

Historical and Current Environmental Influences on an Endemic Great Plains Fish

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ABSTRACT.—Native fishes of the Great Plains are at risk of decline due to disturbances to physical habitat caused by changes in land and water use, as well as shifts in species assemblages driven by the invasion of introduced species with the loss of natives. We used historical and current fish assemblage data in conjunction with current habitat information to assess these influences on an endemic Great Plains stream fish, the plains topminnow (*Fundulus sciadicus*). Of the 31 sites where the plains topminnow occurred historically (1939–1940), it was found in only seven of those sites in 2003–2005. Our results demonstrate a shift in fish assemblage over time that coincides with the loss of plains topminnow. Changes in fish assemblages were characterized by increases in occurrence of exotic, invasive and generalist species with declines in occurrences of native fishes. An information theoretic approach was used to evaluate candidate models of current fish assemblage and physical/chemical habitat on the presence of the plains topminnow. Candidate models that included both instream habitat (*e.g.*, vegetation coverage, undercut banks) and the native fish species assemblage are important to predicting presence of the plains topminnow within its historic range. Conservation of Great Plains fishes including the plains topminnow will need a combination of habitat protection and enhancement.

INTRODUCTION

Freshwater fishes are at risk of decline in range and local abundance resulting in extirpation and eventual species loss due to widespread anthropogenic disturbances. Human impacts are the primary cause of decline and extinction of fishes (Miller *et al.*, 1989; Williams *et al.*, 1989; Richter *et al.*, 1997; Ricciardi and Rasmussen, 1999). Disturbance to physical habitat (*e.g.*, riparian vegetation alteration, impoundments; Gorman and Karr, 1978; Jones *et al.*, 1999; Marchetti and Moyle, 2000; Quinn and Kwak, 2003), water quality (*e.g.*, pollution, sedimentation; Tsai, 1973; Bonner and Wilde, 2002) and the introduction of nonnative species (Soule, 1990; Rahel, 2002) have resulted in fish assemblage shifts, decreased native species diversity, community homogenization, range reduction and extinction.

The Great Plains, one of the largest biomes in North America, has been subjected to widespread disturbances to both its terrestrial and aquatic resources. The native grasslands (162 million ha) of the Great Plains are one of the nations most threatened ecosystems with losses of up to 99.9% in range (Samson and Knopf, 1994). However, the region's aquatic ecosystems have received less attention compared to other regions (Matthews, 1988).

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The conversion of the Great Plains to agriculture and replacement of native terrestrial grazers (*i.e.*, bison) with domesticated livestock have increasingly affected prairie stream ecosystems and fish assemblage structure by increasing siltation and decreasing foraging efficiency of native species (Bonner and Wilde, 2002) and decreasing instream habitat diversity through alteration of riparian habitat and channel morphology (Gorman and Karr, 1978; Jones *et al.*, 1999). The change in physical habitat due to impoundments has also been associated with assemblage changes, usually causing declines in native lotic species and diversity, while nonnative lentic fishes increase in abundance and range (Minckley and Deacon, 1968; Winston *et al.*, 1991; Patton and Hubert, 1993; Marchetti and Moyle, 2000; Minckley *et al.*, 2003; Falke and Gido, 2006).

The conservation and management of freshwater fish is often difficult. For example, detecting long-term changes in fish assemblages is often problematic because of the lack of historical documentation before alterations to the ecosystem occurred (Matthews, 1988; Fausch and Bestgen, 1997). It is this lack of baseline data for which to evaluate the status of species that creates difficulties in preventing the listing of potential species as vulnerable or in need of conservation (Patton *et al.*, 1998; Rahel *et al.*, 1999). Furthermore, historic data often consists of species lists containing presence/absence data that requires caution in comparison to current data (Patton *et al.*, 1998).

The plains topminnow is an endemic species to the Great Plains that appears to be experiencing reduction in its range and declines in local abundance. The historic distribution of the plains topminnow extends from northeastern Oklahoma to southern South Dakota and southwestern Minnesota and northwestern Iowa to eastern Wyoming and Colorado, with disjunct population centers in Nebraska and Missouri (Lee *et al.*, 1980). In Minnesota, Missouri, South Dakota and Wyoming the plains topminnow is currently listed as vulnerable, where Kansas and Oklahoma lists the species as critically imperiled and in Iowa the status is possibly extirpated. Only in Nebraska and Colorado is the status apparently secure with a greater historical distribution occurring in north central Nebraska (Nature Serve, 2006). Within Nebraska, the species is most common in the Sandhills region (Stasiak, 1987) and was historically documented throughout the Platte River drainage (Lee *et al.*, 1980).

Current environmental conditions (*e.g.*, altered stream flows, introduction of nonnative fishes, loss/fragmentation of habitat, water quality degradation) may be severe threats to the plains topminnow and conservation of this species may benefit a number of similar native stream fishes in addition to other aquatic fauna (Rahel and Thel, 2004). Our objectives were to: (1) evaluate the influence of fish assemblage change on plains topminnow presence from the species historical (1939–1940) and current (2003–2005) range within Nebraska and (2) determine the effects the physiochemical characteristics and fish assemblage on the current presence and distribution of the plains topminnow throughout its historic range in Nebraska through the use of information theoretic methods employing *a priori* model selection of predictor variables. We hypothesized that the decline in the plains topminnows historical range is associated with a shift in fish assemblages to non-native species and the presence of the plains topminnow within its current range is negatively associated with habitat modification and degradation.

METHODS

STUDY AREA

We used both historical and current surveys of Nebraska streams to assess past and current plains topminnow distributions. The most extensive historical survey of Nebraska streams was conducted by Raymond E. Johnson between 1939 and 1940 where plains topminnow were

collected at 31 of 215 stream sites throughout Nebraska (Johnson, 1942). These occurrences were primarily located within the Sandhills region of northcentral Nebraska and tributaries to the Platte River. To compare the 1939–1940 plains topminnow distribution with current distributions, we resampled 52 of the 215 sites between 2003 and 2005. Twenty one additional revisited sites on streams with historical presence were included. Although these additional sites did not produce plains topminnow in the historical survey, we assumed there may be a high likelihood of the species presence given these sites were from the same streams where plains topminnow were historically collected, and thus important to understanding potential influences on plains topminnow distribution.

FIELD COLLECTION

Seining was the primary method of fish collection for the historical data. Common seines (1.8, 3.0 and 7.6 m) were used in smaller streams and a 7.6 m bag seine in larger streams. In rare occasions dip nets, hook and line, and rotenone was used to collect fish. All fish collected were identified in the field or preserved and identified in the laboratory. Sampling was conducted from May through Aug. in the summers of 1939 and 1940.

In the summers of 2003 to 2005, the 52 sites sampled by Johnson (1942) were sampled by backpack or tote barge electrofisher in a single upstream pass. Prior to sampling, block nets were established at the upstream and downstream ends to prevent fish movement out of the site. Easily-identified specimens were released in the field and unidentifiable or numerous specimens were preserved in 10% formalin and identified in the laboratory.

Physical and chemical habitat data were measured at all sites sampled in 2003 to 2005 with a modified version of the Environmental Protection Agency's (EPA) protocol for sampling wadable streams (Lazorchak *et al.*, 1998). The only modification to EPA protocol was the reduction of 11 evenly spaced transects to six. Sampling length was calculated to be 40 times the mean stream width (MSW) with a minimum length of 150 m and a maximum length of 300 m. Six evenly spaced transects for each sampling length was used to survey each site's physical habitat. Prior to fish sampling and physical habitat measurements, *in situ* water chemistry was measured at the downstream transect. Dissolved oxygen, specific conductance, water temperature and turbidity were measured using a handheld meter.

Physical habitat was measured after fish sampling at five equally spaced intervals between the six transects. Mean depth was measured at three equally spaced intervals for each transect and the mean of all transects was calculated for each site. Wetted width was measured at 10 equally spaced intervals between transects and the mean of all 50 measurements was calculated. Canopy cover was measured using a modified handheld convex spherical densiometer at six locations at each transect and the percent cover for each transect was averaged for each stream (Lazorchak *et al.*, 1998).

Instream fish cover (filamentous algae, aquatic macrophytes, woody debris, overhanging vegetation and undercut banks) was estimated using a rank of 5 cover classes; absent (0%), sparse (<10%), moderate (10 to 40%), heavy (40 to 75%) and very heavy (>75%) (Lazorchak *et al.*, 1998). Mean rank was then averaged across all six transects for each stream to obtain a value for all of the instream fish cover categories for each stream.

HISTORIC-CURRENT ANALYSIS

We used presence/absence data from the 31 historical sites of plains topminnow presence in our analyses due to a lack of enumeration of all species at all sites in the historical survey. Our goal was to determine if a shift in fish assemblage occurred between historical sites where plains topminnow occurred in the last 65 y. A canonical discriminant analysis (CDA) was used to determine if fish assemblages differed between current and historical sites with

and without plains topminnow (Johnson, 1998). In this analysis, differences in assemblages between sites of historical presence, current presence and current absence of the plains topminnow were examined. Because plains topminnow presence was used to generate the groups, the species was not used in the CDA. Canonical variables were calculated for each group and plotted. The mean value of each group, known as centroids, were compared using Mahalanobis Distance D^2 (Johnson, 1998) to determine if fish assemblage differed between groups. Species that occurred at 5% or less (2 sites) of the 31 sites where the plains topminnow occurred historically were removed (Appendix 1). We then used logistic regression with backward selection to determine the species most associated with the presence/absence of plains topminnow (Johnson, 1998). In this analysis, the presence/absence of plains topminnow was modeled as the response variable with the occurrence of other fish species collected as categorical explanatory variables.

CURRENT HABITAT AND FISH SPECIES MODELING

We used an information theoretic approach to investigate the influence of habitat and fish assemblage on plains topminnow presence. Variables included in the current analysis of plains topminnow presence were selected *a priori* based on the life history of the species. Rahel and Thel (2004) discussed many potential limiting factors in an extensive assessment of the plains topminnow, including physiochemical habitat degradation (*e.g.*, increased turbidity and, thus, loss of aquatic macrophytes required for reproduction, eutrophication, etc.) and the introduction of nonnative competitors and predators. These variables are believed to be associated with the presence of species and thus most useful in the management and conservation of the species.

We conducted two separate principal components analyses (PCA) to reduce the dimensionality of the: (1) current habitat conditions and (2) current fish assemblage (Johnson, 1998; Paukert and Wittig, 2002). Variables selected for the habitat PCA were based *a priori* on variables that may influence presence of plains topminnow (Rahel and Thel, 2004) and included metrics of habitat degradation (*e.g.*, turbidity, macrophytes, etc.). Physiochemical habitat variables were log transformed to better meet the assumptions of normality. Principal components with greater than 10% variance explained were retained from PCA on the correlation matrix of the habitat variables.

Fish species included in the PCA were selected on potential mechanisms limiting plains topminnow presence throughout the region (*e.g.*, competition, predation, native species abundance) and were based on the results of the CDA. These included several native and introduced species; creek chub (*Semotilus atromaculatus*), fathead minnow (*Pimephales promelas*), green sunfish (*Lepomis cyanellus*), largemouth bass (*Micropterus salmoides*), western mosquitofish (*Gambusia affinis*), white suckers (*Catostomus commersonii*), that have been documented in occurrence with or proposed to potentially limit plains topminnow populations throughout the region (Owen *et al.*, 1981; Lynch and Roh, 1996; Patton, 1997; Rahel and Thel, 2004) in addition to species collected both currently within the species range in Nebraska that occurred at greater than 10% (five sites) of the 52 total sites. Relative abundances (number of fish/min. of electrofishing) were log transformed to better meet the assumptions of normality. The two species of greatest abundance, sand shiner (*Notropis stramineus*) and red shiner (*Cyprinella lutrensis*), accounted for 51% (17,601 individuals) of 34,444 fish collected and were ubiquitous across sites. Therefore, these species were excluded from the PCA to remove the disproportional influence of relative abundance. A SCREE plot of eigenvalues was used to determine the number of principal components to retain from the PCA on the variance-covariance matrix, as Johnson (1998) suggested when selecting the number of PC with a PCA on the variance-covariance matrix.

TABLE 1.—Principal component (PC) scores of *a priori* selected physiochemical characteristics with percent and cumulative variance explained of the first 3 principal components

Variable	PC1	PC2	PC3
Algae (% instream cover)	−0.388	0.276	0.093
Macrophytes (% instream cover)	−0.293	0.360	0.436
Overhanging vegetation (% instream cover)	0.266	0.391	0.353
Undercut banks (% instream cover)	0.298	0.351	−0.151
Woody debris (% instream cover)	0.398	0.114	0.069
Canopy cover (% cover)	0.239	0.336	−0.088
Turbidity (NTUs)	0.373	−0.174	0.263
Temperature (°C)	0.012	−0.089	0.582
Dissolved oxygen (mg/L)	−0.334	0.192	0.002
Specific conductivity (µS)	−0.161	−0.328	−0.139
Mean width (m)	0.305	−0.300	0.138
Mean depth (cm)	0.135	0.346	−0.442
Eigenvalue	3.16	2.10	1.46
Percent variance explained	26.31	17.42	12.20
Cumulative variance explained	26.31	43.73	55.93

Multiple logistic regression models were created with SAS (PROC GENMOD, Stokes *et al.*, 1995) to determine the effect of habitat and fish assemblage on the current presence of the plains topminnow using an information theoretic model selection process (Burnham and Anderson, 2002). *A priori* sets of explanatory factors were created from biologically interpretable principal components scores of habitat and fish assemblage variables. Habitat and fish assemblage variables were selected on the basis of previous knowledge for the species and the likely influence of these variables on plains topminnow presence. Three habitat principal components variables and four assemblage principal component variables (Tables 1, 2) were used to create a full model. The full model fit was evaluated using plots of standardized residuals versus predicted values and Hosmer-Lemeshow statistic (Agresti, 1996). Full and reduced habitat and fish assemblage models were created separately using all subsets of the principal component scores for the 52 sites. A total of 23 logistic models of plains topminnow presence were developed from the full model and 22 subsets of explanatory variables to determine the most parsimonious model using Akaike's Information Criteria for small sample sizes (AIC_c). All models were ranked using AIC_c and Akaike weights (w_i) were calculated to compare competing models (Burnham and Anderson, 2002). Models with the lowest AIC_c and highest weights are considered the best approximating models. The relative importance of individual model parameters were evaluated by model averaging where parameter estimates for individual models is scaled and ranked by the Akaike weight (w) given model (i). This approach increases precision and reduces bias given selection uncertainty and is especially useful when multiple candidate models exhibit support of the dependent variable of interest (Burnham and Anderson, 2002). The unconditional confidence intervals for model averaged parameters were created with methods by Buckland *et al.* (1997).

RESULTS

A total of 46 species including the plains topminnow and one hybrid were collected by Johnson (1942) in the 52 sites. Current sampling collected 58 species and one hybrid. Two

TABLE 2.—Principal component (PC) scores of fish species occurring at greater than 10% of 52 sites occurring within plains topminnow range in the Nebraska Sandhills with percent and cumulative variance explained of the first four principal components

Variable	PC1	PC2	PC3	PC4
Bigmouth shiner <i>Notropis dorsalis</i>	0.730	0.079	-0.426	-0.067
Bluegill <i>Lepomis macrochirus</i>	-0.035	-0.050	-0.021	-0.079
Brassy minnow <i>Hypognathus hankinsoni</i>	-0.008	-0.032	0.004	0.038
Brook stickleback <i>Culea inconstans</i>	-0.011	-0.080	0.305	0.105
Central stoneroller <i>Camptostoma anomalum</i>	0.288	-0.034	0.472	-0.006
Channel catfish <i>Ictalurus punctatus</i>	-0.007	-0.056	-0.027	0.016
Common carp <i>Cyprinus carpio</i>	0.062	-0.028	-0.035	0.043
Creek chub <i>Semotilus atromaculatus</i>	0.104	0.457	0.422	-0.154
Emerald shiner <i>Notropis atherinoides</i>	-0.007	-0.015	-0.008	-0.003
Fathead minnow <i>Pimephales promelas</i>	0.162	0.076	-0.118	0.812
Flathead chub <i>Platygobio gracilis</i>	-0.004	-0.020	-0.019	-0.009
Green sunfish <i>Lepomis cyanellus</i>	0.010	-0.060	-0.070	0.038
Iowa darter <i>Etheostoma exile</i>	0.013	-0.018	-0.013	0.097
Largemouth bass <i>Micropterus salmoides</i>	-0.034	0.026	-0.027	-0.171
Longnose dace <i>Rhinichthys cataractae</i>	-0.160	0.680	-0.033	0.316
Plains killifish <i>Fundulus kansae</i>	0.527	-0.068	0.412	0.051
Plains minnow <i>Hypognathus placitus</i>	0.026	0.004	-0.086	0.049
Quillback <i>Carpionodes cyprinus</i>	0.001	-0.016	-0.018	-0.001
River shiner <i>Notropis blennioides</i>	0.160	-0.103	-0.288	-0.108
Shorthead redhorse <i>Moxostoma macrolepidotum</i>	0.009	0.021	-0.033	-0.050
Stonecat <i>Noturus flavus</i>	-0.001	0.012	-0.010	0.008
Western mosquitofish <i>Gambusia affinis</i>	-0.015	-0.018	0.080	-0.017
White sucker <i>Catostomus commersonii</i>	0.099	0.530	-0.179	-0.363
Eigenvalue	0.78	0.58	0.40	0.32
Percent variance explained	25.1	18.6	12.8	10.3
Cumulative variance explained	25.1	43.7	56.4	66.7

species and the one hybrid collected by Johnson but not found currently were common shiner (*Luxilus cornutus*), tadpole madtom (*Noturus gyrinus*) and northern redbelly dace × finescale dace (*Phoxinus eos* × *neogaeus*). Species collected currently but not historically include bluntnose minnow (*P. notatus*), brown trout (*Salmo trutta*), flathead catfish (*Pylodictus olivaris*), freshwater drum (*Aplodinotus grunniens*), gizzard shad (*Dorosoma cepedianum*), Johnny darter (*Etheostoma nigrum*), longnose gar (*Lepisosteus osseus*), northern pike (*Esox lucius*), rock bass (*Ambloplites rupestris*), shortnose gar (*L. platostomus*), smallmouth bass (*M. dolomieu*), western mosquitofish, white crappie (*Pomoxis annularis*) and yellow perch (*Perca flavescens*). One hybrid (green sunfish × bluegill *L. macrochirus* × *cyanellus*) was collected currently, but not historically. Revisiting the 31 sites of historical presence yielded the plains topminnow at only seven sites (22.6%). Five of the 21 sites additionally sampled contained plains topminnow for a total of 12 of the 52 sites visited in 2003–2005.

A total of 38 species (excluding the plains topminnow) were used in the CDA (Fig. 1). Seven species and one hybrid were removed from analysis due to occurrence at less than 5% of the sites. The fish assemblage differed between sites of historical plains topminnow presence and sites currently absent of plains topminnow ($D^2 = 13.9$, $F_{38,22} = 1.84$, $P = 0.065$), whereas assemblage did not differ between sites with historic and current plains topminnow presence ($D^2 = 14.0$, $F_{38,22} = 0.79$, $P = 0.744$). However, the fish assemblage of

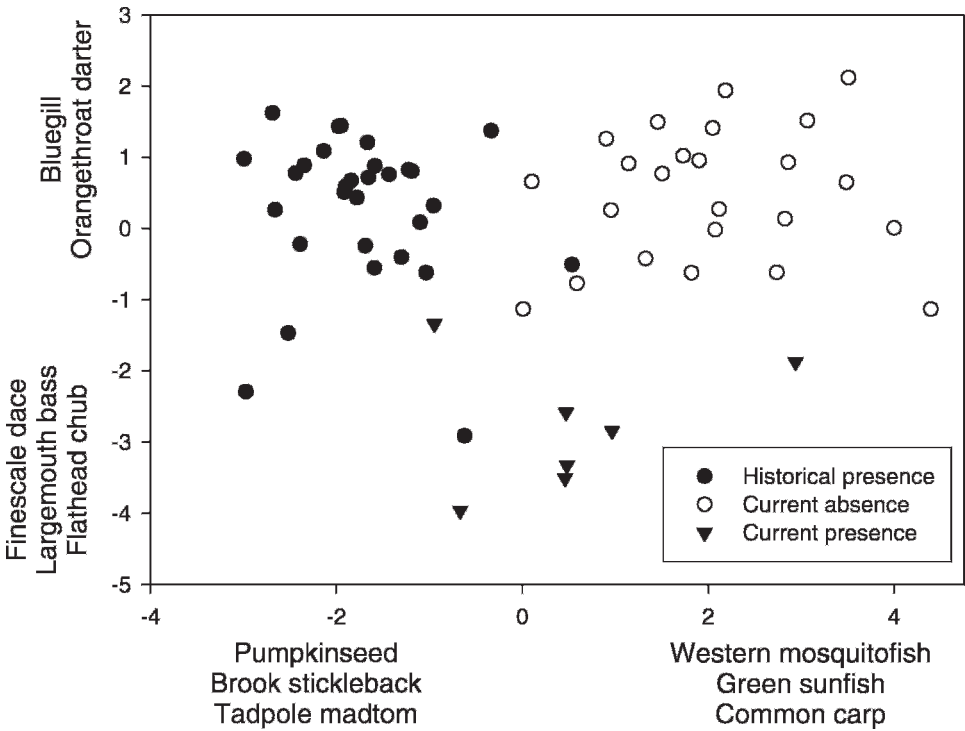


FIG. 1.—Canonical discriminant analysis used to determine fish assemblage differences between sites of historical presence, current presence, and current absence of the plains topminnow in Nebraska from the current sampling (2003–2005) of 31 historical sites sampled in 1939–1940 (Johnson 1942) where plains topminnow was collected

currently sampled sites with and without plains topminnow did not differ ($D^2 = 12.9$, $F_{38,22} = 0.68$, $P = 0.851$). Although current and historical sampling was conducted at the same sites, the fish assemblage changed. The re-sampling of sites with historical plains topminnow presence resulted in differing fish assemblages collected when the plains topminnow was currently absent.

From the logistic regression model, four species: bigmouth shiner (*Noturus dorsalis*), common carp (*Cyprinus carpio*), creek chub and green sunfish, were significant determinants ($P_s < 0.05$) of the historical presence and current absence of plains topminnow. The presence of plains topminnow was approximately 6 times more likely ($P = 0.017$) when bigmouth shiner were present and over 8 times less likely when green sunfish ($P = 0.007$), common carp ($P = 0.011$) and creek chub ($P = 0.010$) were present.

The PCA of 12 habitat variables produced three axes with individual percent variances that explained greater than 10% which summed to 53.9% of the total variance explained (Table 1). The first axis had high loadings (absolute value greater than 0.30) for percent cover of algae, woody debris, turbidity, dissolved oxygen and mean width. Variables with high loadings for the second axis included percent cover of macrophytes, overhanging vegetation, undercut banks, canopy cover, specific conductivity and mean depth, while macrophytes, overhanging vegetation, temperature and depth retained high loadings on the third axis (Table 1).

TABLE 3.—Top logistic regression models (ΔAIC_c less than two) predicting plains topminnow presence as determined by the Akaike's information criterion for small sample size AIC_c rankings. K is the number of parameters included in the model (including intercept), ΔAIC_c change in AIC_c values between models and w_i is the Akaike's weight. Probability of model fit given the data in comparison to the best fitting model in parentheses

Model and parameters	AIC_c	K	ΔAIC_c	w_i
Fish assemblage PC2 and PC4	51.333	3	0.000	0.152
Habitat PC2 and PC3	52.252	3	0.915	0.097 (1.57×)
Fish assemblage PC2, PC3, and PC4	52.324	4	0.985	0.093 (1.63×)
Full	52.475	8	1.061	0.086 (1.77×)
Fish assemblage PC4	52.672	2	1.312	0.078 (1.95×)
Habitat PC2	53.083	2	1.746	0.064 (2.38×)
Fish assemblage PC1, PC2, and PC4	53.094	4	1.755	0.063 (2.41×)

The PCA of 23 fish species relative abundances produced four axes that cumulatively explained 66.7% of the variance in the 52 currently sampled sites. Fish species with high loadings (absolute value greater than 0.30) include bigmouth shiner and plains killifish (*Fundulus kansae*) for the first axis; creek chub, longnose dace (*Rhinichthys cataractae*) and white sucker for the second axis; brook stickleback (*Culea inconstans*), central stoneroller (*Camptostoma anomalum*), creek chub and plains killifish for the third axis; and fathead minnow, longnose dace and white sucker for the fourth axis (Table 2).

The best approximating model selected using AIC_c was the fish assemblage model, containing PC2 and PC3 (Table 3). However, relatively small differences in Akaike weights (w_i) indicate that multiple competing models were also useful in prediction of plains topminnow presence. Seven models had a ΔAIC_c less than two, suggesting these models have similar predictive capability. For all seven models, fish assemblage PC4 was contained in five models, while fish assemblage PC2 was in four. Habitat PC2 was contained in three models, but PC1 and PC3 were in two or fewer models. Fish assemblage PC4 and PC2 (which were gradients of fathead minnow, creek chub, longnose dace and white sucker) and habitat PC2 (a gradient of vegetation coverage, undercut banks, conductivity and mean depth) were prominent predictors of plains topminnow presence. Positive coefficients for habitat PC2 suggest increased probability of plains topminnow presence with increased coverage of macrophytes, overhanging vegetation, undercut banks, canopy cover, mean depth and decreased specific conductivity (Table 4). Positive coefficients for fish PC2 and PC4 suggest increased probability of plains topminnow presence with increased abundances of creek chub, white sucker, longnose dace and fathead minnow (Table 4).

DISCUSSION

Absence of the plains topminnow from more than 75% of the sites where it was historically found suggests this species has experienced declines in occurrence within its most secure range. This apparent decline is paralleled with a loss of other species in the Great Plains (*e.g.*, lack of current occurrence of the federally endangered Topeka shiner and state of Nebraska endangered northern redbelly dace at 215 historically sampled sites revisited in 2003–2005) and a shift towards generalists and exotic species (*e.g.*, green sunfish, creek chub and common carp associated most with current absence of plains topminnow). In addition, several other introduced species (brown trout, northern pike, smallmouth bass and western mosquitofish) were collected in 2003–2005 that were not historically collected.

TABLE 4.—Model averaged parameter estimates, standard error and 95% confidence intervals. Contributing variables with high loadings (absolute value greater than 0.30) from principal component analysis (PCA) listed under model parameter

Model parameters	Estimate	SE	95% CI	
Habitat PC1 Algae, woody debris, turbidity, dissolved oxygen, mean width	0.02	0.051	-0.08	0.12
Habitat PC2 Macrophytes, overhanging vegetation, undercut banks, canopy cover, specific conductivity, mean depth	0.76	0.125	0.51	1.00
Habitat PC3 Macrophytes, overhanging vegetation, temperature, mean depth	0.65	0.089	0.47	0.82
Fish assemblage PC1 Bigmouth shiner <i>Notropis dorsalis</i> Plains killifish <i>Fundulus kansae</i>	-0.43	0.224	-0.87	0.01
Fish assemblage PC2 Creek chub <i>Semotilus atromaculatus</i> Longnose dace <i>Rhinichthys cataractae</i> White sucker <i>Catostomus commersonii</i>	0.80	0.261	0.29	1.32
Fish assemblage PC3 Bigmouth shiner <i>Notropis dorsalis</i> Brook stickleback <i>Culea inconstans</i> Central stoneroller <i>Camptostoma anomalum</i> Creek chub <i>Semotilus atromaculatus</i> Plains killifish <i>Fundulus kansae</i>	-1.43	0.434	-2.28	-0.58
Fish assemblage PC4 Fathead minnow <i>Pimephales promelas</i> Longnose dace <i>Rhinichthys cataractae</i> White sucker <i>Catostomus commersonii</i>	1.46	0.474	0.53	2.39

This decrease in native diversity and increase in non-native, generalist and game species through human modification is consistent with the homogenization of fish assemblages in the Great Plains (Rahel, 2002).

Habitat and fish assemblage structure are both important predictors of the plains topminnow presence. Our analysis suggests instream cover (macrophytes and overhanging vegetation) are associated with the presence of plains topminnow. This is consistent with reproductive needs of the species (Pflieger, 1997) in addition to potential refugia from fish and avian predators (Owen *et al.*, 1981). Native fish assemblages were also strongly associated with plains topminnow presence. Our results are consistent with Patton (1997) who found five of the six native species (brook stickleback absent) contained with our model averaged parameter estimates (that did not contain zero within 95% confidence intervals) with the plains topminnow in Wyoming streams. A study by Maret and Peters (1980) indicated that the plains topminnow was most commonly associated with brook stickleback in Nebraska, and Propst (1982) found the plains topminnow with brook stickleback in the South Platte River drainage.

When mean CPUE for all six native species were compared at all sites only two species, creek chub and brook stickleback had higher abundances at sites absent of plains topminnow. Our findings suggest that both of these native species may have been associated

with plains topminnow absence. Although creek chub was included in the fish assemblage PC2 which received a positive model averaged parameter estimate, it experienced an almost two fold increase in number sites present from the current-historical CDA (Appendix 1). This suggests that the generalist and possibly predatory creek chub may share similar habitats as the plains topminnow but its increase in the last 60 y seems to be associated with the decline of the plains topminnow. As for the brook stickleback, this species has a similar aggressive disposition, habitat requirement and diet to plains topminnow (Becker, 1983) and also increased slightly in number of sites present from the historical survey. If the brook stickleback has a competitive advantage over the plains topminnow, decreases in available habitat may result in decline of plains topminnow populations associated with presence of the brook stickleback. Our results did not imply that the introduced western mosquitofish presence was associated with plains topminnow absences as found by Lynch and Roh (1996) in western Nebraska. However, there was no historical collection of the western mosquitofish by Johnson (1942), and the western mosquitofish was present at 11.5% of the current (2003–2005) 52 sites.

Variation between the fish assemblage collection methods used in historic and current sampling may have influenced the analysis and interpretation of our study. Despite the limitation of our comparison to presence/absence data, our results were potentially influenced by differences in gear efficiency and selectivity. However, our observations of fish assemblages within the plains topminnow's historic range in Nebraska are consistent with patterns of assemblage homogenization (Rahel, 2002) and historical declines of native species in the Great Plains (Patton *et al.*, 1998) and offer insight into the long term changes of fish assemblages relative to an endemic Great Plains fish.

The deleterious affects of human induced disturbances (*i.e.*, dewatering, siltation, introduction of invasive species and impoundment) of Great Plains streams have long been observed (Jewel, 1927). The current risk to these unique systems is increasing with continued loss of native diversity and nonnative expansion, decreased system connectivity and potential for greater variability in environmental conditions (Cross and Moss, 1987; Rahel, 2002; Dodds *et al.*, 2004). Our analysis of historical and current surveys illustrated a fish assemblage shift that was influenced by introduced and generalist species expansion, which may have led to the decline of plains topminnow. Our results also suggest that habitat and fish assemblage are both important predictors of this rare Great Plains fish. The plains topminnow is a model endemic Great Plains species whose further management and conservation could benefit a variety of warmwater native fishes and aquatic fauna of the region (Rahel and Thel, 2004) in addition to further supporting the need for protection of greatly impacted Great Plains ecosystem. Conservation of the plains topminnow and other Great Plains fishes will require further identification of limiting factors affecting their distribution.

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APPENDIX 1.—Summary of fish species occurrences from historical and current sampling used in canonical discriminant analysis (CDA) and principal components analysis (PCA). Percentage of sites present in parentheses

Species	Historical 31 sites (1939–41)	Current revisited 31 sites (2003–05)	Current expanded 52 sites (2003–05)
Bigmouth shiner <i>Notropis dorsalis</i>	21 (67.7)	15 (48.4)	26 (50.0)
Black bullhead <i>Ameiurus melas</i> †	4 (12.9)	2 (6.5)	4 (7.7)
Black crappie <i>Poxomis nigromaculatus</i> ‡	1 (3.2)	1 (3.2)	1 (1.9)
Blacknose dace <i>Rhinichthys atratulus</i> ‡	1 (3.2)	1 (3.2)	1 (1.9)
Bluegill <i>Lepomis macrochirus</i>	2 (6.5)	7 (22.6)	14 (26.9)
Bluegill × green sunfish <i>Lepomis macrochirus</i> × <i>cyaneus</i> ‡	0 (0)	1 (3.2)	2 (3.8)
Bluntnose minnow <i>Pimephales notatus</i> ‡	0 (0)	1 (3.2)	2 (3.8)
Brassy minnow <i>Hybognathus hankinsoni</i>	16 (51.6)	9 (29.0)	11 (21.2)
Brook stickleback <i>Culaea inconstans</i>	3 (9.8)	4 (12.9)	8 (15.4)
Brown trout <i>Salmo trutta</i> ‡	0 (0)	1 (3.2)	3 (5.8)
Central stoneroller <i>Campostoma anomalum</i>	4 (12.9)	6 (19.4)	9 (17.3)
Channel catfish <i>Ictalurus punctatus</i>	4 (12.9)	3 (9.8)	11 (21.2)
Common carp <i>Cyprinus carpio</i>	2 (6.5)	11 (35.5)	21 (40.4)
Common shiner <i>Luxilus cornutus</i> ‡	1 (3.2)	0 (0)	0 (0)
Creek chub <i>Semotilus atromaculatus</i>	12 (38.7)	21 (67.7)	27 (51.9)
Emerald shiner <i>Notropis atherinoides</i>	3 (9.8)	1 (3.2)	6 (11.5)
Fathead minnow <i>Pimephales promelas</i>	21 (67.7)	18 (58.1)	34 (65.4)
Finescale dace <i>Phoxinus neogaeus</i> †	1 (3.2)	2 (6.5)	2 (3.8)
Flathead catfish <i>Pylodictis olivaris</i> ‡	0 (0)	0 (0)	3 (5.8)
Flathead chub <i>Platygobio gracilis</i>	7 (22.6)	5 (16.1)	9 (17.3)
Freshwater drum <i>Aplodinotus grunniens</i> ‡	0 (0)	0 (0)	2 (3.8)
Gizzard shad <i>Dorosoma cepedianum</i> ‡	0 (0)	0 (0)	3 (5.8)
Golden shiner <i>Notemigonus crysoleucas</i> ‡	1 (3.2)	1 (3.2)	1 (1.9)
Grass pickerel <i>Esox americanus</i> ‡	1 (3.2)	1 (3.2)	4 (7.7)
Green sunfish <i>Lepomis cyanellus</i>	8 (25.8)	18 (58.1)	28 (53.8)
Iowa darter <i>Etheostoma exile</i> †	3 (9.8)	2 (6.5)	6 (11.5)
Johnny darter <i>Etheostoma nigrum</i>	0 (0)	3 (9.8)	4 (7.7)
Largemouth bass <i>Micropterus salmoides</i>	3 (9.8)	13 (42.0)	19 (36.5)
Longnose dace <i>Rhinichthys cataractae</i>	9 (29.0)	14 (45.2)	18 (34.6)
Longnose gar <i>Lepisosteus osseus</i> ‡	0 (0)	0 (0)	1 (1.9)
Longnose sucker <i>Catostomus catostomus</i> †	1 (3.2)	4 (12.9)	4 (7.7)
Northern pike <i>Esox lucius</i> †	0 (0)	3 (9.8)	3 (5.8)
Northern redbelly dace × finescale dace <i>Phoxinus eos</i> × <i>neogaeus</i> ‡	1 (3.2)	0 (0)	0 (0)
Orangespotted sunfish <i>Lepomis humilis</i> ‡	1 (3.2)	1 (3.2)	1 (1.9)
Orangethroat darter <i>Etheostoma spectabile</i> †	1 (3.2)	5 (16.1)	5 (9.6)
Pearl dace <i>Margariscus margarita</i> †	2 (6.5)	2 (6.5)	2 (3.8)
Plains killifish <i>Fundulus zebrinus</i>	7 (22.6)	3 (9.8)	7 (13.5)
Plains minnow <i>Hybognathus placidus</i>	5 (16.1)	7 (22.6)	11 (21.2)
Plains topminnow <i>Fundulus sciadicus</i>^a	31 (100.0)	7 (22.6)	12 (23.1)
Pumpkinseed <i>Lepomis gibbosus</i> †	1 (3.2)	3 (9.8)	4 (7.7)
Quillback <i>Carpiodes cyprinus</i>	1 (3.2)	3 (9.8)	8 (15.4)
Rainbow trout <i>Oncorhynchus mykiss</i> †	1 (3.2)	3 (9.8)	5 (9.6)
Red shiner <i>Cyprinella lutrensis</i> ^b	15 (48.4)	15 (48.4)	30 (57.7)
River carpsucker <i>Carpiodes carpio</i>	1 (3.2)	2 (6.5)	10 (19.2)

APPENDIX 1.—Continued

Species	Historical 31 sites (1939–41)	Current revisited 31 sites (2003–05)	Current expanded 52 sites (2003–05)
River shiner <i>Notropis blennioides</i>	5 (16.1)	9 (29.0)	18 (34.6)
Rock bass <i>Ambloplites rupestris</i> †	0 (0)	1 (3.2)	1 (1.9)
Sand shiner <i>Notropis stramineus</i> ^b	18 (58.1)	22 (71.0)	40 (76.9)
Silver chub <i>Macrhybopsis storeriana</i> ‡	0 (0)	0 (0)	4 (7.7)
Shorthead redhorse <i>Moxostoma macrolepidotum</i>	2 (6.5)	11 (35.5)	16 (30.8)
Shortnose gar <i>Lepisosteus platostomus</i> ‡	0 (0)	1 (3.2)	2 (3.8)
Smallmouth bass <i>Micropterus dolomieu</i> ‡	0 (0)	1 (3.2)	2 (3.8)
Speckled chub <i>Extrarius aestivalis</i> ‡	1 (3.2)	0 (0)	1 (1.9)
Stonecat <i>Noturus flavus</i>	1 (3.2)	10 (32.3)	14 (26.9)
Suckermouth minnow <i>Phenacobius mirabilis</i> ‡	1 (3.2)	1 (3.2)	1 (1.9)
Tadpole madtom <i>Noturus gyrinus</i> †	3 (9.8)	0 (0)	0 (0)
Western mosquitofish <i>Gambusia affinis</i>	0 (0)	4 (12.9)	6 (11.5)
Western silvery minnow <i>Hybognathus argyritis</i>	1 (3.2)	3 (9.8)	3 (5.8)
White crappie <i>Pomoxis annularis</i> ‡	0 (0)	1 (3.2)	2 (3.8)
White sucker <i>Catostomus commersonii</i>	11 (35.5)	22 (71.0)	32 (60.4)
Yellow bullhead <i>Ameiurus natalis</i> †	2 (6.5)	4 (12.9)	4 (7.7)
Yellow perch <i>Perca flavescens</i> ‡	0 (0)	1 (3.2)	4 (7.7)

† Species included in the canonical discriminant analysis and not in the PCA

‡ Species not included in the CDA nor PCA due to occurrence below approximately 5% (2 sites) historically and 10% (5 sites) currently

^a Plains topminnow excluded from both CDA and PCA

^b Ubiquitous species of greatest abundance not included in the PCA to reduce model uncertainty