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Assessing Linkages Between Small Impoundments and Long-term Trajectories of Prairie Stream Fish Assemblages

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ABSTRACT.—Most stream fish communities have changed over time in response to common anthropogenic disturbances. Impoundments are a widespread anthropogenic stressor that can negatively impact stream fishes as they alter flow regimes, block movements, and act as fountainheads for the introduction and spread of invasive species. Recent studies, however, have reported the occurrence and reproduction of native fishes in impoundments, suggesting they might benefit some native fishes. Our primary objective was to evaluate whether impoundment construction has led to changes in fish community structure in prairie streams. To accomplish this, we compared fish occupancy in small impoundments (<5 ha) to temporal trends in stream occupancy among species to test whether species' increases in stream occupancy were related to their occupancy in impoundments. We examined stream fish communities in the Upper Cottonwood River basin, Kansas, from 1948–2018, and sampled small impoundments in 2016 and 2017. A third (32%) of fish communities in impoundments were similar to stream assemblages, whereas most impoundments (68%) were dominated by sport or bait fishes. In streams, six species showed increases in occupancy and four species showed decreases since small impoundment construction. Of the species that exhibited increased stream occupancy, five showed a positive, logistical relationship between a species' impoundment occupancy and its increase in stream occupancy. Species declining in stream occupancy experienced continued linear declines and may still be declining. Our research suggests stream fish communities have changed since impoundment construction, and are associated with locally-invasive, native species reaching a new stable state in streams accompanied by declines in other native stream fish species.

INTRODUCTION

Anthropogenic disturbances are becoming more common and diverse (Moreno-Mateos *et al.,* 2017), resulting in altered communities (Adjeroud *et al.,* 2009; Perkin *et al.,* 2015; Hedden *et al.,* 2018). When communities are reshaped by anthropogenic disturbances, native species can become locally imperiled (reduced occupancy), locally invasive (increased occupancy), or remain unaffected (constant occupancy), while nonnative species can typically invade and become established (Johnson *et al.,* 2008; Leprieur *et al.,* 2008). But when fish communities are reshaped by disturbances, the trajectory of change for each species can be variable, depending on the system, type of disturbance, and temporal scale examined. Although species' trajectories are not always uniform, characterizing the change and type of trajectory is important in understanding how disturbances restructure communities and inform predictions of changes in community structure in response to anthropogenic alterations (Buckwalter *et al.,* 2018).

Arguably one of greatest stressors to freshwater stream communities is the construction of impoundments (Collen *et al.,* 2014), which are ubiquitous across many landscapes, with most impoundments being small (<40 ha) (Renwick *et al.,* 2005; Willis and Neal, 2012). Impounding streams modifies hydrology and sediment transport, simplifies stream morphology, increases drought effects, reduces and fragments habitats, and alters fish communities (Ligon *et al.,* 1995; Graf, 1999; Falke and Gido, 2006; Perkin *et al.,* 2016). In

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addition to native fish communities responding to these abiotic changes, stream fishes are also threatened by invasive species, which can colonize impoundments and then move into stream habitats (Havel *et al.,* 2005). The impacts of impoundments also reach beyond aquatic systems and have a broad range of consequences for many terrestrial systems, such as altering food webs and introducing contaminants (Speir *et al.,* 2014; Chumchal and Drenner, 2015). Although there are some efforts to remove impoundments (Watson *et al.,* 2018), the vast majority will likely remain in place (Postel *et al.,* 1996), and despite the ecological consequences, there are calls for new impoundments in some regions (Tundisi *et al.,* 2008; Gopal and Ghosh, 2009; Pine *et al.,* 2017). Therefore, continued research on impoundment impacts to stream communities and possible linkages between these habitats and fish communities is needed, especially for smaller impoundments given that most research of impounded waters has been on large reservoirs (Downing, 2010).

While impoundment construction has broad ecological consequences, the impacts on communities can be confounded if other alterations occur, such as land use changes (Collen *et al.,* 2014). However, the Upper Cottonwood River basin, located within the Flint Hills ecoregion of Kansas, has avoided major land-use disturbance following European settlement as shallow soils and rocky steep slopes made cultivation difficult (Kuchler, 1974; Peterson *et al.,* 2004). Given this minimal change in land cover, responses by the stream fish communities is associated with the construction of the 5704 impoundments (1.2 per 100 ha) in the basin (United States Geological Survey (USGS), 2017; Fig. 1). Given that almost all (99.3%) of the impoundments in the basin are small (<5 ha; USGS, 2017), they likely have localized impacts to the nearby stream's fish community and abiotic conditions. Our objectives were to compare fish communities of prairie streams and small impoundments and evaluate whether stream fish community structure has changed over the last 70 y in association with extensive small impoundment construction. First, we sampled small impoundments within the Cottonwood River basin to examine whether there were differences in fish community structure between streams and small impoundments and determine whether impoundments were occupied by stream-dwelling fishes. Next, we examined species-specific changes in stream occupancy and assessed their trajectories of change over time. Then we evaluated whether species-specific increases in stream occupancy were correlated to their occupancy in impoundments. We predicted fish communities in most impoundments would be dominated by fish stocked for angling opportunities and few native stream fish species would be present. We also predicted stream fish communities would shift in response to impoundment construction, with lentic specialists increasing and lotic specialists decreasing in occupancy (Gido *et al.,* 2010; Perkin *et al.,* 2015). The trajectories of change in occupancy for sensitive native species (*e.g.,* lotic specialist) were predicted largely to converge to a new, stable state (logistic curve), whereas nonnative invaders would experience exponential growth in occupancy. Further, we predicted native species showing abrupt increases in stream occupancy would also have high occupancy in impoundments (*i.e.,* escapement from impoundments).

METHODS

STUDY AREA

The Upper Cottonwood River basin, located within the Flint Hills ecoregion of Kansas, drains approximately 490,000 ha which consists largely of tallgrass prairie (Fig.1). Currently, 62% of the basin is grassland, 29% cultivated crops (primarily hay), 4% developed urban land, and 3% forest (USGS, 2016). Livestock grazing is the primary land use. There are 5704

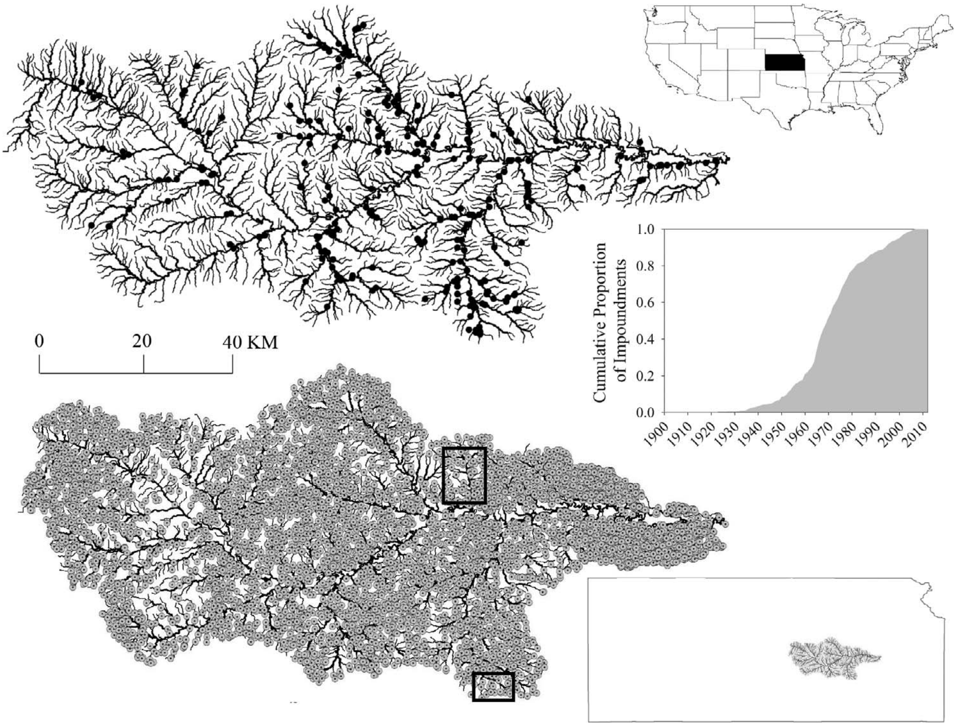

FIG. 1.—Stream survey locations 1948–2018 (top) and the 2018 distribution of impoundments (bottom) in the Upper Cottonwood River basin, Kansas. Black rectangle outlines on impoundment distribution map highlight areas where 31 impoundments were sampled in 2016–2017. Inset graph shows cumulative proportion of impoundments constructed in the area by year.

impoundments within the basin, with most being constructed from 1960–1980 (2012 National Anthropogenic Barrier Dataset of impoundments within 150 km radius of basin) and used as water sources for livestock (Chumchal and Drenner, 2015). Impoundments in this basin were primarily built on low-order streams with earthen embankment dams and outflow pipes and/or spillways to control water levels. Of the 5704 impoundments within the basin, 94% are <1 ha and only 0.7% are >5 ha (USGS, 2017).

DATA COLLECTION

Fish communities were sampled across 31 impoundments (mean size = 0.93 ha, range = 0.06–3.23 ha) in the spring of 2016 and 2017 on the Tallgrass Prairie National Preserve and Flint Hills Tallgrass Prairie Preserve, both owned by The Nature Conservancy. We assumed fish communities in these impoundments were representative of other impoundments in the region, which are mostly on private lands. The stocking history of sampled impoundments was unknown, but if they were stocked it occurred prior to the properties being established as a preserve in 1972 (Flint Hills Tallgrass Prairie Preserve) and 1996 (Tallgrass Prairie National Preserve). Therefore, all sampled impoundments were likely either stocked to create angling opportunities by previous private landowners or naturally colonized by fishes. Impoundment sampling consisted of seining (4.6 m x 1.2 m, 3.2-mm mesh) and backpack electrofishing along approximately 50% of the shoreline as well as gillnets (22.8 m x 1.5 m, 25.4-mm mesh) deployed for 4–12 h in impoundments that were >0.75 ha. In addition to impoundment surveys, we compiled 313 stream fish community samples collected between 1948 and 2018 from the University of Kansas Natural History Museum and Sternberg Museum of Natural History (1948–1997), the Kansas Department of Wildlife, Parks, and Tourism (1994–2010), and Kansas State University (2010–2018). Given the wide range in collection time periods, personnel, and sampling methods, we only considered the presence or absence of species for all analyses.

DATA ANALYSIS

To explore variation in fish communities across impoundments, we used nonmetric multidimensional scaling (NMDS) using a Jaccard distance matrix to reflect occurrences of species rather than abundance. Weighted average scores for species were plotted to visually examine species associated with differing impoundment fish communities. Impoundments with only one species present were removed from this analysis. All analyses were performed in Program R (R Core Team, 2018).

To visually examine whether stream fish community structure shifted over time within the basin, stream samples were plotted with impoundment samples in ordination space (NMDS). Decadal centroids (1948–1959, 1960–1969, 1970–1979, 1980–1989, 1990–1999, 2000–2009, and 2010–2018) were then plotted to examine broad temporal shifts in community structure. The dispersion of samples was examined by plotting ellipses of the standard deviation of point scores for stream samples and impoundment samples. Impoundment samples were categorized into two different groups (a stream fish assemblage or stocked (sport) fish assemblage), based on composition of the fish assemblage (Gabelhouse *et al.,* 2004; Fig. 2A). To test for species-specific changes in stream occupancy (presence/absence data from 1948–2018), we used a model-based approach employing simultaneous Generalized Linear Models (GLM; logit transformed) of multivariate data (*manyglm*) in the *mvabund* package (Wang *et al.,* 2017). This procedure fits a Generalized Linear Model to each species and the log-likelihood ratios for each species are summed and then used as a test statistic via randomization (Warton *et al.,* 2015). Univariate test statistics and accompanying P-values, corrected for multiple testing, were generated using a bootstrap approach with 999 iterations. Given the large number of species (63) and rarity of some species collected over the 70 y time period, only common species (present at >20 of the 313 samples) were examined in the NMDS and *manyglm* analyses. The only predictor of species occupancy in *manyglm* models was the linear effect of year.

After identifying the species showing significant changes in stream occupancy through time from the *manyglm* analysis, we identified the trajectory of change for each species by comparing four models. The four species trajectory models are described as no change (null), linear, exponential, or logistic (Buckwalter *et al.,* 2018). A linear relationship suggests a species is steadily increasing or decreasing and approaching its new, stable occupancy, an exponential relationship suggests a species is rapidly increasing or decreasing with an unknown final occupancy, whereas a logistic relationship suggests a species has changed and reached a new, stable occupancy:

$$\textit{null model}: Y = \mathrm{B0}$$

$$\textit{linear model}: Y = \mathrm{B0} + \mathrm{B1} * \mathrm{X}$$

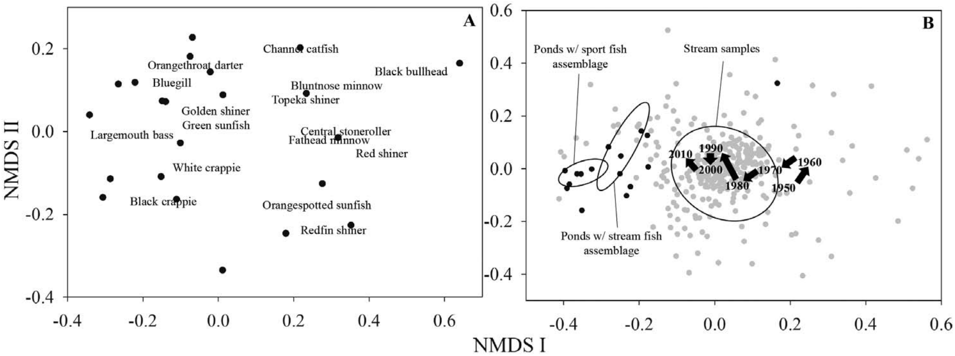

FIG. 2.—Nonmetric multidimensional scaling (NMDS) ordination of 22 small-impoundment surveys (**A**) conducted 2016–2017 in Upper Cottonwood River basin, Kansas. A separate ordination (**B**) included 313 stream fish surveys (grey dots) conducted 1948–2018 in this basin along with the 22 impoundment surveys (black dots). Weighted average scores for each species are plotted for small impoundment surveys (**A**). Impoundments with positive axis I scores had fish communities more associated with species typically found in streams within the basin, whereas negative axis I scores had species frequently found in stocked impoundments. Each decade centroid is shown for stream fish surveys and arrows highlight the shift through time. Ellipses representing the standard deviation of point scores are presented for stream and impoundment surveys, with impoundments surveys categorized based on differing fish assemblages (positive vs negative axis I scores).

$$\textit{two} - \textit{parameter exponetial model}\text{: } Y = \mathrm{B0}*\ e\text{^}(\mathrm{B1}*\ \mathrm{X})$$

$$\textit{three} - \textit{parameter logistic model}\text{: } Y = \mathrm{a}/1 + e\text{^}\Big(- \mathrm{B1}(\mathrm{X} - \mathrm{Xmid})\Big)$$

Where B_0 = y-intercept, B_1 = slope or growth, X = time (year), a = upper asymptote, and X_{mid} = inflection point. Given the variable temporal coverage in stream fish sampling, we partitioned the dataset into time periods for this analysis to remove noise in the data, highlight longer term fluctuations in species presence/absence, and alleviate any temporal or spatial imbalances in sampling (Buckwalter *et al.,* 2018). Time periods were delineated by combining years with minimal fish sampling effort to have >10 samples per period. This resulted in 14 distinct time periods, with time periods encompassing 1 to 11 y (average = 3.4 y) of fish collections (Appendix Table 1). For each time period, we calculated the proportion of sites occupied (occupancy) for each species by taking the number of sites present divided by the total number of sites sampled in that time period. We ranked candidate models based on Akaike's information criterion corrected for small sample size (AIC_c) and considered models with a $\Delta AIC_c < 2$ as supported models (Burnham and Anderson, 2002). Model selection criteria were tabulated using the package *AICcmodavg* (Mazerolle, 2017) in the R statistical environment. To examine whether impoundments contributed to an increase in stream occupancy, we examined native species that increased in stream occurrence and compared those increases to that species' current (2016–2017) impoundment occupancy, using linear regression.

RESULTS

Sixteen fish species were collected from the 31 impoundments. Seven impoundments had no fish species present and two impoundments had only one species (green sunfish *Lepomis*

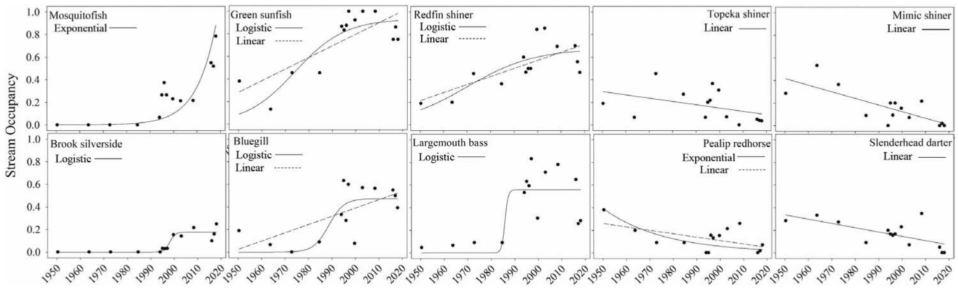

FIG. 3.—Trajectory of change of fish species showing significant increases or decreases in stream occupancy over the 70 y of sampling in the Upper Cottonwood River basin, Kansas. Lines show the top model(s) (linear, exponential, or logistic) for each species. Fish sampling events were delineated into time periods to obtain >10 collections per time period; for collections with more than 1 y sampled, the mean year sampled for that time period is on the x-axis

cyanellus or channel catfish *Ictalurus punctatus*) present. The NMDS from the remaining 22 impoundments showed a separation in impoundment communities along axis I, with higher (positive) scores typically having species that are commonly found in a stream assemblage and lower (negative) scores typically had species that are frequently stocked in impoundments (Fig. 2A). Thirty-two percent (7 of 22) of small impoundments sampled had positive scores on axis I and had communities similar to a native stream assemblage, whereas the remaining 68% (15 of 22) had negative scores and communities typical of a stocked sport fishery (Fig. 2A).

Stream and impoundment fish communities were clearly different, as indicated by the non-overlapping ellipses in multivariate space derived from NMDS (Fig. 2B). Additionally, a directional shift in community structure in stream fish communities was observed through time, with more recent decades of stream fish samples being more similar to impoundment fish communities (Fig. 2B). The *manyglm* analyses examining which species contributed to these directional shifts in community structure through time identified 10 of the 39 (26%) common fish species as having had significant changes in their stream occupancy (Appendix Table 2). Six species increased in stream occupancy over the 70 y study period, and all three trajectories of change (linear, exponential, and logistic) were observed (Fig. 3). The only nonnative fish (western mosquitofish *Gambusia affinis*) exhibiting change in stream occupancy had an exponential increase, whereas the native species exhibiting increases in stream occupancy had either a logistic relationship or both logistic and linear relationships as top models. Of the four species that declined in stream occupancy, the top models for three species exhibited linear declines, whereas the other had both linear and exponential trajectories of change as top models (Table 1; Fig. 3). Five species that increased in stream occupancy (brook silverside *Labidesthes sicculus*, largemouth bass *Micropterus salmoides*, green sunfish, bluegill *Lepomis macrochirus*, and redfin shiner *Lythrurus umbratilis*) all had the logistic model as a top model, indicating they likely had reached a new, stable state. The increases in stream occupancy for these five species were positively related to the species' occurrence in impoundment sampling ($F_{1,4} = 9.25$, $P = 0.056$, $r^2 = 0.76$; Fig. 4).

DISCUSSION

In a basin with substantial impoundment construction over the past 70 y, stream fish community surveys indicated a directional shift through time, with stream communities

TABLE 1.—Model selection output used to evaluate species-specific trajectory of change (increasing or decreasing) in stream occupancy from the Upper Cottonwood River basin, Kansas, 1948–2018. Four models were fitted to changes in stream occupancy and compared using Akaike's information criterion corrected for small sample sizes (AICc). Only species demonstrating significant change from *manyglm* analysis (Appendix Table 1) were tested. Delta_AICc; difference in AICc from top model, AICcWt; weight of evidence for individual models; "-" indicates the fitting algorithm did not converge.

Species	Model	AICc	Delta_AICc	AICcWt	Trend
Western mosquitofish	Exponential	−8.65	0	0.59	Increasing
	Linear	−7.9	0.75	0.41	
	Null	3.85	12.5	0.00	
	Logistic	-	-	-	
Brook silverside	Logistic	−43.43	0	0.99	Increasing
	Linear	−32.12	11.31	0.00	
	Exponential	−31.21	12.22	0.00	
	Null	−24.09	19.34	0.00	
Green sunfish	Logistic	−2.98	0	0.61	Increasing
	Linear	−1.78	1.2	0.33	
	Exponential	1.87	4.85	0.05	
	Null	7.2	10.18	0.00	
Bluegill	Linear	−1.88	0	0.53	Increasing
	Logistic	−0.82	1.06	0.31	
	Exponential	1.62	3.5	0.09	
	Null	2.4	4.28	0.06	
Redfin shiner	Linear	−7.29	0	0.56	Increasing
	Logistic	−5.44	1.85	0.22	
	Exponential	−5.24	2.05	0.20	
	Null	−0.84	6.45	0.02	
Largemouth bass	Logistic	2.65	0	0.81	Increasing
	Linear	6.13	3.48	0.14	
	Null	8.78	6.13	0.04	
	Exponential	11.17	8.52	0.01	
Pealip redhorse	Exponential	−19.99	0	0.50	Decreasing
	Linear	−19.27	0.72	0.35	
	Null	−17.63	2.36	0.15	
	Logistic	-	-	-	
Topeka shiner	Null	−10.89	0	0.56	Decreasing
	Linear	−10.22	0.67	0.40	
	Exponential	−5.78	5.11	0.04	
	Logistic	-	-	-	
Mimic shiner	Linear	−16.97	0	0.80	Decreasing
	Logistic	−14.06	2.91	0.19	
	Null	−8.27	8.7	0.01	
	Exponential	−4.06	12.91	0.00	
Slenderhead darter	Linear	−21.28	0	0.82	Decreasing
	Logistic	−17.2	4.08	0.11	
	Null	−16.56	4.72	0.08	
	Exponential	−6.34	14.94	0.00	

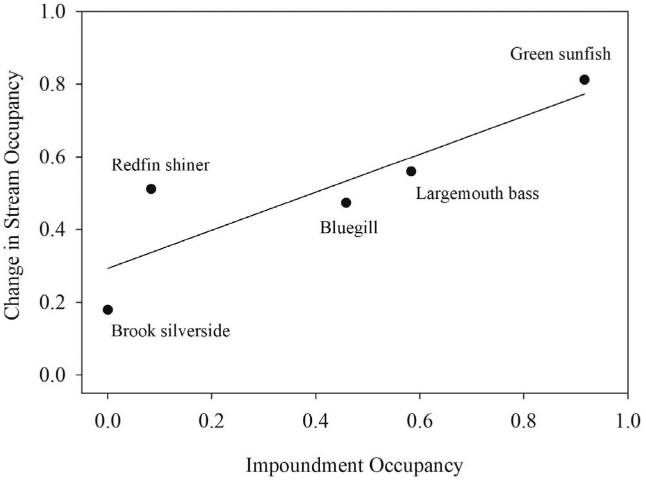

FIG. 4.—Relationship between fish occupancy in small-impoundment sampling conducted 2016–2017 and their increase in stream occupancy. Only native species that experienced an increase in stream occurrence within the Upper Cottonwood River basin, Kansas (1948–2018) were analyzed

becoming more similar to impoundment communities (Fig. 2B). Given are no records that the sampled impoundments have been stocked, it is possible these impoundments were colonized by fish that were upstream from the impoundment when it was constructed or colonized from downstream during high flow events. Therefore, these stream fishes have likely either sustained their populations within these impoundments, or impoundment populations have been supplemented by stream populations. Although it appears that some stream species appear to be able to use impoundments, species-specific responses to impoundment construction through time were variable based on the 70 y of sampling in streams, suggesting differential effects of impoundments across stream fishes.

Several native fish species experienced linear declines in occupancy over time. Declines of these species are potentially attributed to abiotic and biotic stressors associated with small impoundment construction (Taylor *et al.,* 2001; Mbaka and Mwaniki, 2015; Hedden *et al.,* 2018). For example, slenderhead darter (*Percina phoxocephala*) and pealip redhorse (*Moxostoma pisolabrum*) populations are potentially negatively responding to habitat modifications or reduced flow caused by upstream impoundments (Chien, 1985; Slowik *et al.,* 2018), likely because these species have either specific flow requirements for spawning (Brewer *et al.,* 2006) or are sensitive to fragmentation and/or habitat alteration (Reid *et al.,* 2008). Additionally, increased occupancy in predators, such as largemouth bass and green sunfish, may be directly influencing stream fishes through predation or influencing their population through indirect effects on behavior (Bruckerhoff, 2020). For example the mimic shiner (*Notropis volucellus*) and Topeka shiner (*Notropis topeka*) alter their behaviors, movements, and habitat use when predators are present (Hanych *et al.,* 1983; Knight and Gido, 2005); therefore, the indirect effect of forcing these species to suboptimal habitats may have led to poor growth and reproduction (Werner *et al.,* 1983; Harvey, 1991). Lastly, given that the stream occupancies for these species in decline have yet to asymptote and all currently occupy a low proportion (<10%) of sites, the final potential impact of impoundment construction may not be fully observed and could lead to local extirpation of these species. Additionally, it is important to note that rare species were not examined in our analyses; therefore, the observed decline in only 10% of species (4 of 39) could potentially be greater when accounting for rare species, such as hornyhead chub (*Nocomis biguttatus*), which was previously common in parts of the basin but currently are extirpated (Cross and Braasch, 1968; Haslouer *et al.,* 2005).

While impoundment construction may have negative consequences for some native species (Mammoliti, 2002), others appear to be increasing. We observed six species that abruptly increased in stream occupancy, including five natives and one nonnative species (although the basin is on the edge of native distribution for largemouth bass and bluegill). The trajectory of change in stream occupancy by all five of these native species suggests they have reached a new, stable occupancy. This convergence on a new and stable occupancy in streams is likely attributed to these species' ability to thrive in impoundments, as indicted by the positive correlation between species occupancy in impoundments and the magnitude of increase in stream occupancy. These changes illustrate how regional changes in habitat influence local stream fish assemblage structure. It appears that fishes inhabiting small impoundments supplement stream populations through reciprocal exchanges of adults and juveniles (Williams, 2008).

The western mosquitofish was the only nonnative to experience a significant change in stream occupancy. This species exhibited an exponential increase in occupancy, indicating the species is still rapidly colonizing new areas within the basin and has not yet reached its full potential of inhabitation. Impoundments may alter the flow regime and allow for this species to colonize streams at an accelerated rate. For example Murphy *et al.* (2015) found the eastern mosquitofish (*Gambusia holbrooki*) had higher stream presence when there were a greater number of upstream dams that provide a source for colonists and have the potential to dampen flood peaks, giving the species higher colonization rates. Alternatively, western mosquitofish are tolerant and can thrive in prairie streams under extreme environmental conditions (Hopper *et al.,* 2020*a*); therefore, their increase in stream occupancy may be attributed to factors associated with traits of the species as well as small impoundment construction.

Understanding the final state and structure of a community will depend on several factors, but species traits can be used to inform predictions (Mims and Olden, 2012). For instance the ability of fish species to maintain a new and stable state of stream occupancy will depend on life history strategies and the rate of supplementation from impoundments. Equilibrium species, such as largemouth bass and bluegill, have higher survival, growth, and reproduction in more stable habitats (Winemiller and Rose, 1992; Winemiller, 2005); therefore, these species may have difficulty maintaining stream populations without continued supplementation from impoundments. This is especially the case in low-ordered streams for which abiotic conditions are variable and can be severe (Hedden and Gido, *In Press*; Hopper *et al.,* 2020*b*), and where most small impoundments are located. Therefore, species that are more plastic in habitat requirements and can use both impoundment and stream habitats may be favored in these altered systems.

Describing and predicting how fish communities respond to anthropogenic disturbances is a critical step to protecting species from population declines and extirpations. Our research highlights that some impoundments are used by native stream fishes, potentially providing an opportunity for conservation strategies (Chester and Robson, 2013; Hill *et al.,* 2018), but stream communities also have been negatively affected by impoundment construction. Most invasive species have reached a new stable state, likely driven by their occupancy and success in impoundments, whereas species that are experiencing occupancy declines are still declining, potentially related to habitats losses, flow modifications, fragmentation, and other abiotic or biotic modifications following impoundment construction. Therefore, current dynamics of prairie fish stream communities are influenced by the linkage between small impoundments and streams. Future efforts should address survival, growth, and reproduction in newly colonized habitats and the number of fish that escape from and move into impoundments following connection events to better understand the linkage between these habitats.

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APPENDIX TABLE 1.—Fish collections partitioned into 14 time periods to test trajectory of change for species showing significant increases/decreases in stream occupancy in the Upper Cottonwood River basin, Kansas, 1948–2018.

Time period	Years	Mean year	No. Collections (% of total)
1	1948–1952	1950.3	21 (6.7)
2	1958–1967	1963.7	15 (4.8)
3	1969–1978	1972.8	11 (3.5)
4	1979–1990	1984.6	11 (3.5)
5	1994	1994.0	15 (4.8)
6	1995	1995.0	30 (9.6)
7	1996	1996.0	32 (10.2)
8	1997	1997.0	30 (9.6)
9	1998–2001	1999.7	13 (4.2)
10	2002–2004	2003.0	14 (4.5)
11	2007–2010	2008.3	23 (7.3)
12	2016	2016.0	20 (6.4)
13	2017	2017.0	50 (16.0)
14	2018	2018.0	28 (8.9)

APPENDIX TABLE 2.—Analysis of deviance table from *manyglm* output identifying species contributing to fish community change, 1948–2018 in the Upper Cottonwood River basin, Kansas

Species	Deviance	P-value
Western mosquitofish *Gambusia affinis*	48.34	0.001
Brook silverside *Labidesthes sicculus*	20.76	0.001
Mimic shiner *Notropis volucellus*	20.22	0.001
Bluegill *Lepomis macrochirus*	12.59	0.004
Slenderhead darter *Percina phoxocephala*	11.96	0.006
Green sunfish *Lepomis cyanellus*	11.09	0.008
Pealip redhorse *Moxostoma pisolabrum*	10.12	0.011
Redfin shiner *Lythrurus umbratilis*	8.10	0.021
Largemouth bass *Micropterus salmoides*	7.55	0.035
Topeka shiner *Notropis topeka*	7.04	0.050
Stonecat *Noturus flavus*	6.25	0.091
Suckermouth minnow *Phenacobius mirabilis*	5.75	0.122
Black bullhead *Ameiurus melas*	3.92	0.350
Golden redhorse *Moxostoma erythrurum*	3.29	0.480
Carmine shiner *Notropis percobromus*	2.65	0.659
Flathead catfish *Pylodictis olivaris*	2.38	0.718
Spotted bass *Micropterus punctulatus*	2.31	0.723
Bullhead minnow *Pimephales vigilax*	1.81	0.843
Slim minnow *Pimephales tenellus*	1.62	0.858
Yellow bullhead *Ameiurus natalis*	1.53	0.875
Blackstripe topminnow *Fundulus notatus*	1.16	0.946
River carpsucker *Carpiodes carpio*	0.96	0.958
Bluntface shiner *Cyprinella camura*	0.91	0.958
Red shiner *Cyprinella lutrensis*	0.70	0.975
Fantail darter *Etheostoma flabellare*	0.49	0.984
Channel catfish *Ictalurus punctatus*	0.53	0.984
Fathead minnow *Pimephales promelas*	0.63	0.984
Central stoneroller *Campostoma anomalum*	0.39	0.990
Longnose gar *Lepisosteus osseus*	0.29	0.994
Logperch *Percina caprodes*	0.31	0.994
Common carp *Cyprinus carpio*	0.22	0.996
Common shiner *Luxilus cornutus*	0.20	0.996
Bluntnose minnow *Pimephales notatus*	0.12	0.996
Orangethroat darter *Etheostoma spectabile*	0.04	0.997
Orangespotted sunfish *Lepomis humilis*	0.07	0.997
Longear sunfish *Lepomis megalotis*	0.01	0.997
Sand shiner *Notropis stramineus*	0.00	0.997
White crappie *Pomoxis annularis*	0.06	0.997
Creek chub *Semotilus atromaculatus*	0.03	0.997