

**THE UPSTREAM EFFECTS OF RESERVOIRS ON STREAM FISH  
ASSEMBLAGES IN THE GREAT PLAINS**

by

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## ABSTRACT

Reservoirs are important components of modern aquatic ecosystems, and have negative impacts on native aquatic biota both up and downstream. The focus of my thesis research was to investigate the spatial effects of reservoirs on upstream fish assemblages. First, I used a landscape-scale approach to quantify the spatial effects of 19 large reservoirs on upstream fish assemblages at 219 sites in the Great Plains, USA. I hypothesized that fish assemblage structure will vary linearly with distance from a reservoir and that the abundance of reservoir fish species will decline as distance from a reservoir increases. Ordination showed changes in fish assemblage structure occur across river basins and longitudinally within rivers above Kansas reservoirs; additionally, homogenization of fish assemblages appears to have occurred in these basins. The spatial components of reservoir distance and reservoir surface area contributed to a model that predicted the abundance of reservoir species upstream of Kansas reservoirs. Reservoir species abundance significantly declined with distance from a reservoir in 4<sup>th</sup> and 5<sup>th</sup> order streams and reservoir surface area was significantly correlated with the abundance of facultative reservoir species upstream of the respective reservoir. At a smaller spatial scale, I evaluated differences in fish assemblages in twenty 3<sup>rd</sup> through 5<sup>th</sup> order streams that differed in their degree of connectivity to reservoirs. Streams were sampled at the confluence with a river/reservoir and halfway between the stream origin and the confluence. Differences in species richness between connectivity categories were not apparent, but confluence sites had significantly higher total, nonnative, and reservoir species richness than middle sites. Variability in fish assemblage-structure upstream of reservoirs was explained by catchment area, stream size, gradient, and reservoir

connectivity. Discriminant function analysis correctly classified reservoir-influenced streams based on the presence of red shiners (*Cyprinellus lutrensis*) and bluntnose minnows (*Pimepahles notatus*) and the absence of sand shiners (*Notropis stramineus*). Lastly, at the scale of individual streams, I intensively sampled across pool habitats in two streams directly connected to a reservoir. In these pools, the abundance of common reservoir species was related to pool size, turbidity, and canopy cover, but not proximity to the reservoir. Thus, across spatial scales, I observed that the upstream effects of reservoirs on Kansas stream fish assemblages are highly localized; the spatial effect of reservoir proximity is strongest close to reservoirs and quickly attenuates with distance. Because of high abundances of nonnative and reservoir species at stream-reservoir interfaces, it is important to conserve streams not influenced by reservoirs, as these habitats may provide important refugia for native fishes in a reservoir-dominated landscape.

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## INTRODUCTION

Aquatic ecosystems are among the most imperiled systems on the planet (Moyle and Cech 1996). Increasing water demands have modified habitats for many aquatic organisms resulting in changes in aquatic population and community dynamics (e.g., Schlosser 1991). Perhaps the greatest form of habitat modification that threatens aquatic biodiversity has been the construction and operation of dams, which can affect lotic habitats and organisms at a variety of spatial and temporal scales (Cross and Moss 1987, Mammoliti 2002). Impoundments affect stream communities by modifying water quality and hydrologic regimes, serving as barriers to movement and dispersal of aquatic organisms, and facilitating the introduction and survival of non-native species. Whereas there is a wealth of information on the downstream effects of reservoirs, information on the upstream effects are relatively scarce.

Impoundments interfere with upstream and downstream movement of stream fishes. Dams block spawning migrations (Lucas and Baras 2001) and compensatory movements of stream fishes in response to adverse conditions (e.g., flooding or drought, Luttrell et al. 1999). Downstream dispersal of adults and larvae can be limited by reservoirs. For example, several prairie-river fishes produce semi-buoyant eggs which are slightly demersal, non-adhesive, and drift downstream as they develop (Taylor and Miller 1990). These species are more susceptible to extirpation by the presence of impoundments because eggs settle out in reservoirs before hatching (Luttrell et al. 1999).

The colonization of non-native species is also facilitated by the impoundment of streams (Ross 1991, Moyle and Light 1996). Reservoirs serve as a vector for introduced species because intentionally stocked sportfish (Moyle 1976, Minckley 1979) and baitfish

(Wiley and Wydoski 1993) often move into connected river systems. Introduced species also pass downstream of a reservoir through the dam (Sorenson et al. 1998), where they prey on endemics (Holden 1979, Marsh and Douglas 1997, Dudley and Matter 2000) or compete for limiting resources (Brown and Moyle 1991). Thus these efforts to “improve” the reservoir fishery often result in changes to native fish assemblages due to negative interactions with the introduced species (Li et al. 1987, Fuller et al. 1999).

Species introduced into reservoirs can also affect upstream lotic fish assemblages. Introduced facultative riverine fishes (e.g. common carp, gizzard shad) migrate out of the reservoir upstream, where they potentially compete with native species (Erman 1973). The movement of reservoir species into streams can force native species further upstream to suboptimal habitats (Winston et al. 1991). In addition, when conditions upstream become unfavorable, native species are also forced into the reservoir, where they are at increased risk of predation by resident piscivorous sportfish (Winston et al. 1991).

Information on the effects of introduced species upstream of impoundments is relatively scarce when compared to studies on downstream effects. This is surprising considering the importance of headwater systems to the life history stages of stream fishes (Schlosser 1987), many of which are potentially impacted by downstream reservoir construction. The disruption of stream continuity by reservoirs (Ward and Stanford 1983) is particularly important for species in which different life history stages, such as spawning and over-wintering, are carried out across multiple spatial scales (Fausch et al 2002, Schlosser 1991).

Empirical studies report changes in stream fish assemblages in response to impoundments across a variety of spatial scales (Pigg et al. 1998, Gido and Brown 1999, Gido et al. 2002). Winston et al. (1991) found several minnow species, common in the upstream watershed pre-impoundment, to have been extirpated post-impoundment. The authors offered several hypotheses for the extirpation, including: compensatory movement of the minnows into the reservoir where they were preyed upon, invasion of the upstream reaches by reservoir-based piscivores, and the possibility that the populations above the reservoir were metapopulations, now cut off from their downstream spawning source. In the Solomon River in north-central Kansas, Eberle et al. (2002) reported an association among impoundments, the extirpation of native species (32% extirpated), and the introduction of non-native species (51% persist) upstream of impoundments. Most of the studies cited here concerning the upstream effects of impoundments, and others within the literature (Taylor et al. 2001, Crisp et al. 1984, Luttrell et al. 1999), quantify temporal influences of impoundments relative to conditions prior to dam construction.

Gido et al. (2004) showed a non-linear pattern with the number of introduced species and distance from dams in the Great Plains. Although the relationship was weak, the number of introduced species attenuated with distance from reservoirs. This implies a spatial effect of reservoirs, but close proximity alone did not predict high abundances of introduced species. Quantifying the spatial effect of reservoirs on native species has important conservation implications; as these spatial effects would need to be taken into account in any management plan in a reservoir-influenced system.

My thesis research is focused on characterizing the spatial influence of reservoirs on stream fish assemblages in the Great Plains. In particular, what are the spatial effects of reservoirs on stream fish assemblages? Do assemblages vary as a function of distance upstream of a reservoir, with respect to the number of introduced species? Are assemblages in streams directly connected to the reservoir affected more than those upstream some distance, or not directly connected? Which introduced species are more prevalent among assemblages with increasing distance from the reservoir? By examining changes in fish assemblage structure across a variety of spatial scales, I provide evidence to show that the effects of reservoirs, in general, have a localized effect on stream fish assemblages in this region.

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## Chapter 1<sup>1</sup>

Spatial effects of reservoirs on stream fish assemblages in the Great Plains, U.S.A.

### ABSTRACT

Reservoirs are important components of modern aquatic ecosystems, and have been shown to have negative impacts on native aquatic biota both up and downstream. We used a landscape scale GIS approach to quantify the spatial effects of 19 large reservoirs on upstream fish assemblages at 219 sites in the Great Plains, USA. We hypothesized that fish assemblage structure will vary linearly with distance from a reservoir and that the abundance of reservoir fish species will decline as distance from a reservoir increases. Ordination showed changes in fish assemblage structure occur across river basins and longitudinally within rivers above Kansas reservoirs; additionally, homogenization of fish assemblages appears to have occurred in these basins. The spatial components of reservoir distance and reservoir surface area were important variables in predicting the abundance of reservoir species upstream of Kansas reservoirs. Reservoir species abundance significantly declined with distance from a reservoir in 4<sup>th</sup> and 5<sup>th</sup> order streams ( $r^2 = 0.24$ ,  $P < 0.001$  and  $r^2 = 0.33$ ,  $P = 0.002$ , respectively). Reservoir surface area (ha) was significantly correlated with the abundance of facultative reservoir species upstream of the respective reservoir ( $r^2 = 0.427$ ,  $P = 0.002$ ). We found that the upstream effects of reservoirs on Kansas stream fish assemblages are highly localized; the spatial effect of reservoir proximity is strongest closer to reservoirs and attenuates with distance in mid-sized streams. The results of this study are promising when conservation implications are considered. If spatial effects of reservoirs are confined to

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short distances upstream, we would predict intact native assemblages above reservoirs. And, if land surrounding reservoirs is protected, streams in watersheds near reservoirs with relatively minor anthropogenic impacts could be suitable targets for conservation action.

## INTRODUCTION

Dams and their associated impoundments are prevalent features on the modern landscape. Seventy-seven percent of all major rivers in the northern hemisphere are impounded or diverted for irrigation (Dynesius and Nilsson, 1994). In North America, nearly every major river basin contains an impoundment (Benke, 1990), and in Kansas large reservoirs (> 800 ha surface area) occupy all major basins within the state (KDHE, 1996).

Detrimental effects of dams on downstream fish assemblages have been well documented in recent decades. Dams alter physical habitat by causing channel degradation, substrate entrainment, and streambank erosion (Williams and Wolman, 1984; Kondolf, 1997), which leads to habitat simplification and a reduction of native fish species diversity (Berkman and Rabeni, 1987). Changes in thermal regimes below dams have been implicated in the extirpation of stenothermal fish species in these habitats (Vanicek *et al.*, 1970; Holden and Stalnaker, 1975; Edwards, 1978). Dams also alter downstream hydrology, which is an important factor in structuring stream fish assemblages (Poff and Allan, 1995). Unnatural patterns in streamflow variability below dams can reduce diversity of lotic fishes (Cushman, 1985; Bain *et al.*, 1988) and facilitate the establishment of introduced species (Propst and Gido, 2004).

Reservoirs also can affect structure of upstream fish assemblages. In the Great Plains, reservoirs have been implicated in the extirpation (Winston *et al.*, 1991; Lienesch *et al.*, 2000) or decline (Luttrell *et al.*, 1999; Wilde and Ostrand, 1999) of native fishes in stream networks upstream of the impoundments. In addition, species richness of macrohabitat generalists (Herbert and Gelwick, 2003) and centrarchids (Fritz, 1968; Taylor *et al.*, 2001) has been shown to increase upstream of impoundments.

There is evidence to suggest that the spatial proximity of reservoirs influences stream fish assemblages in the Great Plains. Gido *et al.* (2004) showed a constraint envelope effect (Brown, 1995) with the number of introduced species and distance upstream from dams in the Great Plains. Although the relationship was weak, the number of introduced species attenuated with distance from reservoirs; however, other factors appeared to constrain introduced species in close proximity to reservoirs, suggesting that stream fish assemblage structure is influenced by the species pool that occurs in the adjacent reservoir. Fishes introduced into reservoirs enter tributary streams at certain times of year (Martinez *et al.*, 1994). Thus, piscivorous sportfish stocked into Great Plains reservoirs (i.e. *Micropterus* spp., *Sander* spp., and *Morone* spp.) may migrate into nearby streams, where the presence of piscivores could have negative effects on native fish assemblages (Matthews *et al.*, 1994). Reservoirs in the Great Plains also provide habitat for large river species [i.e. *Ictiobus* sp., (Stephens, 1986; Gido and Matthews, 2000)]; thus these species may be present in streams near reservoirs. Quantifying the abundance and distribution of reservoir species in adjoining streams has important conservation implications, in that if reservoirs have only localized effects on

native fish assemblages, many areas in close proximity to reservoirs could become targets for management action.

We used a landscape-scale GIS approach to investigate the spatial effect of proximity to a reservoir on upstream stream fish assemblage structure in Kansas streams. Our objectives were to 1) investigate gradients in fish assemblage structure in streams upstream of 19 Kansas reservoirs, 2) characterize the relative influence of distance to a reservoir and other environmental factors that could structure the upstream fish assemblages of Kansas reservoirs, and 3) quantify the effect of reservoir distance on the abundance of reservoir species at upstream sites. We hypothesized that fish assemblage structure will vary linearly with distance from a reservoir, due to both longitudinal processes and spatial effects of the reservoir. We also hypothesized that the abundance of reservoir fish species will decline as distance from a reservoir increases.

## METHODS

### STUDY AREA

Streams above 19 Kansas reservoirs (>800 ha surface area) were selected for this study (Fig. 1.1). Reservoirs ranged in surface area from 837 ha to 6677 ha (Table 1.1). Sample streams were located within five major river basins in Kansas, corresponding to National Resource and Conservation Service (NRCS) Hydrologic Unit Code (HUC) level 4 (Table 1.1), and were located within the Great Plains ecoregion (Omernik, 1987). Primary land uses in this region were agriculture (72%) and grass/rangelands (25%); urban, forest, and water uses comprised approximately 1% of the area (USGS, 1994).

## MATERIALS AND METHODS

Fish assemblage data from 219 sites was collected by the Kansas Department of Wildlife and Parks (KDWP) during statewide summer surveys between 1995 and 2003. Sampling protocol followed that of the United States Environmental Protection Agency's (EPA) Environmental Assessment and Monitoring Program (Lazorchak *et al.*, 1998). At each site, a reach 40 times the average wetted width of the stream (minimum 150 m and maximum 300 m) was sampled with a combination of straight and bag seines (4.7 mm mesh) and a DC-pulsed backpack electrofishing unit. Upstream and downstream ends of the site were blocked with nets (4.7 mm mesh). One pass was made from downstream to upstream with the electrofishing gear, and one pass was made upstream to downstream seining suitable habitats. Fish were identified to species and each site was georeferenced for entry into a geographic information systems (GIS) point data layer.

Landscape-scale environmental habitat variables were generated from multiple GIS layers using ArcGIS 8.2 software (ESRI, 2002) and tested for their relationships with gradients in fish assemblage structure. Streams within this study area were delineated from a digital map of Kansas streams, based on the 1:100,000 National Hydrography Dataset (USGS, 1997). Stream valley segments (segment of stream from one confluence to the next) were classified by Strahler stream order (Strahler, 1957), link magnitude (Shreve, 1967), and gradient (m/km). Each segment was also assigned a surficial geology class (silt, clay, loam, etc.) taken from the NRCS State Soil Geographic database (STATSGO). Reservoirs selected for this study were represented digitally from a GIS polygon layer of Kansas water bodies (Surface Waters Information Management System) put together by the Kansas Department of Health and Environment and developed from

the EPA's RF3 digital cartographic hydraulic database. Major river basins were defined from a data set of digital hydrologic unit boundaries (HUC-BOUND) (USDA, 1993). Each basin was represented by a unique HUC. Basins were considered at the HUC-4 level in this study.

The closest distance (in km along a respective tributary stream) to a downstream reservoir was measured for each site in ArcGIS 8.2. Using GIS tools, each site was joined to a valley segment, and the environmental attributes for that valley segment were appended to the fish abundance information for those sites. Sites were also classified into which HUC-4 river basin they were located, and the watershed area upstream of each reservoir (ha) was calculated (Table 1.1).

Correspondence analysis (CA) was used to summarize variability in fish assemblage structure across sites. CA is a multivariate ordination technique useful in analyzing a species by sample matrix (Gauch, 1982). Axes gradient lengths provide a measure of faunal turnover, and sample scores four standard deviations apart should have few species in common (Gauch 1982; ter Braak, 1995). Rare species (<0.5% relative abundance across all samples) were excluded from these analyses because they are more likely to occur randomly in samples and not represent true differences in assemblage structure across space or time. Species abundance data were square-root transformed and scaling was focused on inter-species distances using biplot scaling.

Canonical correspondence analysis (CCA) was used to detect relationships between environmental variables (stream size, gradient, surficial geology, and reservoir distance) and spatial variation in the fish assemblage structure. Rare species (<0.5% relative abundance) were again excluded from analysis (see above CA description for

justification). CCA is a modification of CA that selects a linear combination of environmental variables to maximize the dispersion of species scores (ter Braak, 1995). This analysis produces a diagram with vector arrows that represent the relative importance of environmental factors in describing variation in the fish assemblage. Monte Carlo simulations (500 iterations) were used to test whether eigenvalues from the CCA were significantly greater than those generated from a randomized matrix. We used a variance partitioning procedure (Borchard et al., 1992) to estimate the variation explained by reservoir distance as compared to the three other environmental variables. Variance partitioning splits the variation explained by the CCA into separate components: a pure spatial component (reservoir distance), a pure environmental component (stream size, gradient, surficial geology), and the spatial component of environmental influence (overlap between spatial and environmental components). The environmental matrix was subdivided into two matrices, one with reservoir distance serving as a covariate, the other with stream size, gradient, and surficial geology as covariates. CA and CCA analyses were performed using CANOCO ver. 4.5 software (ter Braak and Smilauer, 2002).

Both non-native piscivorous and native species are typically abundant in Great Plains reservoirs. Facultative reservoir species were defined as those that typically occur, or are stocked, in reservoirs and may only require streams for a portion of their life history. These species were classified by a combination of field collections (J. Falke, unpublished data), and a review of species accounts from reservoirs in this region (Eberle *et al.*, 2000; Gido *et al.*, 2002) (Table 1.2).

Linear regression was used to test the association between log-transformed facultative reservoir species abundance and distance from a reservoir using SPSS for Windows ver. 11.0 (SPSS, 2001). Individual regressions were conducted separately for streams with the same Strahler order to control for the effects of stream size. Abundances of facultative reservoir species were pooled for each site. Linear regression also was used to test the dependence between reservoir surface area (ha) and the abundance of facultative reservoir species collected at the upstream sites of each reservoir.

Multiple regression was used to predict the abundance of facultative reservoir species as a function of environmental conditions (stream size and watershed area) and spatial factors (reservoir distance and river basin). Oakes et al. (unpublished data) showed that among twenty-five landscape level variables, watershed area and stream size were among the most important predictors of stream fish assemblage structure in the Big Blue River basin (a subset of the data used in this study). We were also interested in variability in the reservoir species pool among river basins, so a variable indicating which river basin a site was located in was included in order to control for biogeographic effects across this large study area. An information-theoretic approach was used to find the most parsimonious set of independent variables to predict facultative reservoir species abundance (Burnham and Anderson, 1998). First, a set of *a priori* candidate models was selected that contained sets of explanatory variables that were of biological significance. Multiple regression analysis was then performed using facultative reservoir species abundance as the dependent variable and the sets of explanatory environmental variables as independent variables using SPSS. Akaike's information-criterion (AIC) was used to

select the best approximating model by comparing each of the candidate models simultaneously. The AIC scores were adjusted for bias due to a small sample size ( $AIC_c$ ), and Akaike weights ( $w_i$ ) were calculated. Thus, the model with the lowest  $AIC_c$  and the highest  $w_i$  was considered the best model.

Lastly, relative abundance of each fish species at each site was correlated with the distance to a reservoir (km) at each respective site using Pearson's product moment correlation ( $r_p$ ). This analysis was conducted to reveal positive or negative relationships between species abundances and increasing distance from a reservoir.

## RESULTS

The distances of the 219 sample sites above the 19 reservoirs ranged from 0.6 km to 275.4 km ( $\bar{x} = 71.0$  km,  $SE \pm 4.0$ ). A total of 162,358 individuals, representing 56 species, were represented in these collections. *Cyprinella lutrensis* was the most abundant species, representing 33.7% of the total number of individuals, followed by *Notropis stramineus* (11.4%) and *Campostoma anomalum* (8.0%) (Table 1.3). *Cyprinella lutrensis* also was collected at the highest proportion of sites (91.4%) followed by *Lepomis cyanellus* (86.4%), and *Notropis stramineus* (73.6%) (Table 1.3).

Ordination showed major changes in fish assemblage structure occur across river basins and longitudinally within rivers above Kansas reservoirs. The first CA axis accounted for 26.1% of the variation in fish assemblage structure across the 219 sample sites and had a gradient length of 4.9 standard deviations (Fig. 1.2), indicating an almost complete turnover in species composition between samples located at opposite ends of the ordination. Low Axis I scores represented samples collected in the Smoky Hill and

Republican river basins, with intermediate scores (approximately -1 to 1) representing samples collected in the Kansas river basin. Sites located in the Middle Arkansas River basin also had intermediate Axis I scores. High Axis I scores represented sites located in the Neosho-Verdigris River basin. Species with low Axis I scores represented those characteristic of the Kansas River basin, such as *Notropis dorsalis*, whereas species with high Axis I scores were more characteristic of the Arkansas river basin (e.g., *Pimephales tenellus*). Axis II of the ordination explained 15.6% of the overall variation in this assemblage and had a gradient length of 5.2 standard deviations. Sites with high Axis II scores were composed of species associated with reservoirs and large rivers (i.e., *Notropis atherinoides*, *Morone chrysops*, and *Dorosoma cepedianum*), while sites with low Axis II scores contained species more characteristic of headwater areas (i.e., *Phoxinus erythrogaster*, *Luxilus cornutus*, and *Noturus exilis*). Sites mainly composed of ubiquitous species were clustered around the centroid (0,0 on Axis I and II, respectively) and included species such as *Lepomis macrochirus*, *Micropterus salmoides*, and *Phenacobius mirabilis*.

Environmental variables explained 64.9% of the overall variation in the unconstrained assemblage structure (i.e., variation from the indirect gradient analysis) along Axis I and II of the CCA, and confirmed the importance of stream size in structuring the fish assemblage (Fig. 1.3). Eigenvalues from the first two axes of the CCA were significantly different from random (Axis I  $P = 0.002$ , Axis II  $P = 0.002$ ). Stream size (stream order and link magnitude), gradient, and geology explained the most variation in the assemblage structure. Axis I represented a contrast between sites with high gradient and sites in larger streams. High gradient, small streams were characterized

by a headwater fish assemblage, and included species such as *Luxilus cornutus*, *Camptostoma anomalum*, and *Phoxinus erythrogaster*. Sites in low gradient, larger streams included species such as *Morone chrysops*, *Aplodinotus grunniens*, and *Lepisosteus osseus*. Axis II represented a gradient in geology and reservoir distance; sites with lower scores were farther from a reservoir and had silty clay loam surficial geology. Sites with high Axis II scores were near reservoirs, and included species such as *Pimephales vigilax*, *Sander vitreus*, and *Ictiobus cyprinus*. Sites with low Axis II scores represented sites in the Neosho-Verdigris basin, which were located many kilometers (60 – 100 km) upstream of reservoirs in this basin, and included species such as *Minytrema melanops*, *Pimephales tenellus*, and *Notropis volucellus*.

The pure effect of reservoir distance in the CCA explained 12.8% of the overall variation in the unconstrained assemblage structure (i.e., variation from the indirect gradient analysis). The pure environmental component (Strahler order, link magnitude, gradient, and geology) explained 87.2% of the overall variation. There was no overlap between the spatial and environmental components. Each of the covariate models was significantly different from random ( $P = 0.002$ ; Table 1.4), suggesting proximity to a reservoir does influence fish assemblage structure independent of other confounding environmental factors.

The spatial components of reservoir distance and reservoir surface area were important variables in predicting the abundance of facultative reservoir species upstream of Kansas reservoirs. Facultative reservoir species abundance significantly declined with distance from a reservoir in 4<sup>th</sup> and 5<sup>th</sup> order streams ( $r^2 = 0.24$ ,  $P < 0.001$  and  $r^2 = 0.33$ ,  $P = 0.002$ , respectively; Figure 1.4). Reservoir surface area (ha) was significantly correlated

with the abundance of facultative reservoir species upstream of the respective reservoir ( $r^2 = 0.427$ ,  $P = 0.002$ ; Figure 1.5).

The best model selected by the AIC included measures of stream size, reservoir distance, hydrologic unit, and a quadratic function of stream size (included due to the possibility of a unimodal effect of stream size on facultative reservoir species abundance; Table 1.5). The Akaike weight ( $w_i = 0.92$ ) of this model suggests it was approximately 18.4 times more likely to be the best approximating model than the next candidate model ( $w_i = 0.05$ ).

The relative abundance of nine species at sites upstream of reservoirs was significantly ( $P \leq 0.05$ ) correlated with reservoir distance (Table 1.3). Relative abundances of *Pimephales tenellus* and *Minytrema melanops* showed the highest positive correlation with reservoir distance. *Dorosoma cepedianum* and *Morone chrysops* showed the highest negative correlation, indicating high relative abundances of these species near reservoirs.

## DISCUSSION

We found that fish assemblage structure above Kansas reservoirs was structured along a gradient of stream size and watershed area. In addition, zoogeography, and the proximity to reservoirs explained notable variation in assemblage structure. Longitudinal (Huet, 1959; Edds, 1993; Schlosser, 1990; Zorn *et al.*, 2002) and zoogeographic (Cross and Moss, 1987; Hugueny and Lévêque, 1994; Matthews and Robison, 1998) patterns in fish assemblages are well studied in lotic systems. However, we were interested in quantifying the specific effects of reservoirs.

By splitting the variation explained by the environmental variables into spatial and environmental components, we were able to examine the influence of reservoir distance independent of other confounding environmental factors due to longitudinal zonation of the fish assemblages. The spatial component of reservoir distance explained 13% of the variability in fish assemblage structure and also was useful in predicting the abundance of facultative reservoir species at sites upstream of reservoirs. This suggests that spatial proximity to a reservoir does influence structure of stream fish assemblages; however, this effect is weak and appears to be greatest in reaches in close proximity to a reservoir.

Facultative reservoir species abundance attenuated with distance from Kansas reservoirs in mid-sized streams (4<sup>th</sup> and 5<sup>th</sup> order). Habitat in these streams may be more suitable for facultative reservoir species as most of these species are native to larger streams in their native range (e.g. *Ictiobus cyprinus*, *Carpionodes carpio*). Smaller streams (1<sup>st</sup> through 3<sup>rd</sup> order) did not show significant relationships with facultative reservoir species abundance at upstream sites. Large-bodied facultative reservoir species may not colonize smaller streams due to unsuitable habitat, and the stochastic nature of environmental conditions of small streams in the Great Plains (Schlosser, 1987; Dodds *et al.*, 2004). More information on the effects of proximity to a reservoir at a finer spatial scale could provide insight to further quantify the influence of reservoirs on smaller tributary streams.

The positive relationship between reservoir surface area and reservoir species abundance at upstream sites suggests that larger reservoirs may facilitate emigration of reservoir fishes upstream. Because large reservoirs also have larger tributary rivers, the

habitat suitability of these river corridors may be more favorable to facultative reservoir species than tributaries of smaller impoundments because of the more reservoir-like conditions (e.g. low flow) conditions in tributaries of large reservoirs. The source pool of individuals and species would also be expected to be greater in larger reservoirs, possibly increasing colonization opportunities upstream. However, the number of reservoir species at upstream sites was not related to the size of a reservoir, indicating that a few species may be driving the relationship between reservoir species abundance and reservoir size. Finally, larger reservoirs would also be expected to have larger upstream drainage area, thus providing more area for colonization.

Facultative reservoir species abundance at upstream sites also was partially explained by river basin, indicating the variable effects of reservoir distance among reservoirs within a river basin. Distribution of native facultative reservoir species in the Great Plains would explain some of this disparity in effects among basins; for example, *Pimephales vigilax* is native to the Arkansas River basin; however, it is an invasive species in the Kansas River basin (Cross and Collins, 1995). If reservoirs are facilitating populations of this species, we might expect these effects to be stronger within its native range, where overall habitat conditions may be more suitable. Negative effects of reservoirs on upstream assemblages also may be more noticeable in basins which are initially more species rich. The potential for competitive interactions between the native fauna and reservoir influenced species may be greater in these systems. The species by reservoir distance correlation analysis indicated that several species native to the Neosho-Verdigris basin (*Pimephales tenellus*, *Minytrema melanops*, and *Notropis volucellus*)

showed significant positive association with increasing reservoir distance. The Neosho-Verdigris basin is the most species rich basin in this study area (Cross and Collins, 1995).

Introduced species and extirpations of native species lead to the homogenization of fish assemblages across North America and the Great Plains (Gido and Brown, 1999; Rahel, 2000; Gido *et al.*, 2004). Our large-scale analysis of stream fish assemblage structure across a large prairie ecosystem corroborates findings of these studies. We found that stream fish assemblage structure at sites upstream of reservoirs showed a high degree of overlap, regardless of major river basin. Some of this homogeneity stems from the zoogeographic distribution of fishes in Kansas (Cross and Moss 1987; Cross and Collins 1995), as many common species are native to both basins (i.e. *Cyprinella lutrensis*, *Lepomis cyanellus*, *Campostoma anomalum*). Much of this overlap also may result from inter-basin transfers of fishes; i.e. from the Arkansas basin into the Kansas basin (i.e. *Notropis atherinoides*, *Pimephales vigilax*), or from inter-range expansion of fishes due to bait bucket introductions, etc. (i.e. *Notropis dorsalis*). These factors may have exacerbated assemblage homogenization in Kansas.

The impoundment of rivers in Kansas over the past 40 years has resulted in changes to fish assemblage structure upstream of impoundments, likely due in part to introduced species. Other studies in the Midwest found substantial changes in stream fish assemblages in reservoir systems post-impoundment (Pigg *et al.*, 1998; Taylor *et al.*, 2001; but see Gido *et al.*, 2002 for alternative). Reservoirs may play a large part in this process by providing favorable habitat for introduced species (Fernando and Holčík, 1991; Holčík, 1998), and by serving as a vector for both intentional and unintentional introductions (Moyle, 1986). Thus these systems may already be saturated by introduced

species. This could explain the lack of linear effects between distance and reservoir fish abundance in smaller streams; reservoir species were found in smaller stream reaches throughout the system from 0 to 250 km upstream of reservoirs. Comparisons at a finer spatial scale between impounded systems and relatively un-impacted systems could elucidate patterns in introduced species colonization in reservoir systems.

The results of this study also are promising when conservation implications are considered. If spatial effects of reservoirs are confined to short distances upstream, we would predict intact native assemblages above reservoirs. And, if land surrounding reservoirs is protected, streams in watersheds near reservoirs with relatively minor anthropogenic impacts could be suitable targets for conservation action. Caution must be exercised in smaller streams isolated by reservoirs (discharge directly into a reservoir) because little is known about the metapopulation dynamics of streams that are isolated by reservoirs (Winston *et al.*, 1991). For example, if environmental conditions become degraded in these streams, reservoirs may not act as a corridor to suitable refugia for displaced stream fishes. This factor has been implicated in the extirpation of some species which carry out their life history at multiple spatial scales (i.e. those with drifting larvae; Winston *et al.*, 1991; Luttrell *et al.*, 1999).

Upstream effects of reservoirs on Kansas stream fish assemblages are highly localized. The spatial effect of reservoir distance is strongest closer to reservoirs and attenuates with distance in mid-sized streams. These results agree with other studies conducted at finer spatial scales in the Great Plains (Meador and Matthews, 1992; Lienesch *et al.*, 2000). More research into the spatial effects of reservoirs at smaller

spatial scales is needed to clarify patterns of reservoir species abundance in nearby streams in order to quantify preferred habitat by these species outside of a reservoir.

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Table 1.1. List of Kansas reservoirs used in this analysis. Surface area (hectares), the number of sites upstream from each reservoir, the major river basin (HUC level 4), and the watershed area (HUC level 8, hectares) are included for each reservoir.

Reservoir	Surface area (ha)	Year of impoundment	Number of sites	River basin	Watershed area (ha)
Cedar Bluff	2705	1950	7	Smoky Hill	391302
Cheney	3904	1964	13	Middle Arkansas	251214
Clinton	3068	1977	7	Kansas	422635
Council Grove	1060	1964	4	Neosho-Verdigris	292438
El Dorado	3061	1981	2	Middle Arkansas	252016
Elk City	1359	1966	6	Neosho-Verdigris	182050
Fall River	1055	1948	5	Neosho-Verdigris	222983
John Redmond	3025	1964	20	Neosho-Verdigris	292438
Kanapolis	1529	1948	11	Smoky Hill	800044
Keith Sebelius	836	1964	2	Republican	266858
Kirwin	1977	1955	3	Smoky Hill	359655
Marion	2488	1967	4	Neosho-Verdigris	243785
Milford	6676	1967	27	Republican	6449000
Perry	4733	1969	33	Kansas	299658
Toronto	1127	1960	3	Neosho-Verdigris	308686
Tuttle Creek	6257	1959	50	Kansas	2490000
Waconda	4058	1969	12	Smoky Hill	1282545
Webster	1402	1956	4	Smoky Hill	301110
Wilson	3390	1964	6	Smoky Hill	500679

Table 1.2. List of facultative reservoir species, species codes (first three letters of genus and species epithet), each species relative abundance (proportion among facultative reservoir species), and the number of sites occupied.

Species	Species code	Relative abundance (%)	Number of sites occupied
<i>Aplodinotus grunniens</i>	APLGRU	4.20	46
<i>Carpiodes carpio</i>	CARCAR	6.08	79
<i>Cyprinus carpio</i>	CYPCAR	6.14	105
<i>Dorosoma cepedianum</i>	DORCEP	36.53	58
<i>Ictiobus bubalus</i>	ICTBUB	0.34	19
<i>I. cyprinus</i>	ICTCYP	0.23	7
<i>Ictalurus punctatus</i>	ICTPUN	19.26	122
<i>Lepomis macrochirus</i>	LEPMAC	8.61	90
<i>Lepisosteus osseus</i>	LEPOSS	0.48	25
<i>L. platostomus</i>	LEPPLA	0.07	6
<i>Micropterus salmoides</i>	MICSAL	6.87	114
<i>Morone chrysops</i>	MORCHR	1.06	22
<i>Notropis atherinoides</i>	NOTATH	5.78	16
<i>Pimephales vigilax</i>	PIMVIG	0.94	20
<i>Pomoxis annularis</i>	POMANN	1.23	50
<i>P. nigromaculatus</i>	POMNIG	0.11	6
<i>Pylodictis olivaris</i>	PYLOLI	1.93	46
<i>Sander vitreus</i>	SANVIT	0.14	5

Table 1.3. Pearson's product-moment correlation ( $r_p$ ), between individual species abundance and distance to a reservoir (km), significance value (P), abundance (Total), mean relative abundance (Rel. abund.), and percentage of samples that contained each species (% of collections).

Code	Species	$r_p$	p	Total	Rel. abund.	% of collections	Code	Species	$r_p$	p	Total	Rel. abund.	% of collections
PIMTEN	<i>Pimephales tenellus</i>	0.3124	0.0000	706	0.20	6.82	POMANN	<i>Pomoxis annularis</i>	-0.0053	0.9377	170	0.17	22.73
MINMEL	<i>Minytrema melanops</i>	0.2256	0.0007	6	0.00	2.27	MOXMAC	<i>Moxostoma macrolepidotum</i>	-0.0068	0.9196	228	0.13	10.00
MOXERY	<i>Moxostoma erythrurum</i>	0.1835	0.0063	221	0.07	11.36	NOTFLA	<i>Noturus flavus</i>	-0.0150	0.8249	415	0.23	23.64
PIMPRO	<i>Pimephales promelas</i>	0.1671	0.0131	7661	7.21	70.91	LEPHUM	<i>Lepomis humilis</i>	-0.0151	0.8242	856	0.93	37.73
NOTVOL	<i>Notropis volucellus</i>	0.1646	0.0145	139	0.05	3.18	CARCAR	<i>Carpionodes carpio</i>	-0.0226	0.7394	837	0.53	35.91
CYPLUT	<i>Cyprinella lutrensis</i>	0.1499	0.0262	61693	33.73	91.36	CAMANO	<i>Campostoma anomalum</i>	-0.0339	0.6174	15229	7.97	72.27
PHEMIR	<i>Phenacobius mirabilis</i>	0.1284	0.0573	3072	1.65	56.82	CYPCAR	<i>Cyprinus carpio</i>	-0.0488	0.4712	846	0.84	47.73
LEPMEG	<i>Lepomis megalotis</i>	0.1279	0.0582	705	0.30	14.55	PIMNOT	<i>Pimephales notatus</i>	-0.0495	0.4654	7303	3.42	55.45
MICPUN	<i>Micropterus punctulatus</i>	0.1234	0.0678	75	0.03	7.27	NOTRUB	<i>Notropis rubellus</i>	-0.0528	0.4359	460	0.28	3.18
LYTUMB	<i>Lythrurus umbratilis</i>	0.0899	0.1842	1740	1.06	18.64	LEPCYA	<i>Lepomis cyanellus</i>	-0.0586	0.3873	4379	5.33	86.36
PERPHO	<i>Percina phoxocephala</i>	0.0889	0.1892	845	0.37	10.91	ETHNIG	<i>Etheostoma nigrum</i>	-0.0637	0.3471	144	0.07	5.91
AMEMEL	<i>Ameiurus melas</i>	0.0875	0.1963	1136	1.74	49.09	SANVIT	<i>Sander vitreus</i>	-0.0639	0.3457	19	0.01	2.27
LEPOSS	<i>Lepisosteus osseus</i>	0.0751	0.2673	66	0.04	11.36	POMNIG	<i>Pomoxis nigromaculatus</i>	-0.0651	0.3362	15	0.01	2.73
ETHSPE	<i>Etheostoma spectabile</i>	0.0525	0.4383	4434	2.62	47.27	SEMATR	<i>Semotilus atromaculatus</i>	-0.0653	0.3349	7239	5.91	57.73
AMENAT	<i>Ameiurus natalis</i>	0.0410	0.5455	509	0.89	42.27	ICTCYP	<i>Ictiobus cyprinellus</i>	-0.0655	0.3332	31	0.08	3.18
FUNNOT	<i>Fundulus notatus</i>	0.0252	0.7106	180	0.12	8.64	CYPCAM	<i>Cyprinella camura</i>	-0.0656	0.3327	1459	0.67	5.45
APLGRU	<i>Aplodinotus grunniens</i>	0.0224	0.7409	581	0.29	20.91	NOTDOR	<i>Notropis dorsalis</i>	-0.0687	0.3105	805	0.32	5.45
ICTBUB	<i>Ictiobus bubalus</i>	0.0188	0.7820	47	0.03	8.64	NOTATH	<i>Notropis atherinoides</i>	-0.0730	0.2807	796	0.38	7.27
HYBPLA	<i>Hybognathus placitus</i>	0.0047	0.9452	90	0.06	2.27	PHOERY	<i>Phoxinus erythrogaster</i>	-0.0790	0.2433	87	0.02	2.27
LEPPLA	<i>Lepisosteus platostomus</i>	0.0020	0.9764	9	0.01	2.73	LEPMAC	<i>Lepomis macrochirus</i>	-0.0813	0.2295	1186	0.96	40.91
ETHCRA	<i>Etheostoma cragini</i>	0.0015	0.9822	289	0.20	4.55	CATCOM	<i>Catostomus commersonii</i>	-0.0815	0.2285	297	0.29	19.55
							MICSAL	<i>Micropterus salmoides</i>	-0.0865	0.2012	946	1.23	51.82
							NOTEXI	<i>Noturus exilis</i>	-0.0916	0.1758	392	0.18	6.36
							NOTSTR	<i>Notropis stramineus</i>	-0.0921	0.1736	20279	11.39	73.64
							GAMAFF	<i>Gambusia affinis</i>	-0.1012	0.1344	1238	1.07	18.64
							ICTPUN	<i>Ictalurus punctatus</i>	-0.1038	0.1248	2653	1.56	55.45
							FUNZEB	<i>Fundulus zebrinus</i>	-0.1039	0.1244	1879	1.01	10.45
							PIMVIG	<i>Pimephales vigilax</i>	-0.1085	0.1084	129	0.08	9.09
							LUXCOR	<i>Luxilus cornutus</i>	-0.1096	0.1049	605	0.24	6.82
							PYLOLI	<i>Pylodictis olivaris</i>	-0.1129	0.0947	266	0.17	20.91
							LABSIC	<i>Labiesthes sicculus</i>	-0.1192	0.0776	1216	0.58	11.82
							NOTCRY	<i>Notemigonus crysoleucas</i>	-0.1307	0.0529	59	0.06	11.82
							PERCAP	<i>Percina caprodes</i>	-0.1341	0.0470	513	0.28	19.09
							MORCHR	<i>Morone chrysops</i>	-0.1472	0.0290	146	0.07	10.00
							DORCEP	<i>Dorosoma cepedianum</i>	-0.1941	0.0038	5032	2.83	26.36

Table 1.4. Sum of all canonical eigenvalues, F-values and p-values from the CCA species-environment variance partitioning procedure. CCA = full model, Distance = reservoir distance as a covariable, Environment = environmental variables as covariables.

Model	Sum	F	P
CCA	0.501	4.46	0.002
Distance	0.437	11.22	0.002
Environment	0.064	6.52	0.002

Table 1.5. Top linear models for predicting reservoir species abundance above 19 Kansas reservoirs as determined by Akaike information criterion (AIC) rankings. The  $AIC_c$  is the AIC corrected for small sample size,  $K$  is the number of parameters in the fitted model (including the intercept), and the  $\Delta AIC_c$  is the difference between the candidate model and the model with the lowest  $AIC_c$ . The Akaike weights ( $w_i$ ) sum to 1.0.

Model and parameters	$AIC_c$	$K$	$\Delta AIC_c$	$w_i$
Distance, Strahler order, Strahler order <sup>2</sup> , link magnitude, link magnitude <sup>2</sup> , river basin	655.20	11	0	0.92
Distance, Strahler order, Strahler order <sup>2</sup> , river basin	661.45	9	5.83	0.05
Distance, link magnitude, link magnitude <sup>2</sup> , river basin	663.38	9	7.75	0.02
Distance, Strahler order, link magnitude, river basin	665.59	9	9.11	0.01

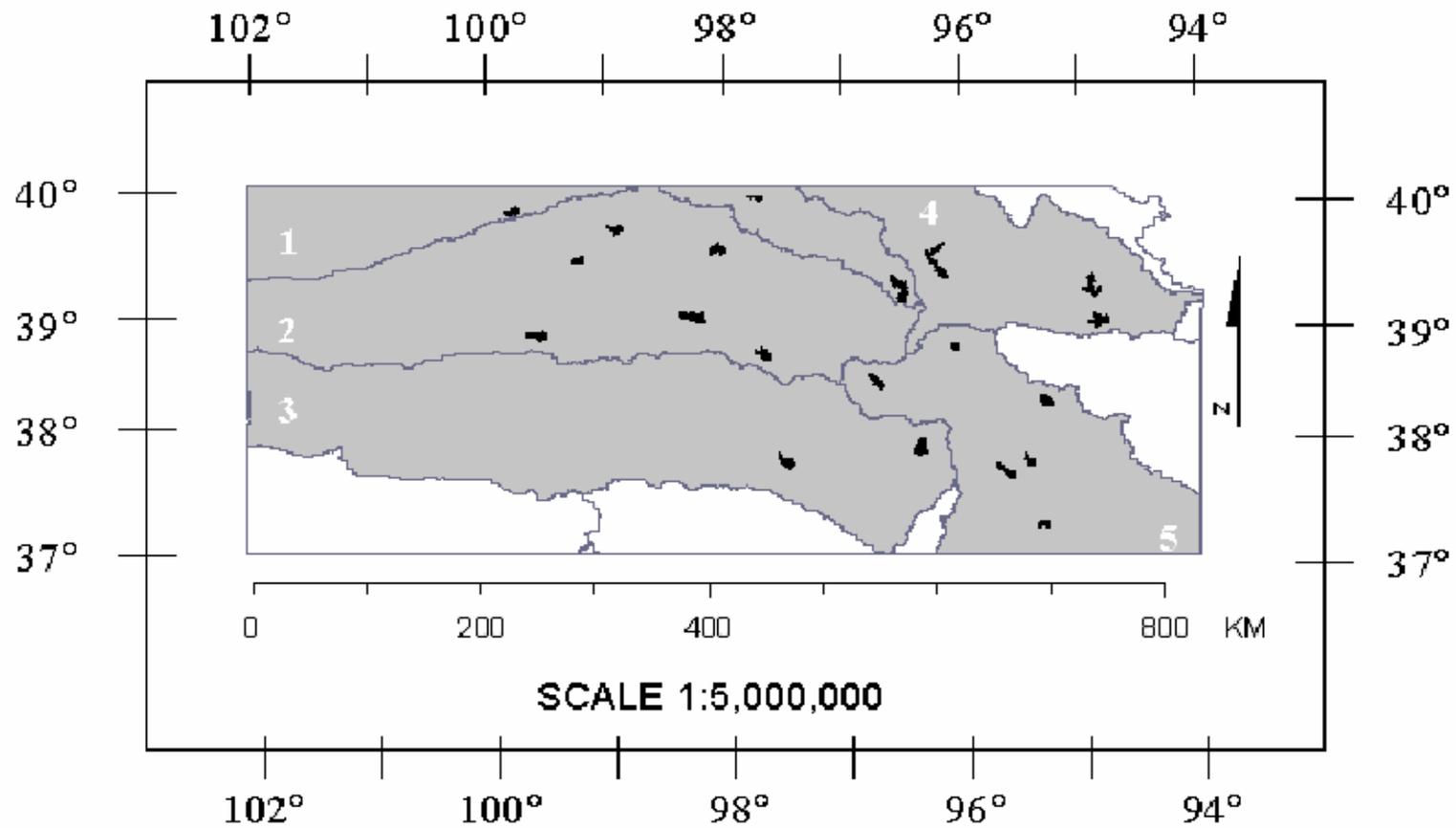


Figure 1.1. Map of the nineteen reservoirs analyzed in this study, and their location in the state of Kansas, USA. Shaded areas are river basins (USGS 4-digit hydrologic units). 1 = Republican, 2 = Smoky Hill, 3 = Kansas, 4 = Middle Arkansas, 5 = Neosho-Verdigris.

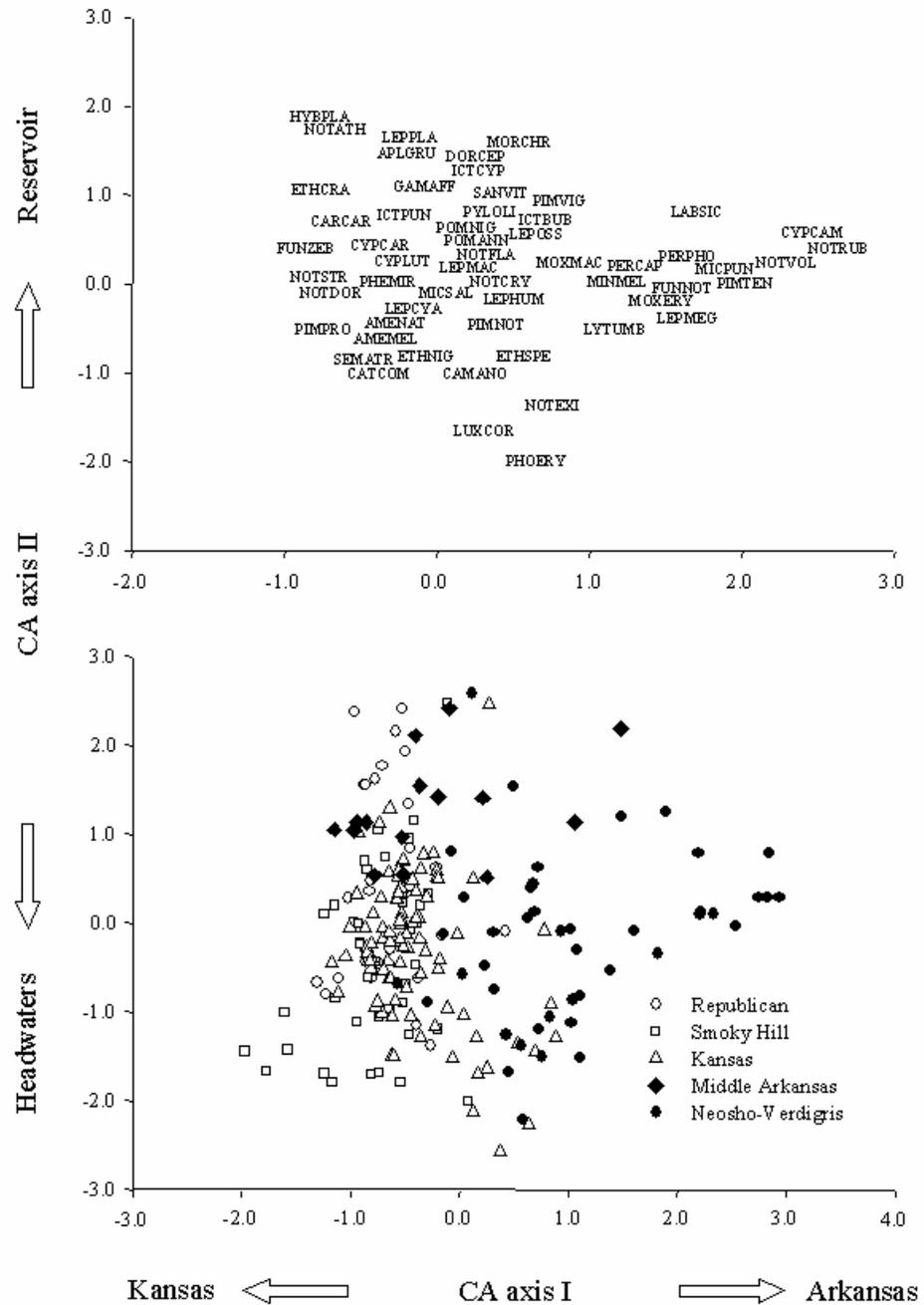


Figure 1.2. Correspondence analysis of fish community data across 219 sites upstream of reservoirs in Kansas. First and second axes had eigenvalues of 0.357 and 0.213 and explained 26.1% and 15.6% of the variation in community structure, respectively. Top panel shows species scores and lower one gives sample scores. Species codes are the first three letters of the genus plus the first three letters of the specific epithet, and are explained in Table 1.3. Symbols on the lower plot correspond to river basins, as explained in legend.

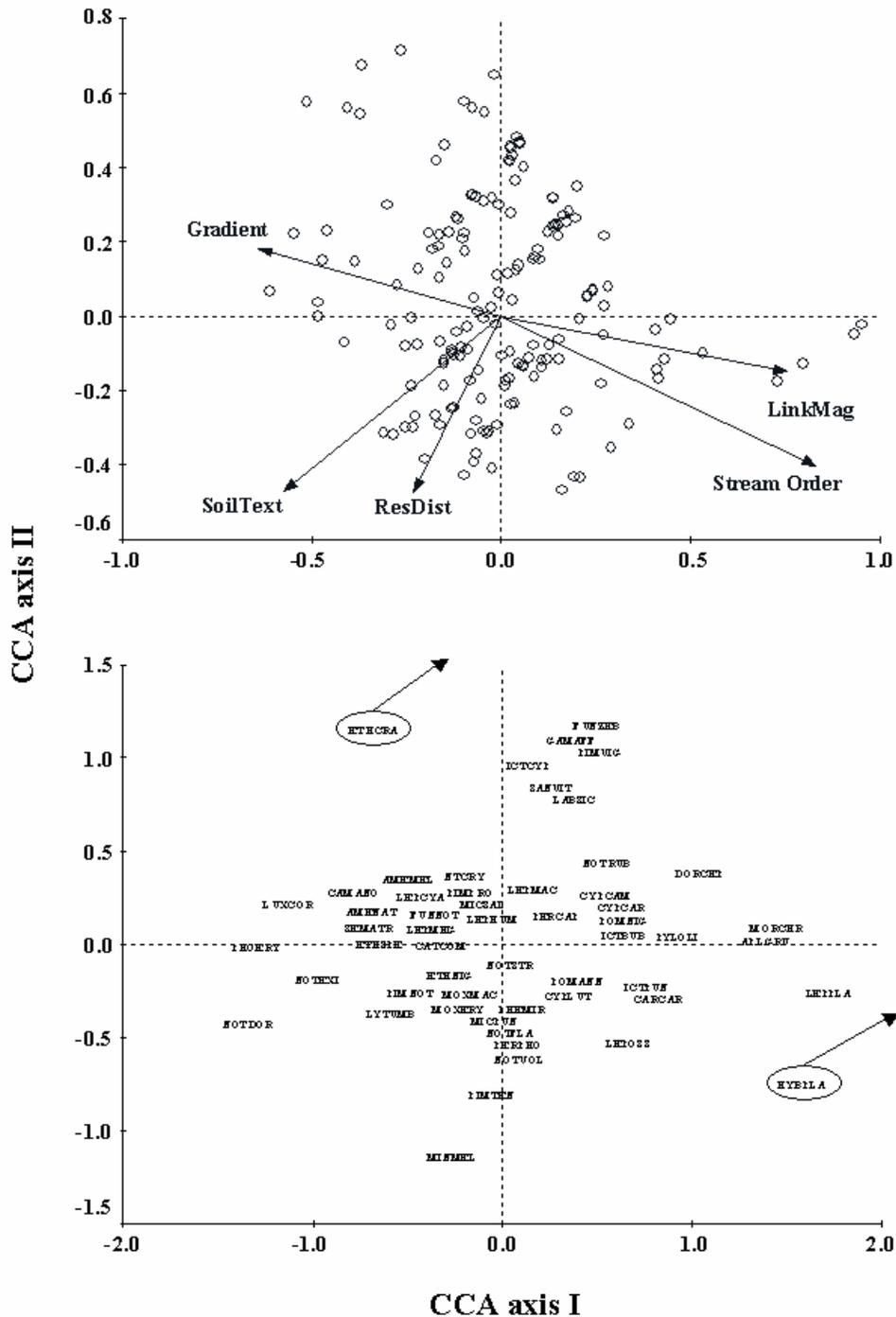


Figure 1.3. Canonical correspondence analysis of fish community data across 219 sites upstream of reservoirs in Kansas. First and second axes of the ordination had eigenvalues of 0.146 and 0.055, respectively. Top graph shows the site scores and environmental correlates, whereas lower panel gives species scores. Species codes are the same as in Figure 1.2. Species enclosed in ovals had scores that extend beyond the scale of the graph. Abbreviation code: Gradient = gradient (m/km), SoilText = soil texture class, ResDist = reservoir distance (km), StreamOrder = Strahler stream order, LinkMag = Shreve link magnitude.

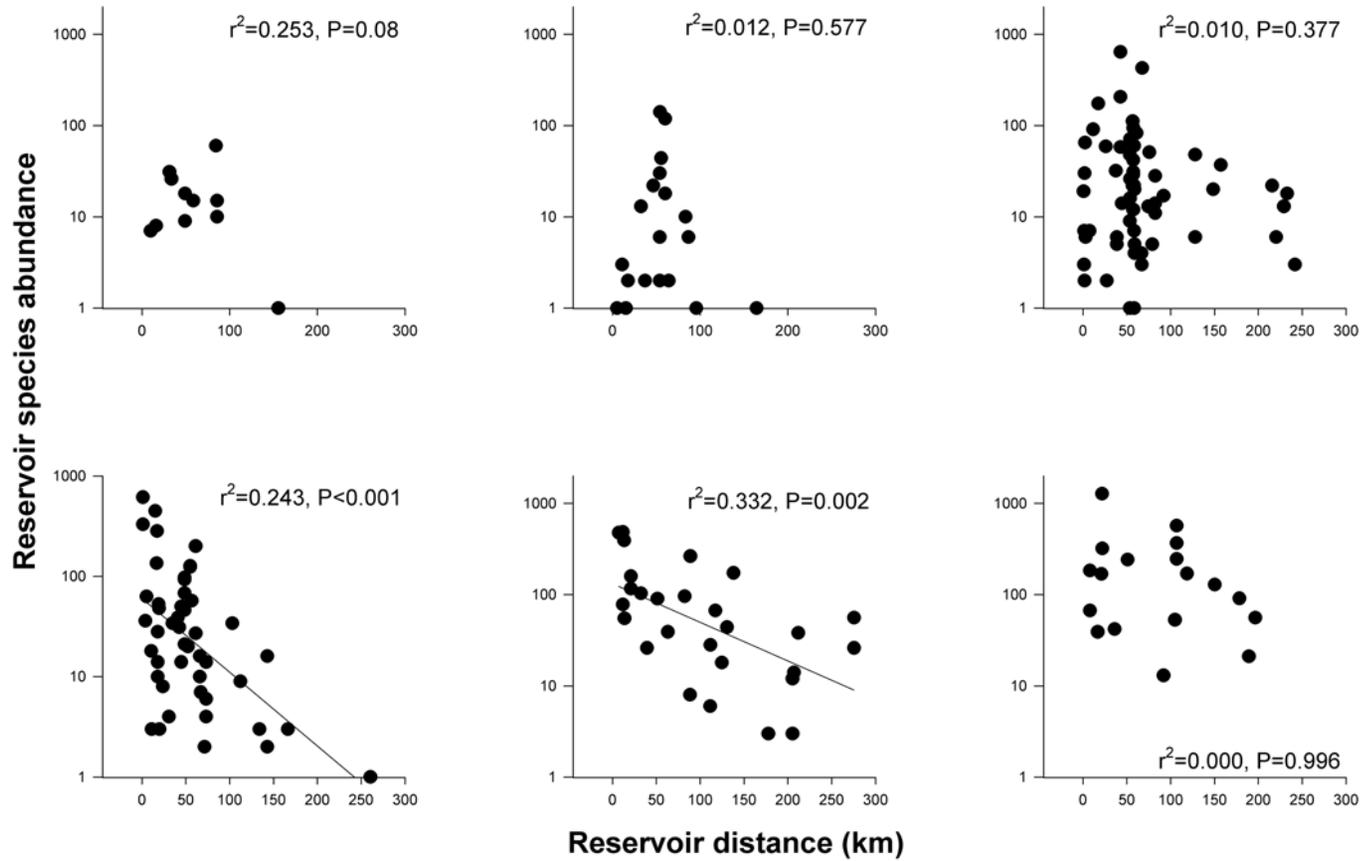


Figure 1.4. Correlation between reservoir species abundance (y-axes) and reservoir distance (km)(x-axes), broken down by Strahler stream order. Plots in the top row represent orders 1 through 3, from left to right, and plots in the bottom row represent orders 4-6, from left to right.

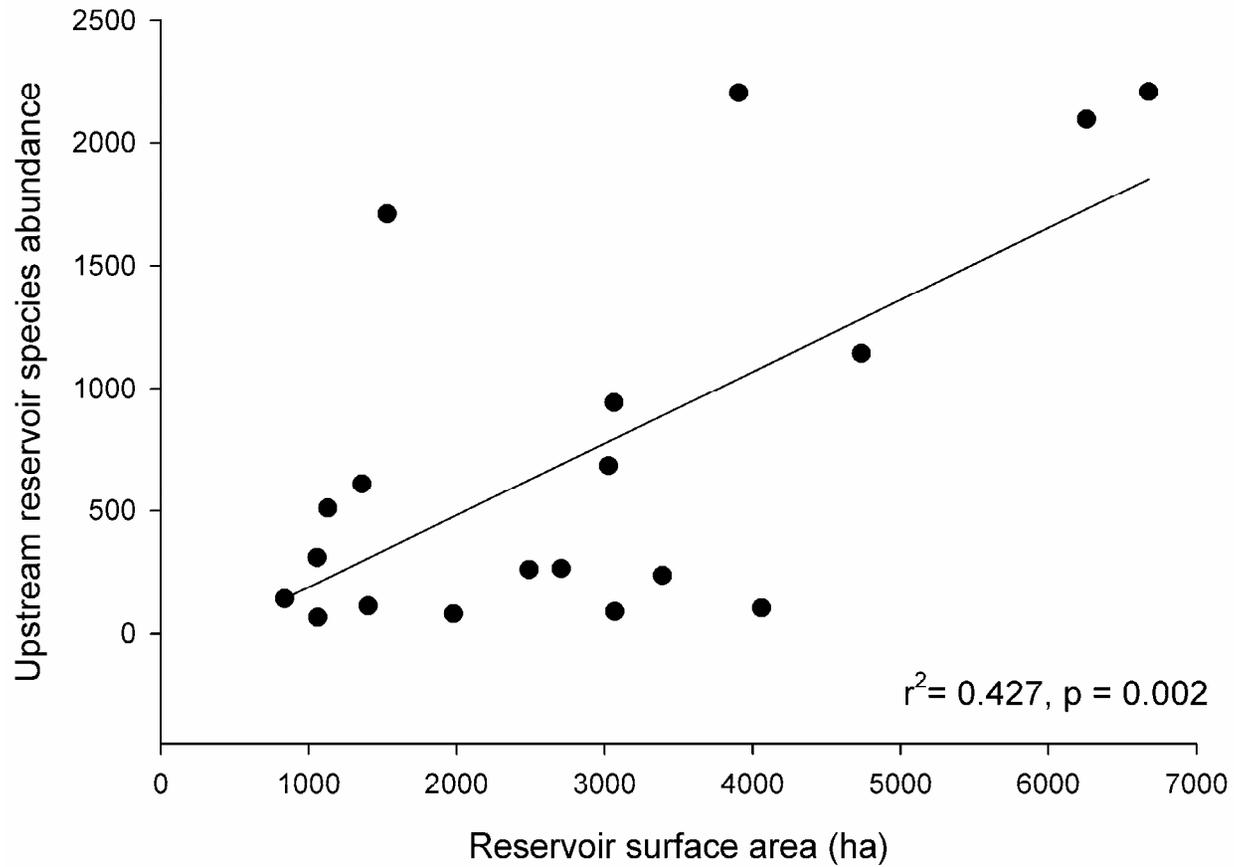


Figure 1.5. Linear correlation between reservoir surface area (ha) of 19 Kansas reservoirs and the abundance of facultative reservoir species (pooled) at KDWP sites in each reservoirs respective watershed.

## Chapter 2<sup>1</sup>

### Effects of reservoir connectivity on stream fish assemblages in the Great Plains

#### ABSTRACT

The upstream effects of reservoirs on fish assemblage-structure are highly localized in the Flint Hills of north-eastern Kansas, U.S.A. We evaluated differences in fish assemblages in 20 3rd through 5th order streams that differed in their degree of connectivity to reservoirs. Streams were sampled at the confluence with a river/reservoir and halfway up the streams wetted width. Differences in species richness between connectivity categories were not apparent, but confluence sites had significantly higher total, nonnative, and reservoir species richness than middle sites. Variability in fish assemblage-structure upstream of reservoirs was explained by catchment area, stream size, gradient, and reservoir connectivity. Discriminant function analysis correctly classified reservoir-influenced streams based on the presence of red shiners and bluntnose minnows and the absence of sand shiners. Intensive sampling across pool habitats within two streams directly connected to a reservoir indicated that abundance of common reservoir species abundance was related to pool size, turbidity, and canopy cover, but not proximity to the reservoir. Conservation of streams not influenced by reservoirs may be critical, as these streams may provide important refugia for native fishes in a reservoir-dominated landscape.

#### INTRODUCTION

Fragmentation of habitats by humans has negatively affected native biota worldwide (Noss and Csuti 1997), including species extinctions and alterations of

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<sup>1</sup> Formatted for submission to Canadian Journal of Fisheries and Aquatic Sciences

community structure (Wilcox and Murphy 1985; Saunders et al. 1991). Stream habitats and biota are heavily reliant on transport processes (e.g., Vannote et al. 1980), thus breaches in connectivity can severely alter ecosystem processes (Ward 1983). In North America, nearly every major river basin contains an impoundment (Benke 1990). However, because most impoundments in North America are relatively young (< 30 years old), there is little information on the long-term consequences of dams on stream communities.

Dams negatively affect native fishes in downstream reaches by altering habitat (Berkman and Rabeni 1987), thermal regimes (Vanicek et al. 1970; Holden and Stalnaker 1975), flow regimes (Cushman 1985; Bain et al. 1988), and by facilitating introduced species (Marchetti and Moyle 2001; Propst and Gido 2004). Whereas upstream effects of dams are less understood (Pringle 1997), recent studies have reported changes in assemblage structure (Penczak 2004), increased richness of macrohabitat generalists (Herbert and Gelwick 2003), decreased juvenile survival (Ponton and Copp 1997), and decreased diversity (Reyes-Gavilan et al. 1996) in streams above reservoirs.

In the Great Plains, where reservoir density is high, the potential for negative impacts by dams on native fish assemblages through disruption of connectivity is widespread. In addition, life history attributes of prairie stream fishes make them particularly vulnerable to the upstream effects of dams (Luttrell et al. 1999; Lienesch et al. 2000). For example, species with drifting larvae are negatively affected when larvae or eggs drift into a reservoir and are either consumed by predators or settle to the bottom (Winston et al. 1991). In addition, reservoirs provide a source of dispersal for bait-bucket or sportfish introductions (Gido et al. 2004). Thus, understanding the spatial effects of

reservoirs on upstream fish assemblages is critical for conservation of native fishes in the Great Plains.

We investigated the effects of stream connectivity to a reservoir on upstream fish assemblage-structure. Our objectives were to 1) investigate if fish assemblage-structure differs among streams with different levels of connectivity to reservoirs, and 2) quantify factors that control fish assemblage-structure within two tributary streams that are directly connected to a reservoir. We predicted that nonnative and common reservoir species richness would be highest in streams that are directly connected to reservoirs. We also predicted that within streams isolated by a reservoir, fish assemblages would be structured along a gradient of spatial proximity to the reservoir and physical habitat. In addition, nonnative and common reservoir species abundance is predicted to decline in isolated streams as distance from a reservoir increases.

## METHODS

### Study Area

Study streams were located within the Flint Hills ecoregion (Omernik 1987) located in northeast Kansas, U.S.A (Fig. 2.1). Geology in the region consists mainly of shale and cherty limestone, resulting in shallow, rocky soils. Due to these geological factors, agriculture (row crop farming and small grain farming) within this region is restricted to floodplain areas. Land-cover within the study catchments was dominated by grasses ( $N = 20$ ;  $\bar{x} = 66.5\%$ ; Table 2.1). Mean proportion of small grain and row crop agriculture was 24.2% for the catchments. The combined land under urban, forest, and other uses was  $< 7\%$  for all catchments (Table 2.1). When compared to catchments

dominated by agriculture, stream water quality in the Flint Hills is relatively pristine (Dodds and Oakes 2004).

Our study streams were centered around two large reservoirs (Fig. 2.1), Milford (6257 ha) and Tuttle Creek (6676 ha). Milford Reservoir impounds the Republican River and was constructed in 1967. Tuttle Creek Reservoir impounds the Big Blue River and was completed in 1959. These reservoirs are both operated by the U.S. Army Corps of Engineers and their primary uses are water storage, flood control, and recreation.

#### Effects of Reservoir Connectivity Among Streams

Study streams were selected based on their level of connectivity with a reservoir (Fig. 2.1) to test the effects of reservoirs on fish assemblage-structure. Directly connected streams (DC) had their confluence within the body of a reservoir (i.e., in a cove). Indirectly connected streams (IC) had their confluence with the main-stem of the impounded river, upstream of a reservoir. Finally, control streams (CT) were not connected to a reservoir and had their confluence with the Kansas River (Fig. 2.1). Forty-one sites on 20 streams were selected for the study, and a GIS coverage was used to quantify physical attributes across stream segments where sampling occurred. Streams selected were wadeable (maximum depth typically < 1.5 m), and had similar stream size, catchment land-use, and catchment surficial geology. Stream order was calculated from a modified version of the National Hydrography Dataset (USGS 1997) and surficial geology was based on soil measurements obtained from the STATSGO database (NRCS 1994). Land-cover was characterized for each catchment using the National Land Cover Database (USGS, 1994) by calculating proportions of each land-use category within a

catchment (Table 2.1). Sites on study streams also were selected based on longitudinal position. Middle sites (MID) were approximately midway (4.4 – 15.8 km,  $\bar{x} = 11.0$  SE = 1.3) up the wetted portion of the stream. Confluence sites (CFL) were at the confluence of the stream with a reservoir or river. Downstream ends of DC-CFL sites were located approximately where stream flow subsided. For IC and CT sites, the downstream end of CFL sites was located at the confluence of the stream and river.

Fish assemblage data for 17 of the 41 sites were collected by the Kansas Department of Wildlife and Parks (KDWP) during summers from 1995-2003. The other sites were visited between July and September 2003. At each site a reach 40 times the average width of the stream (minimum 150 m, maximum 300 m) was sampled. This allowed for equal effort per unit of area (Lazorchak et al., 1998). Sites were blocked with nets at the upstream and downstream ends, and fishes were collected using a DC backpack electrofishing unit and seines (4.7 mm mesh). One upstream pass was made electrofishing, and one downstream pass was made seining suitable habitats. Large fishes (> 200 mm) were identified in the field and released, and small fishes were preserved in 10% formalin, returned to the laboratory, and transferred to 70% isopropyl alcohol for sorting and identification.

Our data analysis focused on changes in fish community structure, with a specific evaluation of changes in the abundance of facultative reservoir and nonnative species. We identified facultative reservoir species in our collections as those that typically occur, or are stocked in reservoirs and may only require streams for a portion of their life history. These species were identified by a combination of field collections (J. Falke, unpublished data) and a review of species accounts from reservoirs in this region (Eberle

et al. 2000; Gido et al. 2002a) (Table 2.2). Nonnatives were classified based on distribution information given in Cross (1967) and Cross and Collins (1995) (Table 2.2).

Two-way analysis of variance (ANOVA; Zar, 1996) was used to test for effects of reservoir connectivity (DC, IC, and CT) and longitudinal position (MID or CFL) on total species richness, native species richness, nonnative species richness, and reservoir species richness. Due to multiple comparisons among richness categories, differences were considered significant at a Bonferroni adjusted alpha level ( $\alpha = 0.0125$ ).

Although we attempted to match streams based on habitat and landuse of stream segments, we used redundancy analysis (RDA) to characterize relationships between catchment-scale environmental variables (see Table 2.2 for variable explanations) and spatial variation in the fish assemblage-structure. RDA is a canonical form of principle components analysis that selects a linear combination of environmental variables to maximize the dispersion of species scores (ter Braak, 1995). This analysis produces a diagram with vector arrows that represent the relative importance of environmental factors in describing variation in the fish assemblage. Monte Carlo simulations (500 iterations) were used to test whether eigenvalues from the RDA were significantly ( $P \leq 0.05$ ) greater than those generated from a randomized matrix. Our analysis was first conducted using the entire dataset. We expected that differences would be apparent between CFL and MID sites due to longitudinal variation in stream size and associated physical and chemical properties within a given stream. Subsequently, we conducted RDA on CFL and MID sites separately to focus on differences in fish assemblages among connectivity groups. To isolate the amount of variation explained by connectivity to the reservoir, we used a partial RDA (ter Braak, 1995) in which physical habitat

variables served as covariates, and the ordination was only constrained by connectivity. A Monte-Carlo procedure (500 iterations) was performed to test if the RDA axes were significantly different from random. We used RDA instead of other multivariate ordination techniques (e.g., canonical correspondence analysis) because of the short gradient lengths of our measured environmental variables. CANOCO software, version 4.5 (ter Braak and Smilauer 2002) was used for the RDA analysis.

Discriminant function analysis (DFA) was used to identify species that could be used to classify streams into reservoir connectivity groups. DFA uses linear combinations of predictor variables to maximize the separation between groups (i.e., reservoir connectivity). DFA is an appropriate method for ecological classification of samples based on a suite of predictor variables (Legendre and Legendre 1998). Independent variables (i.e., species) that were unrelated to connectivity type or that were redundant with other variables were removed from the analysis with a stepwise procedure. For these analyses, variables with partial correlation coefficients with  $F$ -values  $> 3.84$  were entered and those with  $F < 2.71$  were removed. Within-group covariance matrices were used and prior probabilities were computed from group sizes. All models were generated using SPSS, version 11.0. Individual connectivity models were evaluated using a leave-one-out procedure, in which one site was excluded, a model was constructed using  $n-1$  sites, and the excluded site was predicted using this model.

#### Effects of Reservoir Connectivity Within Streams

Because our results indicated a localized effect of reservoirs on stream fish assemblages (see results), we intensively sampled two streams that were directly

connected to Tuttle Creek Reservoir during summer 2004. Both streams, Baldwin Creek and Mill Creek, drain directly into Tuttle Creek Reservoir from the west (Figure 2.1).

We sampled fish assemblages and physical habitat in pools located between the CFL and MID sites sampled in 2003. Twelve pools were sampled on Baldwin Creek and 10 pools on Mill Creek (Fig. 2.1). Pools were blocked off with nets to prevent escape of fishes from the pool. One pass was made through each pool electrofishing suitable habitats (woody debris, boulders, rock piles, etc.) using a DC backpack electrofishing unit. Then, each pool was seined until no additional species were captured (3-10 passes). Large fishes (> 200 mm) were identified in the field and released, and small fishes were preserved in 10% formalin, and returned to the laboratory for sorting and identification.

We also measured physical habitat for each pool. Prior to fish collection, conductivity ( $\mu\text{s/L}$ ) and temperature ( $^{\circ}\text{C}$ ) were measured using a YSI Systems Model 30 meter, and three water samples (500-1000 ml) were filtered (1  $\mu\text{m}$  pore size) on site for total dissolved solids, organic matter, and inorganic matter. Following fish collection, we noted the presence or absence of small and large woody debris, and emergent, submergent, and floating aquatic macrophytes. Canopy cover was quantified using a densiometer at 3 positions in the pool: upstream, middle, and downstream. Pool length (m) was measured along the thalweg of the pool, from the upstream block net to the downstream block net. Based on pool length, five equally spaced transects perpendicular to streamflow were measured. Along each transect, five equally spaced stations were identified and depth (m) and dominant substrate were measured. Substrate was classified according to a modified Wentworth scale (Cummins, 1962) as fines (<2 mm), gravel (2-

15 mm), pebble (16-63 mm), cobble (64-256 mm), boulder (256-1024 mm) and bedrock (>1024 mm).

Prior to evaluating changes in fish assemblage-structure and physical habitat in pools with increasing distance from a reservoir, we quantified differences in physical habitat and assemblage structure between Mill and Baldwin creeks. To test between-stream variability in physical habitat, we used discriminant function analysis with stream as the grouping variable and physical habitat measurements as independent variables. As above, classification success was evaluated using the leave-one-out cross validation technique. We also were interested in testing whether physical habitat of pools varied as distance from a reservoir increased. For each stream, we summarized the physical habitat data into two variables (axes PC1 and PC2) using principle components analysis (PCA) and then used correlation analysis to quantify the association between reservoir distance and PCA axes scores. For this PCA, we focused scaling on intervariable correlations and variable scores were divided by their standard deviation for standardization. To remove effects of unit sizes within the physical habitat variables, we centered and standardized the variables prior to analyses.

We chose two measures of similarity to compare assemblages in each pool to the fish assemblage-structure in the pool closest to the reservoir to test for changes in assemblage structure with increasing distance from a reservoir. Jaccard's index of similarity (Jaccard, 1908) was used to test for similarity in species presence/absence, and percent similarity index (PSI; Renkonen, 1938) to test similarities in species relative abundances. PSI and Jaccard's similarity values were obtained using NTSYSpc software ver. 2.10 (Rohlf, 2000).

To isolate the amount of variation explained by proximity to the reservoir, we used a partial RDA (ter Braak, 1995) in which physical habitat variables served as covariates, and the ordination was only constrained by reservoir distance (see above for partial RDA explanation). Finally, we used multiple regression to investigate the influence of physical habitat features on the abundance of reservoir species. The pooled abundance of reservoir species captured by backpack electrofishing and seining was used as the dependent variable for this analysis. Physical habitat variables and reservoir distance served as independent variables. We used stepwise forward selection ( $P \leq 0.05$ ) to include significant variables in the model. SPSS software ver. 11.0 was used for the multiple regression analysis.

## RESULTS

### Effects of Reservoir Connectivity Among Streams

A total of 37,104 individuals representing 49 species were collected at the 41 sites. These streams were numerically dominated by minnows: red shiners were most abundant (22.51% of total individuals collected), followed by central stoneroller (21.99%) and bluntnose minnow (6.61%) (Table 2.2). Green sunfish were collected at the most sites (38 sites), followed by central stoneroller (37 sites), orangethroat darter (37 sites), red shiner (37 sites), and bluntnose minnow (35 sites).

Two-way ANOVA indicated no significant difference in the four richness categories among stream connectivity types (all  $P > 0.34$ ; Fig. 2.2A-D); however, total nonnative, and reservoir species richness were both significantly higher at CFL sites than at MID sites (all  $P$ -values  $< 0.001$ ; Fig. 2.2A-D). On average, there were 6 more species

(30%) at CFL than MID sites (Fig. 2.2A), and this difference was most pronounced in directly connected streams. Nonnative species richness was approximately 50% higher (2 spp.) at CFL sites than at MID sites. Lastly, reservoir species richness showed a similar pattern, with CFL sites having 78%, or 7 more reservoir species, on average, than MID sites. Although not significant, directly connected streams had the highest mean nonnative and reservoir species richness among the connectivity categories, and richness differences between CFL and MID sites was most pronounced in these streams.

Redundancy analysis characterized the association between fish assemblage-structure and habitat across the 41 sample sites (Fig. 2.3). Cumulatively, Axis I and II explained 68.2 % of the variability in the fish assemblage across sites. Stream size, watershed area, and gradient were important explanatory variables in the RDA. Species typical of small, headwater streams (i.e. creek chub, southern redbelly dace, and white sucker) had low axis I scores, and were found in sites with high gradients and small watershed area. Reservoir species (i.e., gizzard shad, white bass, and saugeye) had high axis I scores and were typical of low gradient, confluence sites. Not surprisingly, separation between MID and CFL site scores was apparent, indicating substantial differences in assemblages between longitudinal positions.

Axis I and II of the RDA used to characterize fish assemblage and habitat associations among the 18 CFL sites explained 67.0% of the variability in the fish assemblage across sites (Fig. 2.4). Stream gradient, stream size, watershed area, and the proportion of agricultural landuse within the catchment were important explanatory variables. Species characteristic of DC-CFL sites included emerald shiner, river carpsucker, and golden shiner (all classified as reservoir species) and had low axis I

species scores. Axis I represented a gradient of stream size, with sites on smaller streams having high axis I site scores, and sites on larger streams having low axis I scores (Fig. 2.4). In general, CT sites had high axis II scores, whereas DC and IC sites had low to moderate axis I scores. Connectivity in CFL sites was an important predictor of fish assemblage-structure, as evidenced by a significant axis I ( $F = 3.764$ ,  $P = 0.001$ ) relationship between assemblage-structure and connectivity when environmental variables were entered as covariables.

Axis I and II of the RDA used to characterize fish assemblage and habitat associations among the 23 MID sites explained 64.2% of the variability in the fish assemblage across sites (Fig. 2.5). Stream size, watershed area, and gradient were important explanatory variables. Species typical of headwater assemblages (i.e. central stoneroller, southern redbelly dace, orangethroat darter, and creek chub) had lower axis I scores, whereas relatively larger stream sites, characterized by species such as red shiner, redbelly shiner, and sand shiner, had higher axis I scores. Site scores did not cluster according to reservoir connectivity in ordination space (Fig. 2.5), indicating weak effects of reservoir connectivity at MID sites. This was confirmed by a non-significant relationship between assemblage structure and connectivity when environmental variables were included as covariables ( $F = 1.159$ ,  $P = 0.168$ ).

For CFL sites, 77.8% of sample sites were correctly classified according to reservoir connectivity using DFA. DC and CT sites were classified 100% correctly, but IC sites were only classified correctly for 1 of 5 sites (20%). IC sites were evenly misclassified as DC or CT sites 40% of the time, respectively. Three species were entered into the analysis from the stepwise procedure: sand shiner, red shiner, and

bluntnose minnow. Discriminant function 1 separated the connectivity categories based on high abundances of red shiner at IC sites, and high abundances of sand shiner at CT sites. Bluntnose minnow was associated with DC sites. Group means in discriminant function 1 through 2 were significantly different from one another (Wilks' Lamda = 0.11,  $P < 0.001$ ).

MID sites were assigned to connectivity categories 87.0% correctly using the DFA cross-validation approach. In this case, IC sites were classified correctly 100% of the time, indicating high predictability at these sites. DC and CT sites were grouped correctly 75.0% and 88.9% of the time, respectively. Species entered into the model for MID sites were redbfin shiner, yellow bullhead, and central stoneroller. Discriminant function 1 separated the reservoir connectivity groups based on high abundances of redbfin shiner at CT sites, yellow bullhead at IC sites, and central stoneroller at DC sites. Group means in discriminant function 1 through 2 were significantly different from one another (Wilks' Lamda = 0.13,  $P < 0.001$ ).

#### Effects of Reservoir Connectivity Within Streams

A total of 8,369 individuals representing 26 species were captured in the 22 pools in Baldwin and Mill Creeks, and species richness ranged from 6-18 across pools. Minnows numerically dominated the collections, as southern redbelly dace (27.1%; relative abundance), central stoneroller (25.2%), and common shiner (18.8%) were the most common species collected.

Discriminant function analysis revealed differences in physical habitat parameters between the two streams. Sample sites from the two streams were correctly classified

based on physical habitat 91% of the time (Wilks Lamda = 0.321,  $P < 0.001$ ). Compared to Mill Creek, Baldwin Creek was a smaller stream, containing smaller, shallower pools, with a higher proportion of canopy cover. Based on these results, we analyzed each stream separately.

PCA I of physical habitat was significantly correlated with distance from a reservoir for both Baldwin and Mill Creeks ( $r^2 = 0.52$ ,  $P = 0.009$ ;  $r^2 = 0.43$ ,  $P = 0.04$ ; respectively). Regardless, fish assemblage-structure did not vary across sites with increasing reservoir distance based on a consensus of several analyses. Mean Jaccard's index of similarity based on presence/absence of fish species between the site nearest the confluence and all other sites was 0.50 (SE = 0.04) for Baldwin Creek and showed no pattern with reservoir distance (Fig 2.6A). Similarly, mean Jaccard's index of similarity for Mill Creek was 0.61 (SE = 0.04), and did not show a pattern with reservoir distance (Fig 2.6B). When we considered patterns in species abundances based on percent similarity, there also was no correlation with reservoir distance. Mean PSI values across sites for Baldwin Creek were  $0.50 \pm 0.04$  SE (Fig 2.6A) and  $0.64 \pm 0.06$  for Mill Creek (Fig 2.6B). Further, when we partitioned variation in the dataset into that explained by physical habitat versus reservoir distance using RDA for each stream, we found that the first axis of the partial RDA using physical habitat variables as covariates was non-significant for both streams ( $P$ -values  $> 0.12$ ). This suggests that variability in fish assemblage-structure in these streams was better explained by physical habitat parameters than reservoir proximity.

Finally, in Baldwin Creek, reservoir species abundances quickly declined to zero as reservoir distance increased, but in Mill Creek, reservoir species were present in pools

throughout the stream, but were always in low abundance (Fig 2.7). Using stepwise procedure to select variables in a multiple regression model using both streams, we found that organic matter, pool volume, percent canopy cover, and maximum depth explained 74% of the variation in reservoir species abundance in pools within our study area ( $P < 0.001$ ).

## DISCUSSION

Our data suggest the influence of reservoirs on stream fish assemblage-structure in connected streams is highly localized. We found that total, native, nonnative, and reservoir species richness were all higher near reservoirs than at sites farther upstream. Although this pattern may partly be explained by within stream longitudinal processes, many of the species that make up the difference in richness between confluence and middle sites migrate to confluence sites from the reservoir. In isolated, directly connected streams, there was a pronounced difference in species richness between CFL and MID sites. Was the paucity of species at upstream sites in these isolated streams due to reservoir effects? At least two species have been extirpated from streams directly connected to our study reservoirs, Topeka shiner and carmine shiner (Minckley and Cross 1959, Cross and Collins 1995). Loss of refugia from stochastic abiotic conditions combined with downstream habitat changes from reservoir construction is cited as the primary cause of decline in these species (Cross 1967; Cross and Collins 1995). This effect could explain the lower richness observed in the directly connected stream MID sites.

The observed differences in assemblage structure between CFL and MID sites was not unexpected, as longitudinal processes can influence assemblage structure in lotic systems (Horwitz 1978; Schlosser 1987). However, assemblages at sites near confluences may be influenced more by an “edge” effect, rather than to longitudinal processes. This was illustrated by the co-occurrence of large river/reservoir and small stream species at these sites. Thus, assemblage structure at confluences is likely influenced by a combination of upstream longitudinal processes and emigration from downstream reservoirs.

Both RDA and DFA were used to evaluate the effects of connectivity on fish assemblage-structure. Ordination of fish assemblages at CFL sites revealed differences in structure among connectivity types. Separation in site scores was apparent among CT streams and the other two connectivity types. In general, piscivorous reservoir species (e.g., largemouth bass, white bass, white crappie) were more closely associated with DC and IC streams, whereas confluence sites in CT streams were associated with native species (e.g., sand shiner, redbfin shiner, black bullhead). One confounding factor was the disproportionate coverage of agriculture in the catchment of IC streams. This could have influenced differences in assemblage structure independent of reservoir connectivity. Classification of sites into connectivity groups was partially successful using DFA. IC-CFL streams were evenly misclassified between DC and IC groups, indicating a possible gradient of connectivity effects between the three categories. The abundance of sand shiner was a strong predictor of connectivity type at CFL sites; DC and IC streams had very low abundances of this typically common species as compared to CT streams. Low abundance of sand shiners in these reservoir-influenced sites may be a cause for concern,

as other minnows with similar life history traits (e.g. western silvery minnow, plains minnow, speckled chub) have drastically declined in incidence and abundance upstream of reservoirs in these systems (Cross and Collins 1995; Gido et al. 2002b).

As indicated by RDA, whole assemblage structure at MID sites did not appear to be influenced by connectivity, although DC and IC sites generally had higher abundances of bluegill, largemouth bass, and white crappie than CT sites. However, the presence of these species could be influenced by numerous small impoundments in DC and IC watersheds. Overall, differences in common fish assemblage-structure in middle sites were not related to connectivity with a reservoir. Rather, assemblage structure at middle sites was primarily driven by catchment area, stream size, and gradient. Classification into connectivity types also was successful for middle sites using DFA. The presence of redbfin shiner accurately predicted if a site occurred in a CT stream. Although currently not a species of concern, redbfin shiner commonly occurs with other species that have been cited as being imperiled in Kansas, including Topeka shiner and common shiner (Gido et al., 2002b). Low incidence of redbfin shiner at upstream sites in DC and IC streams may stem from influences of downstream reservoirs or habitat degradation due to agricultural practices within the watershed.

Within stream patterns in fish assemblage-structure were weakly linked to proximity to the reservoir. Abundance of reservoir species in pools along Mill Creek did not vary with distance from the reservoir, whereas the abundance of these species declined in pools furthest from the reservoir in Baldwin Creek. Mill Creek is a larger stream than Baldwin Creek (4<sup>th</sup> vs. 3<sup>rd</sup> order), with deeper pools and more complex habitat, thus the observed pattern may be related to more available suitable habitat for

reservoir species in Mill Creek. However, patterns in assemblage structure in Baldwin Creek also may have been heavily influenced by recent hydrologic events. Specifically, pools 5-10 in Baldwin Creek were completely dried during 2003 (H. Klaassen, pers. comm.). High abundances of more typical “headwater” species (e.g., southern redbelly dace and central stoneroller) in pools near the confluence, and relatively low species richness/abundance in pools 5-10 may reflect a failure of the headwater species to recolonize upstream.

Reservoirs and adjacent stream confluences could act as barriers simply due to the presence of predators (*sensu*, Fraser et al. 1995), thus negative interactions between native and reservoir species at these stream-reservoir interfaces could explain the observed differences in species richness in the 2003 dataset between confluence and middle sites in isolated streams. Downstream compensatory movement of fishes into pools near the reservoir places them at higher risk of predation due to the presence of piscivorous reservoir species. Quantification of competitive and predator-prey interactions among reservoir and native species at the stream-reservoir interface is needed to determine the consequences of connectivity to these habitats. Overall, we found that patterns in reservoir species abundances in directly connected streams agreed between spatial scales, with abundances quickly curtailing to zero as distance from a reservoir increased.

Multiple regression analysis with reservoir species abundance as the dependent variable allowed us to evaluate factors that favored the abundance of these species. In Baldwin and Mill creeks, reservoir species preferred large, deep pools with relatively high turbidity, and a low proportion of canopy cover. In our study area, these conditions

are typical of pools near reservoirs, where large pools result from longitudinal catchment geomorphological processes, and canopy cover has been reduced by numerous inundations by the reservoir in high-water years. Higher turbidity (as indicated by relatively high amount of organic matter within these pools) may result from a combination of upstream inputs and silt deposition from prior inundation during the spring. Lack of these conditions at upstream sites, as well as the more stochastic nature of environmental conditions upstream, may prevent reservoir species from colonizing upstream pools in this study area.

In conclusion, overall assemblage structure observed among streams in the Flint Hills agreed with patterns observed in Baldwin and Mill Creeks. We found significantly higher abundances of nonnative and reservoir species in close proximity to reservoirs; however, their abundance quickly declined as distance from a reservoir increased, with the exception of Mill Creek. These observations at moderate and small spatial scales agree with previous patterns observed at large spatial scales in streams upstream of Kansas reservoirs (J. Falke and K. Gido unpublished data).

Understanding the influences of stream connectivity to reservoirs has several implications for conservation of native fishes in the Great Plains. First, streams isolated by reservoirs may not be suitable targets for conservation (e.g., land acquisition or restocking) if downstream compensatory movement of fishes, when upstream conditions become unsuitable, places them at higher risk of competition or predation. This is particularly apparent, given our finding that downstream pools had higher abundances of nonnative and reservoir species. Second, although indirectly connected streams would seem to be better choices for conservation, streams in this region also are more heavily

impacted by agriculture than streams of other connectivity types. With this in mind, careful selection of catchments using landscape-scale analysis could target indirectly connected streams with relatively low proportions of agriculture in their catchments.

Choosing streams for conservation efforts is critically needed in the Great Plains, as there are a large number of imperiled fishes (Haslouer et al., in review), and the majority of streams within Kansas are impacted by human activities. Although our data suggest a localized effect of reservoirs on stream fish assemblages, it is important to note that our control streams were not free of human alteration (e.g., changes in water chemistry). Thus we recommend careful consideration of the life history of native species as well as the reservoir connectivity when selecting target areas for conservation. Conservation of streams not influenced by reservoirs may be critical, as these streams may provide important refugia for native fishes in a reservoir-dominated landscape.

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TABLE 2.1. Study streams, catchment area (km<sup>2</sup>), Strahler order, and proportion of within-catchment land-uses. Water includes streams and impoundments. Urban includes high and low intensity residential, and commercial transportation. Forest includes deciduous, evergreen and mixed forests. Grasslands includes grasslands and pasture. Agriculture includes row crops and small grains. Wetlands includes woody and emergent/herbaceous wetlands.

Stream	Connectivity category	Catchment area (km <sup>2</sup> )	Strahler order	%Water	%Urban	%Forest	%Grasslands	%Agriculture	%Wetlands
Baldwin	DC	34.00	3	2.11	0.61	7.64	72.21	14.84	0.83
Carnahan	DC	89.62	3	1.55	0.20	5.75	75.13	14.52	0.30
Cedar	IC	179.66	3	0.39	0.01	2.74	57.06	38.32	0.74
Fancy	IC	473.44	5	0.28	0.04	0.98	55.52	42.10	0.44
Fourmile	CS	23.67	3	1.19	3.39	6.12	64.93	20.91	0.48
Huntress	IC	79.19	4	0.17	0.93	0.81	46.90	49.90	0.25
Kitten	CS	14.65	3	0.40	0.82	7.34	76.60	13.94	0.42
Madison	DC	47.55	3	0.32	0.18	5.34	82.12	11.22	0.29
Mall	IC	116.96	3	0.65	0.18	1.59	57.01	39.49	0.48
McDowell	CS	268.26	4	0.42	0.36	2.14	77.74	11.61	0.35
McIntyre	DC	62.67	3	2.75	0.04	3.25	79.57	8.66	0.36
Mill	DC	106.72	3	1.60	0.20	5.25	63.95	26.46	0.48
Mulberry	IC	26.55	3	0.03	0.41	3.15	55.60	40.49	0.14
North Otter	IC	71.08	4	0.24	0.00	2.44	65.40	31.32	0.33
Rock	CS	612.62	5	0.58	0.27	3.79	68.91	22.25	0.48
Sevenmile	CS	98.46	3	2.17	1.74	13.22	64.99	12.89	0.66
Swede	IC	90.64	3	0.51	0.01	4.42	66.63	27.49	0.41
Threemile	CS	74.04	3	0.48	4.09	15.08	69.41	3.13	0.32
Timber	DC	96.16	3	0.77	0.31	5.01	67.90	23.08	0.61
Walnut	IC	75.02	3	1.27	0.20	3.19	62.96	31.21	0.70
Mean	-	132.05	-	0.89	0.70	4.96	66.53	24.19	0.45
SE	-	34.28	-	0.17	0.25	0.83	2.05	2.92	0.04

TABLE 2.2. Relative abundance (proportion among total collected) and number of sites occupied for fish species collected at 41 sites in the Flint Hills, Kansas USA. Codes are the first three letters of the genus and specific epithet. Analysis categories are (Native = N, Reservoir = R, Introduced = I).

Scientific name	Common Name	Code	Category	Sites occupied	Relative abundance
<i>Ameiurus melas</i>	Black bullhead	AMEMEL	N	20	0.52
<i>A. natalis</i>	Yellow bullhead	AMENAT	N	25	0.32
<i>Aplodinotus grunniens</i>	Freshwater drum	APLGRU	N,R	6	0.05
<i>Campostoma anomalum</i>	Central stoneroller	CAMANO	N	37	21.99
<i>Carpionodes carpio</i>	River carpsucker	CARCAR	N,R	11	0.23
<i>C. cyprinus</i>	Quillback	CARCYP	N,R	2	0.02
<i>Catostomus commersonii</i>	White sucker	CATCOM	N	28	0.56
<i>Cyprinella lutrensis</i>	Red shiner	CYPLUT	N	37	22.51
<i>Cyprinus carpio</i>	Common carp	CYPCAR	R,I	17	0.36
<i>Dorosoma cepedianum</i>	Gizzard shad	DORCEP	N,R	12	2.89
<i>Etheostoma nigrum</i>	Johnny darter	ETHNIG	N	18	0.57
<i>E. spectabile</i>	Orangethroat darter	ETHSPE	N	37	5.06
<i>Gambusia affinis</i>	Western mosquitofish	GAMAFF	I	18	3.86
<i>Ictalurus punctatus</i>	Channel catfish	ICTPUN	N,R	19	0.23
<i>Ictiobus bubalus</i>	Smallmouth buffalo	ICTBUB	N,R	9	0.20
<i>I. cyprinellus</i>	Bigmouth buffalo	ICTCYP	N,R	11	0.80
<i>Lepisosteus osseus</i>	Longnose gar	LEPOSS	N,R	11	0.12
<i>L. platostomus</i>	Shortnose gar	LEPPLA	N	2	0.01
<i>Lepomis cyanellus</i>	Green sunfish	LEPCYA	N	38	2.99
<i>L. humilis</i>	Orangespotted sunfish	LEPHUM	N	25	2.80
<i>L. macrochirus</i>	Bluegill	LEPMAC	R,I	27	1.73
<i>L. macrochirus x L. cyanellus</i>	Bluegill x Green sunfish hybrid	LEPHYB	-	5	0.05
<i>L. megalotis</i>	Lonear sunfish	LEPMEG	N	9	0.48
<i>L. microlophus</i>	Redear sunfish	LEPMIC	I	1	0.01
<i>Luxilus cornutus</i>	Common shiner	LUXCOR	N	21	5.01
<i>Lythrurus umbratilis</i>	Redfin shiner	LYTUMB	N	19	4.00
<i>Micropterus salmoides</i>	Largemouth bass	MICSAL	R,I	26	1.09
<i>Morone chrysops</i>	White bass	MORCHR	N,R	9	0.57
<i>Moxostoma erythrurum</i>	Golden redbreast	MOXERY	N	2	0.01
<i>M. macrolepidotum</i>	Shorthead redbreast	MOXMAC	N	8	0.10
<i>Notemigonus crysoleucas</i>	Golden shiner	NOTCRY	R,I	5	0.03
<i>Notropis atherinoides</i>	Emerald shiner	NOTATH	R,I	6	0.05
<i>N. percbromus</i>	Carmine shiner	NOTPER	N	20	1.06
<i>N. stramineus</i>	Sand shiner	NOTSTR	N	3	1.08
<i>N. topeka</i>	Topeka shiner	NOTTOP	N	1	0.05
<i>Noturus exilis</i>	Slender madtom	NOTEXI	N	10	0.17
<i>N. flavus</i>	Stonecat	NOTFLA	N	13	0.19
<i>Percina caprodes</i>	Logperch	PERCAP	N	15	0.14
<i>Phenacobius mirabilis</i>	Suckermouth minnow	PHEMIR	N	23	0.62
<i>Phoxinus erythrogaster</i>	Southern redbelly dace	PHOERY	N	7	5.26
<i>Pimephales notatus</i>	Bluntnose minnow	PIMNOT	N	35	6.61
<i>P. promelas</i>	Fathead minnow	PIMPRO	N	24	0.61
<i>P. vigilax</i>	Bullhead minnow	PIMVIG	R,I	9	0.27
<i>Pomoxis annularis</i>	White crappie	POMANN	R,I	10	1.02
<i>P. nigromaculatus</i>	Black crappie	POMNIG	N,R	1	0.00
<i>Pylodictis olivaris</i>	Flathead catfish	PYLOLI	N,R	3	0.01
<i>Sander vitreus x S. canadensis</i>	Saugeye	SANHYB	R,I	5	0.03
<i>Semotilus atromaculatus</i>	Creek chub	SEMATR	N	29	3.64

TABLE 2.3. Codes and descriptions of landscape-scale environmental variables.

Variable code	Variable description
WSHED	Catchment area (km <sup>2</sup> )
STRAHLER	Strahler stream order
GRADIENT	Stream reach gradient (m/km)
WATER	Proportion of streams and impoundments in catchment
URBAN	Proportion of urban area in catchment
FOREST	Proportion of forested area in catchment
GRASS	Proportion of grasslands in catchment
AG	Proportion of agriculture in catchment
WET	Proportion of wetlands in catchment

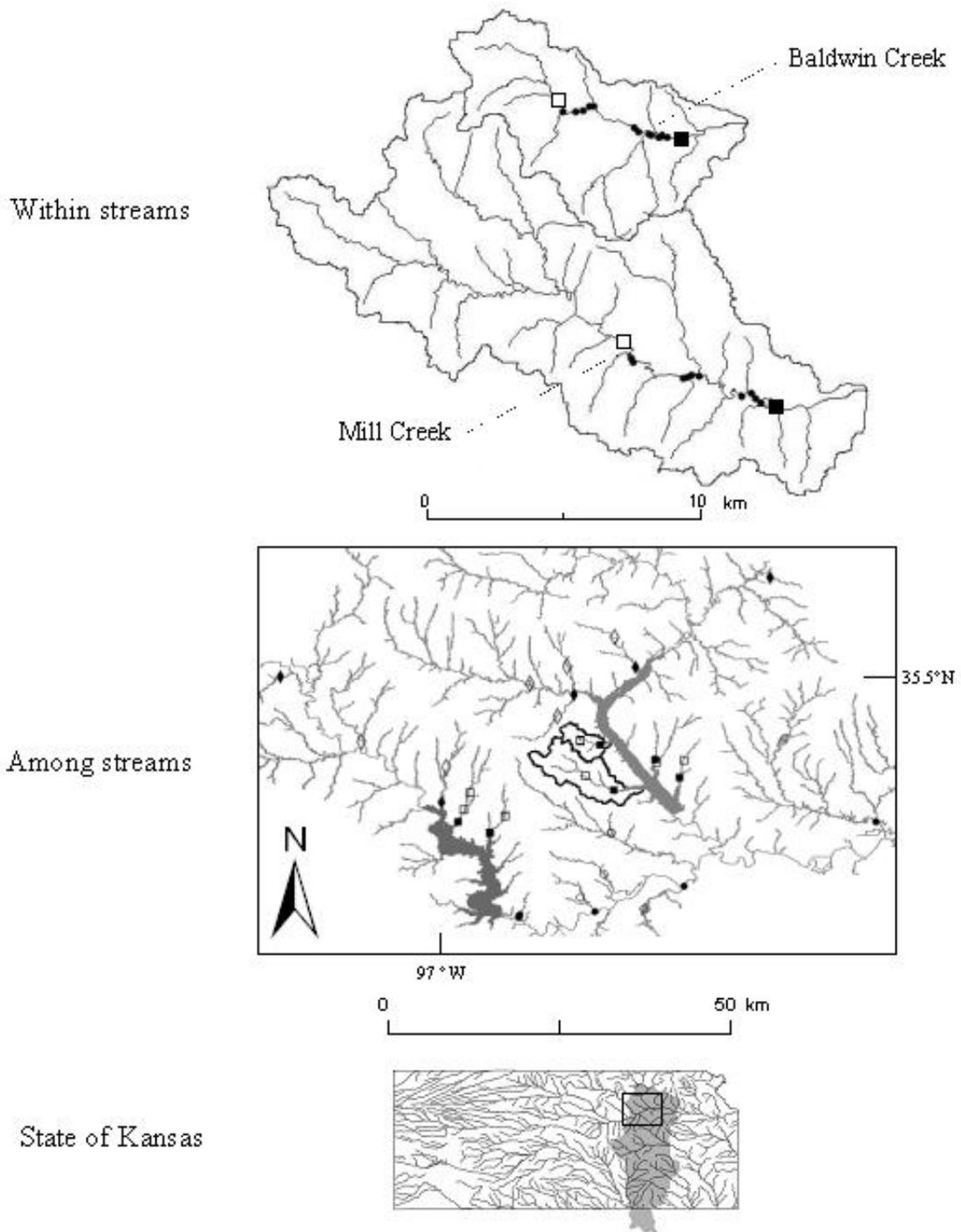


Fig. 2.1. Study area showing locations of sample sites within streams (N = 41 sites), and among streams (N = 22 pools) in the Flint Hills, Kansas, USA. Among streams, solid symbols represent CFL sites; open symbols represent MID sites. Square symbols are DC streams, diamonds are IC streams, and circles are CT streams. Within streams, pools are represented by closed circles. Location of 2003 MID and CFL sites are provided for reference (closed squares = CFL sites, open squares = MID sites).

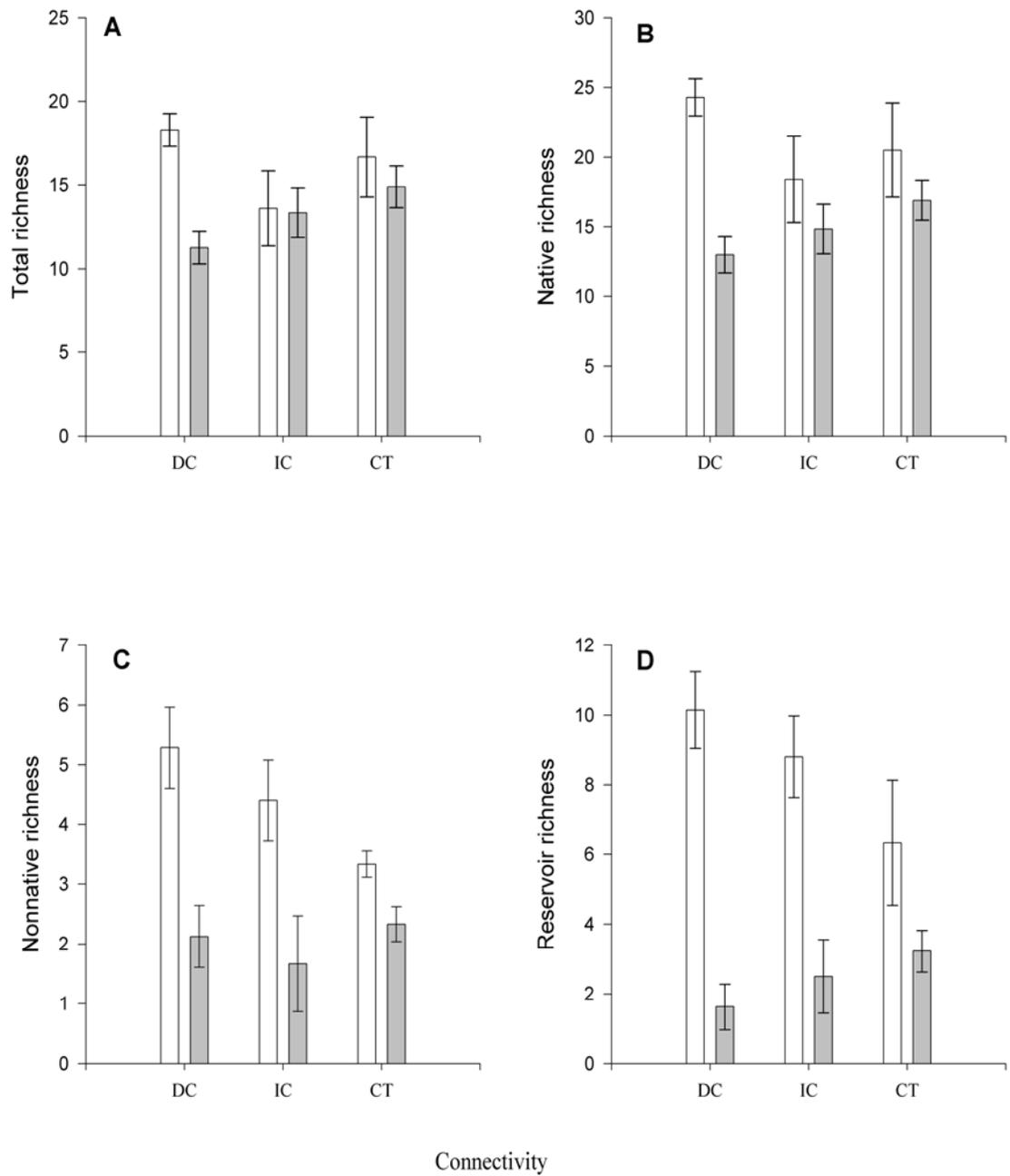


Fig. 2.2. Mean ( $\pm 1$  SE) total species richness (A), native species richness (B), nonnative species richness (C), and reservoir species richness (D) among streams that differ in connectivity to a reservoir (DC = directly connected, IC = indirectly connected, CT = connected to the Kansas River) and between longitudinal positions. Open bars are CFL sites, and closed bars are MID sites.

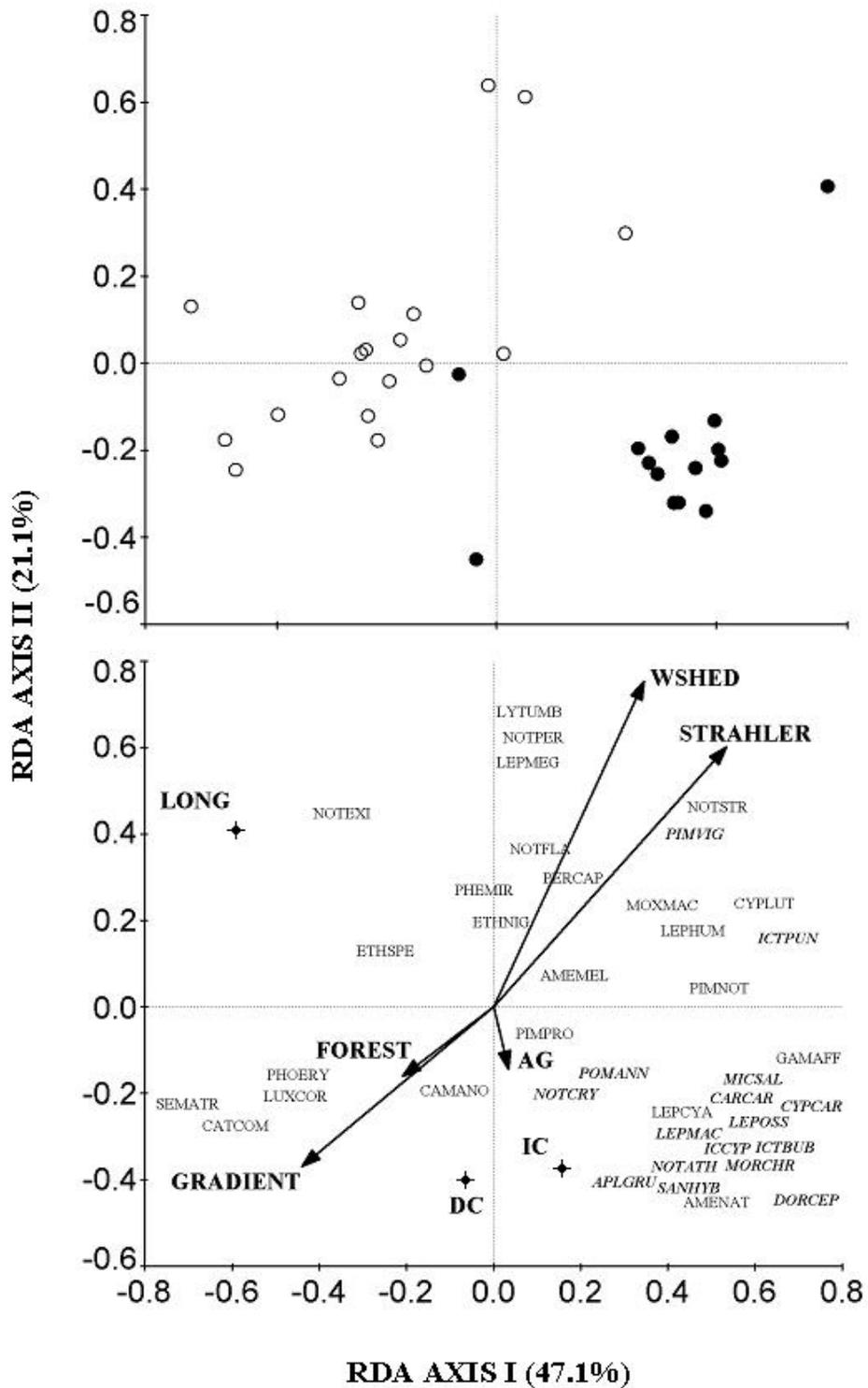


Fig. 2.3. Association of fish species and environmental variables, longitudinal position (LONG), and connectivity type (DC = directly connected, IC = indirectly connected, CT = connected to the Kansas River) (bottom panel) from a RDA. Site scores are plotted in the top panel, open circles represent MID sites, closed circles represent CFL sites. Species codes are defined in Table 2.1, environmental variables are defined in Table 2.3.

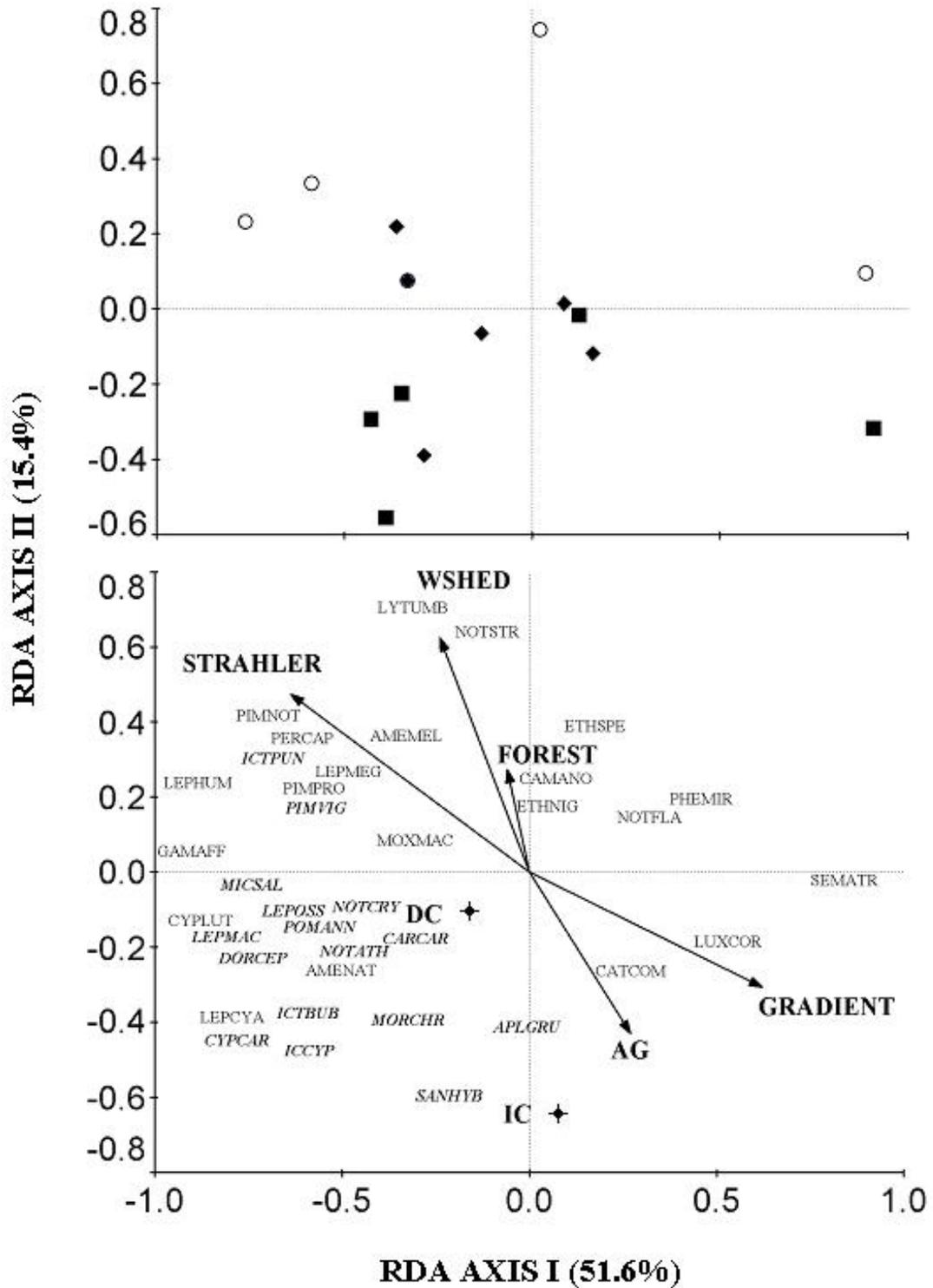


Fig. 2.4. Association of fish species environmental variables, and connectivity type (DC = directly connected, IC = indirectly connected, CT = connected to the Kansas River) (bottom panel) at CFL sites from a RDA. Site scores are plotted in the top panel, diamonds represent DC sites, squares represent IC sites, and circles CT sites. Species codes are defined in Table 2.1, environmental variables are defined in Table 2.3.

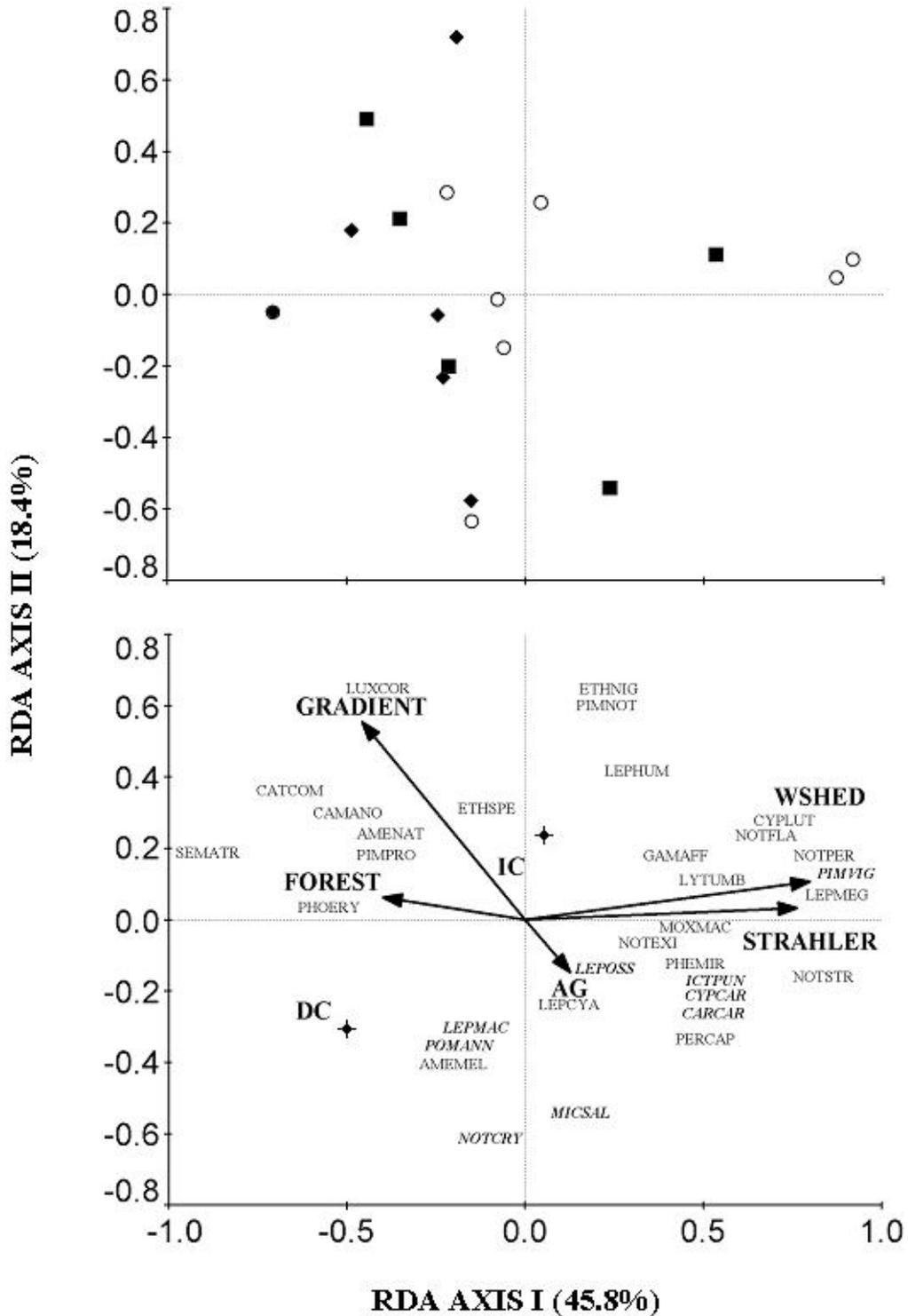


Fig. 2.5. Association of fish species environmental variables, and connectivity type (DC = directly connected, IC = indirectly connected, CT = connected to the Kansas River) (bottom panel) at MID sites from a RDA. Site scores are plotted in the top panel, diamonds represent DC sites, squares represent IC sites, and circles CT sites. Species codes are defined in Table 2.1, environmental variables are defined in Table 2.3.

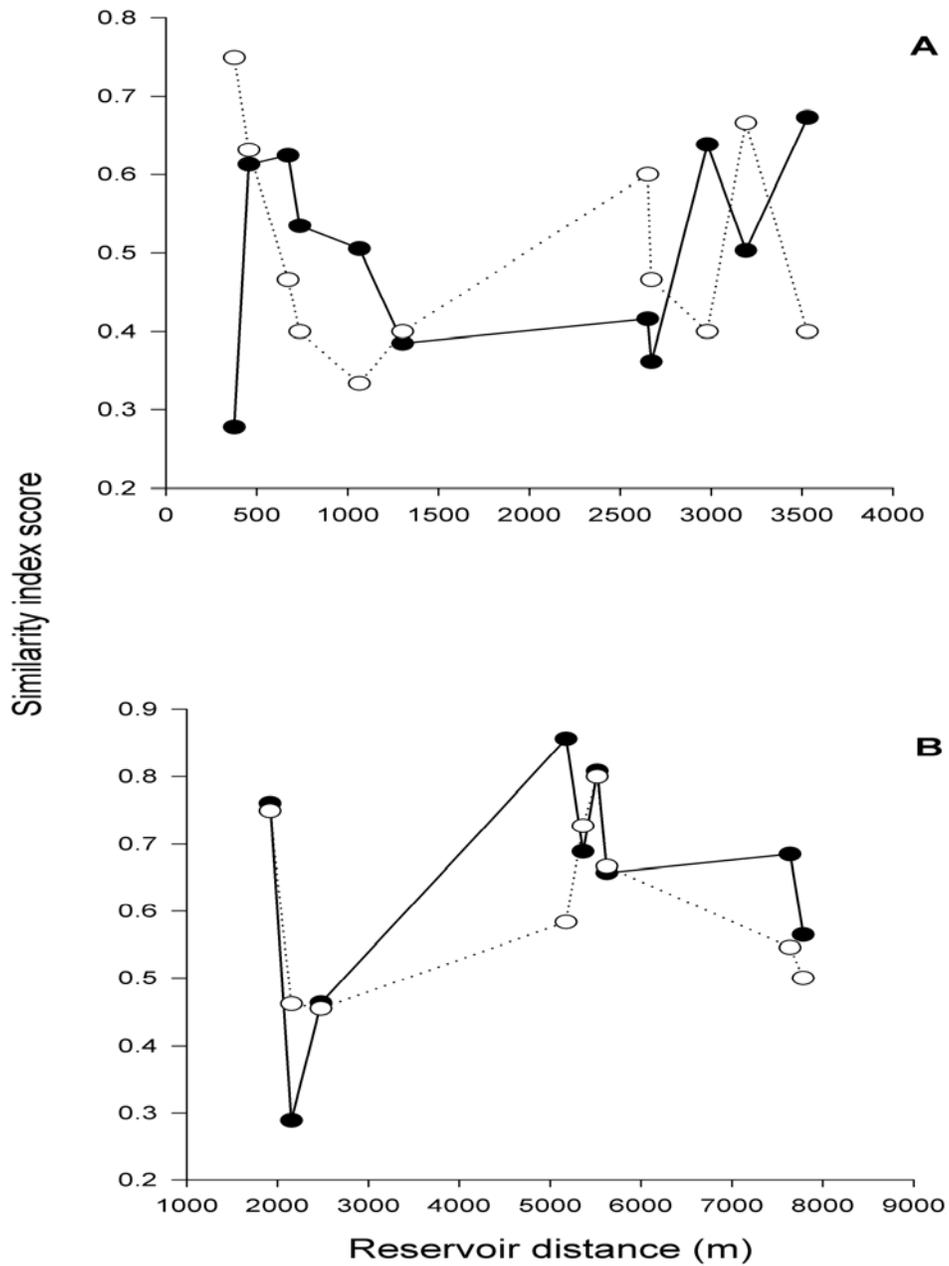


Fig. 2.6. Similarity of pool fish assemblages in Baldwin (top panel) and Mill (bottom panel) creeks versus distance from a reservoir. Assemblage similarity was compared between each pool and the pool closest to the reservoir using PSI (closed circles) and Jaccard's (open circles) indices.

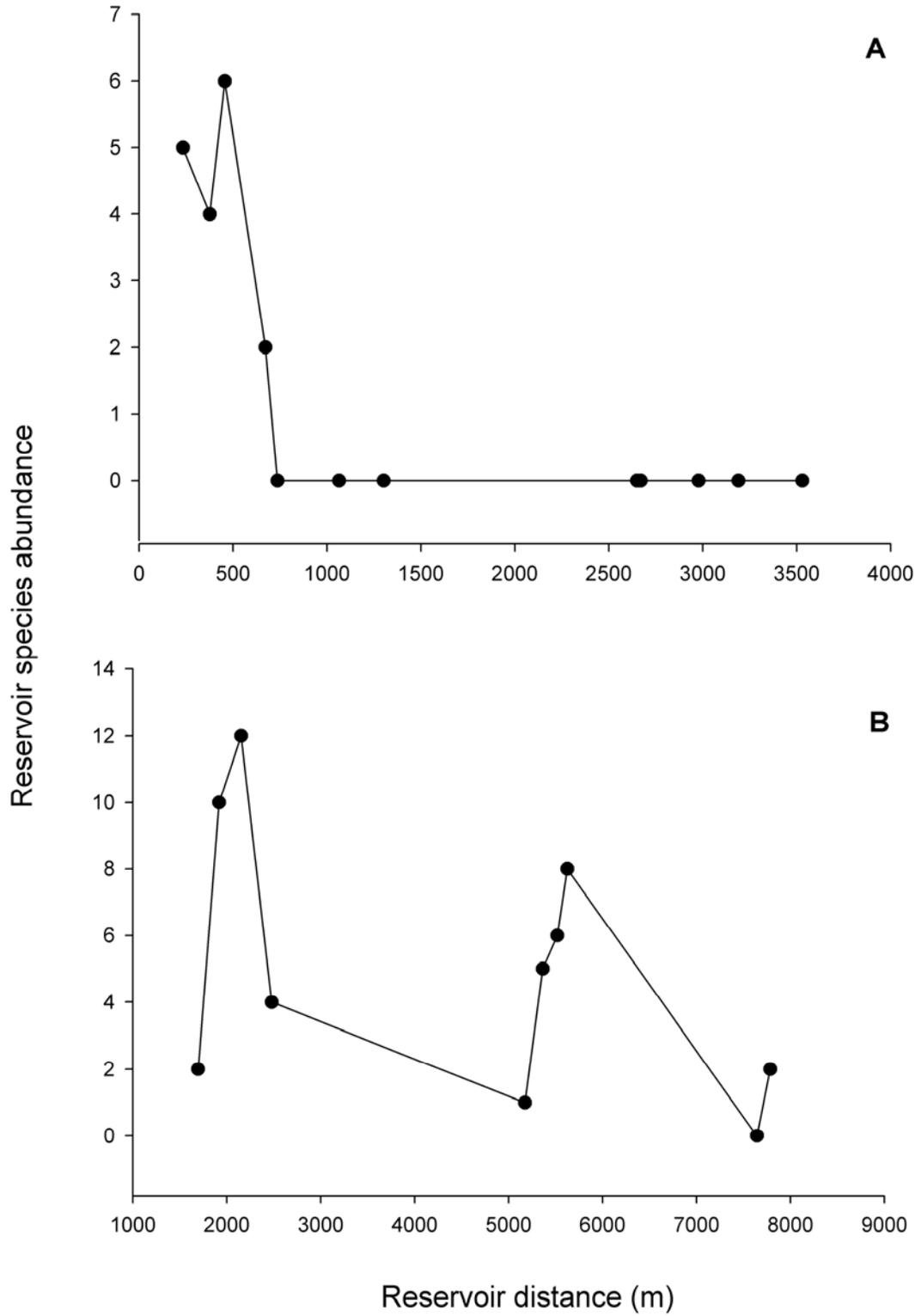


Fig. 2.7. Abundance of reservoir species in pools in Baldwin (A; top panel) and Mill (B; bottom panel) creeks versus distance from a reservoir (m).