 41-79-C-007, Kansas City, Missouri.

Cross, F. B., and R. E. Moss. 1987. Historic changes in fish communities and aquatic habitats in plains streams of Kansas. Pages 155–165 in W. J. Matthews and D. C. Heins, editors. Community and evolutionary ecology of North American stream fishes. University of Oklahoma Press, Norman.

DeGrandchamp, K. L., J. E. Garvey, and R. E. Colombo. 2008. Movement and habitat selection by invasive Asian carps in a large river. Transactions of the American Fisheries Society 137:45–56.

Doyle, M. W., and J. M. Harbor. 2003. A scaling approximation of equilibrium timescales for sand-bed and gravel-bed rivers responding to base-level lowering. Geomorphology 54:217–223.

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

Eberle, M. E. 2007. Homogenization of fish faunas and concurrent anthropogenic impacts on plains streams in western Kansas: 1854–2003. Fort Hays Studies, Fourth Series 4, Fort Hays State University, Hays, Kansas.

Eitzmann, J. L., A. S. Makinster, and C. P. Paukert. 2007. Distribution and growth of blue sucker in a Great Plains river, USA. Fisheries Management and Ecology 14:255–262.

Eitzmann, J. L., and C. P. Paukert. 2010a. Longitudinal differences in habitat complexity and fish assemblage structure of a Great Plains river. American Midland Naturalist 163:14–32.

Eitzmann, J. L., and C. P. Paukert. 2010b. Urbanization in a Great Plains river: effects on fishes and food webs. River Research and Applications 26:948–959.

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.

Gaeuman, D., and R. B. Jacobson. 2006. Acoustic bed velocity and bed load dynamics in a large sand bed river. Journal of Geophysical Research 111:F02005.

Galat, D. L., C. R. Berry, W. M. Gardner, J. C. Hendrickson, G. E. Mestl, G. J. Power, C. Stone, and M. R. Winston. 2005. Spatiotemporal patterns and changes in Missouri River fishes. Pages 249–291 in J. N. Rinne, R. M. Hughes, and B. Calamusso, editors. Historical changes in large river fish assemblages of the Americas. American Fisheries Society, Symposium 45, Bethesda, Maryland.

Gerken, J. E., and C. P. Paukert. 2009. Threats to paddlefish habitat: implications for conservation. Pages 173–183 in C. P. Paukert and G. D. Scholten, editors. Paddlefish management, propagation, and conservation in the 21st century: building from 20 years of research and management. American Fisheries Society, Symposium 66, Bethesda, Maryland.

Gido, K. B., W. K. Dodds, and M. E. Eberle. 2010. Retrospective analysis of fish community change during a half-century of landuse and streamflow changes. Journal of the North American Benthological Society 29:970–987.

Guy, C. S., P. J. Braaten, D. P. Herzog, J. Pitlo, and R. S. Rogers. 2009. Warmwater fish in rivers. Pages 59–84 in S. A. Bonar, W. A. Hubert, and D. W. Willis, editors. Standard methods for sampling North American freshwater fishes. American Fisheries Society, Bethesda, Maryland.

Harvey, B. C. 1986. Effects of suction gold dredging on fish and invertebrates in two California streams. North American Journal of Fisheries Management 6:401–409.

Harvey, B. C., and T. E. Lisle. 1998. Effects of suction dredging on streams: a review and an evaluation strategy. Fisheries 23(8):8–17.

Haslouer, S. G., M. E. Eberle, D. R. Edds, K. B. Gido, C. S. Mammoliti, J. R. Triplett, J. T. Collins, D. A. Distler, D. G. Huggins, and W. J. Stark. 2005. Current status of native fish species in Kansas. Transactions of the Kansas Academy of Science 108:32–46.

Hayer, C. A., and E. R. Irwin. 2008. Influence of gravel mining and other factors on detection probabilities of coastal plain fishes in the Mobile River basin, Alabama. Transactions of the American Fisheries Society 137:1606–1620.

Holcík, J. 2003. Changes in the fish fauna and fisheries in the Slovak section of the Danube River: a review. Annales de Limnologie 39:177–195.

Hoover, J. J., K. A. Boysen, J. A. Beard, and H. Smith. 2011. Assessing the risk of entrainment by cutterhead dredges to juvenile lake sturgeon (*Acipenser fulvescens*) and juvenile pallid sturgeon (*Scaphirhynchus albus*). Journal of Applied Ichthyology 27:369–375.

Hoover, J. J., A. Turnage, and K. J. Killgore. 2009. Swimming performance of juvenile paddlefish: quantifying risk of entrainment. Pages 141–155 in C. P. Paukert and G. D. Scholten, editors. Paddlefish management, propagation, and conservation in the 21st century: building from 20 years of research and management. American Fisheries Society, Symposium 66, Bethesda, Maryland.

Houston, J. 1990. Status of the black buffalo, *Ictiobus niger,* in Canada. Canadian Field-Naturalist 104:98–102.

Hoxmeier, R. J. H., D. D. Aday, and D. H. Wahl. 2009. Examining interpopulation variation in bluegill growth rates and size structure: effects of harvest, maturation, and environmental variables. Transactions of the American Fisheries Society 138:423–432.

Hughes, R. M., J. N. Rinne, and B. Calamusso. 2005. Introduction to historical changes in large river fish assemblages of the Americas. Pages 1–12 in J. N. Rinne, R. M. Hughes, and B. Calamusso, editors. Historical changes in large river fish assemblages of the Americas. American Fisheries Society, Symposium 45, Bethesda, Maryland.

Junk, W. J., P. B. Bayley, and R. E. Sparks. 1989. The flood pulse concept in river-floodplain systems. Canadian Special Publication of Fisheries and Aquatic Sciences 106:110–127.

Kanehl, P., and J. Lyons. 1992. Impacts of in-stream sand and gravel mining on stream habitat and fish communities, including a survey on the Big Rib River, Marathon County, Wisconsin. Wisconsin Department of Natural Resources, Research Report 155, Madison.

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

Kondolf, G. M., M. Smeltzer, and L. Kimball. 2002. Freshwater gravel mining and dredging issues. Center for Environmental Design Research, University of California, Berkeley.

Langer, W. H. 2003. A general overview of the technology of in-stream mining of sand and gravel resources, associated potential environmental impacts, and methods to control potential impacts. U.S. Geological Survey, Open-File Report OF-02-153, Reston, Virginia.

Liu, C. C., C. L. Shieh, C. A. Wu, and M. L. Shieh. 2009. Change detection of gravel mining on riverbeds from the multi-temporal and high-spatial-resolution Formosat-2 imagery. River Research and Applications 25:1136–1152.

Martin, C. R., and T. B. Hess. 1986. The impacts of sand and gravel dredging on trout and trout habitat in the Chattahoochee River, Georgia. Georgia Department of Natural Resources, Federal Aid in Sport Fish Restoration, Project F-26–13, Atlanta.

Meador, M. R., and A. O. Layher. 1998. Instream sand and gravel mining: environmental issues and regulatory process in the United States. Fisheries 23(11):6–13.

Miranda, L. E. 2009. Standardizing electrofishing power for boat electrofishing. Pages 223–230 in S. A. Bonar, W. A. Hubert, and D. W. Willis, editors. Standard methods for sampling North American freshwater fishes. American Fisheries Society, Bethesda, Maryland.

Neely, B. C., M. A. Pegg, and G. E. Mestl. 2009. Seasonal use distributions and migrations of blue sucker in the middle Missouri River. Ecology of Freshwater Fish 18:437–444.

Padmalal, D., K. Maya, S. Sreebha, and R. Sreeja. 2008. Environmental effects of river sand mining: a case from the river catchments of Vembanad Lake, southwest coast of India. Environmental Geology 54:879–889.

Paukert, C., J. Schloesser, J. Fischer, J. Eitzmann, K. Pitts, and D. Thornbrugh. 2008. Effect of instream sand dredging on fish communities in the Kansas River USA: current and historical perspectives. Journal of Freshwater Ecology 23:623–633.

Perkin, J. S., and K. B. Gido. 2011. Stream fragmentation thresholds for a reproductive guild of Great Plains fishes. Fisheries 36:371–383.

Pflieger, W. L. 1997. The fishes of Missouri, revised edition. Missouri Department of Conservation, Jefferson City.

Quist, M. C., K. I. Bonvechio, and M. S. Allen. 2009. Statistical analysis and data management. Pages 171–194 in S. A. Bonar, W. A. Hubert, and D. W. Willis, editors. Standard methods for sampling North American freshwater fishes. American Fisheries Society, Bethesda, Maryland.

Quist, M. C., J. S. Tillma, M. N. Burlingame, and C. S. Guy. 1999. Overwinter habitat use of shovelnose sturgeon in the Kansas River. Transactions of the American Fisheries Society 128:522–527.

Rempel, L. L., and M. Church. 2009. Physical and ecological response to disturbance by gravel mining in a large alluvial river. Canadian Journal of Fisheries and Aquatic Sciences 66:52–71.

Richter, B. D., D. P. Braun, M. A. Mendelson, and L. L. Master. 1997. Threats to imperiled freshwater fauna. Conservation Biology 11:1081–1093.

Rinaldi, M., B. Wyżga, and N. Surian. 2005. Sediment mining in alluvial channels: physical effects and management perspectives. River Research and Applications 21:805–828.

Schrank, S. J., and C. S. Guy. 2002. Age, growth, and gonadal characteristics of adult bighead carp, *Hypophthalmichthys nobilis,* in the lower Missouri River. Environmental Biology of Fishes 64:443–450.

Schultz, D. W., J. E. Garvey, and R. C. Brooks. 2007. Backwater immigration by fishes through a water control structure: implications for connectivity and restoration. North American Journal of Fisheries Management 27:172–180.

Simons, Li and Associates. 1984. Analysis of channel degradation and bank erosion in the lower Kansas River. U.S. Army Corps of Engineers, Kansas City District, MRD Sediment Series 35, Kansas City, Missouri.

SonTek/YSI. 2011. RiverSurveyor S5/M9, system manual, firmware version 1.50. SonTek/YSI, San Diego, California.

Ter Braak, C. J. F. 1995. Ordination. Pages 91–173 in R. H. G. Jongman, C. J. F. Ter Braak, and O. F. R. Van Tongeren, editors. Data analysis in community and landscape ecology. Cambridge University Press, Cambridge, UK.

Trautman, M. B. 1981. Fishes of Ohio. Ohio State University Press, Columbus.

U.S. Bureau of Reclamation. 1997. Design of small dams. U.S. Bureau of Reclamation, Denver.

van Rijn, L. C. 1984. Sediment transport, part III: bed forms and alluvial rough-
ness. Journal of Hydraulic Engineering 110:1733–1754.

Wildhaber, M. L., S. H. Holan, G. M. Davis, D. W. Gladish, A. J. DeLonay,
D. M. Papoulias, and D. K. Sommerhauser. 2011. Evaluating spawning
migration patterns and predicting spawning success of shovelnose stur-
geon in the lower Missouri River. Journal of Applied Ichthyology 27:
301–308.

Williamson, C. J., and J. E. Garvey. 2005. Growth, fecundity, and diets of newly
established silver carp in the middle Mississippi River. Transactions of the
American Fisheries Society 134:1423–1430.

Wyżga, B., A. Amirowicz, A. Radecki-Pawlik, and J. Zawiejska. 2009. Hydro-
morphological conditions, potential fish habitats and the fish community in
a mountain river subjected to variable human impacts, the Czarny Dunajec,
Polish Carpathians. River Research and Applications 25:517–536.