STRUCTURAL ORGANIZATION OF GREAT PLAINS STREAM FISH ASSEMBLAGES: IMPLICATIONS FOR SAMPLING AND CONSERVATION

by

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ABSTRACT

We investigated stream fish assemblages in Nebraska and Kansas to determine the effects of habitat and sampling methodologies on the community structure and abundance of prairie stream fishes of the Great Plains. We intensively sampled four mid-sized (9.9 m to 28.9 m wide), wadable streams to determine the sampling effort needed to assess the status and trends of fish communities. The number of reaches (<1 km) required to estimate segment (20-30 km) species richness decreased with increased reach length (10, 20, 40, or 60 mean stream width [MSW]) whereas total sampling effort decreased with more and shorter reaches. Only after all 10 reaches was total species richness obtained with 40 to 60 MSW. The number of reaches needed to detect 50% changes in fish relative abundance at 0.8 statistical power was 99 (range 7-630) and decreased with increased reach length. A greater number of reaches was needed to detect 90% of species richness and 25% changes in relative abundance when community similarity and habitat heterogeneity was lower. Our results suggest homogenous stream segments require more reaches to characterize fish community structure and monitor trends in fish abundance and a greater number of shorter reaches may be better than fewer longer (e.g. 40 or larger MSW) reaches. Effects of local environmental influences on the structure of fish assemblages were evaluated from 159 sites in two regions of the Great Plains with limited anthropogenic disturbance. These least disturbed regions offered an opportunity to evaluate the structure and natural variation of streams and fish assemblages within the Great Plains. We used canonical correspondence analyses to determine the influence of environmental conditions on species abundances, species occurrences, and assemblage characteristics. Analysis of regions separately indicated

that similar environmental factors structured streams and fish assemblages, despite differences in environmental conditions and species composition between regions. Variance in fish abundance and assemblage characteristic data from both regions was best explained by metrics of stream size and habitat features linked with stream size (width, depth, conductivity, instream cover). Our results provide a framework and reference for least disturbed conditions and assemblage structure in North American prairie streams.

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Chapter 1

Spatial scale of stream fish assemblage and abundance estimates: effects of sampling effort, community structure, and habitat heterogeneity

Abstract: The objective of this study was to determine the sampling effort required to detect changes in species richness and relative abundance within four Great Plains USA streams. The number of reaches (<1 km) required to estimate segment (20-30 km) species richness decreased with increased reach length (10, 20, 40, or 60 mean stream width [MSW]) whereas total sampling effort decreased with more and shorter reaches. Only after all 10 reaches was total species richness obtained with 40 to 60 MSW. The number of reaches needed to detect 50% changes in fish relative abundance at 0.8 statistical power was 99 (range 7-630) and decreased with increased reach length. A greater number of reaches was needed to detect 90% of species richness and 25% changes in relative abundance when community similarity (Jaccard's similarity) and habitat heterogeneity was lower. Our results suggest homogenous stream segments require more reaches to characterize fish community structure and monitor trends in fish abundance and a greater number of shorter reaches may be better than fewer longer (e.g. 40 or larger MSW) reaches.

Introduction

Stream ecosystems are often sampled to determine the structure of fish assemblages and status of individual species. These data are used to develop management and conservation decisions (i.e. conservation of areas of greatest diversity),

assess ecosystem health (i.e. index of biotic integrity, etc.) (Karr 1981; Fausch et al. 1990), determine fish distributions (Rahel and Hubert 1991), and understand fish community structure dynamics (Olden et al. 2006). The accuracy and quality of fish assemblage data is therefore critical to the validity of these decisions. However, it is often impossible to enumerate all species of a community (Krebs 1998), which creates difficulties in determining the amount of sampling effort required to assess a community (e.g. stream segment). Furthermore, the number of species collected increases as stream area or length and the number of samples increases, due to the species-area relationship (Angermeier and Schlosser 1989; Lyons 1992; Peterson and Rabeni 1995). Species are also not evenly distributed due to heterogeneity of habitat within the stream (Gorman and Karr 1978; Angermeier and Smogor 1995) and variability in spatial distributions of rare species (Lyons 1992; Paller 1995) also affects species detection.

The relationship of fish assemblages with in a reach to greater spatial scales (e.g. segments, entire streams, catchments, etc.) is not well understood. Fausch et al. (2002) proposed that the fundamental problem with the current conservation and management of stream fish is the lack of scientific research relevant over large spatial and temporal scales, and argued the current spatial gap of knowledge exists between reaches (i.e. < 1 000 m) and segments (i.e. 1 to 100 km). Furthermore, it has become increasingly easy to utilize large spatial scale data with geographic information systems (e.g. landcover) in conjunction with fish assemblage data collected at relatively small spatial scales. It is therefore important to understand the relationship between fish assemblage data collected at sampled reaches to that of entire stream segments.

Many studies have attempted to determine the sampling effort (stream length; based on mean stream width [MSW]) required in wadable streams reaches to collect a high percentage (90-100%) of the species present (Lyons 1992; Angermeier and Smogor 1995; Paller 1995; Patton et al. 2000; Dauwalter and Pert 2003). The number MSWs needed to reach asymptotic species richness ranged from as few as five in Wisconsin streams (Lyons 1992) to as much as 158 in South Carolina coastal plain streams (Paller 1995). Therefore considerable discrepancy exists on the sampling effort required in streams at individual reaches. Results from these studies suggest that the sampling effort required to reach asymptotic species richness in streams may vary among regions and stream size. However, the need to develop standardized methods for sampling fish assemblages have resulted in established protocols that recommend a sampling length of 40 MSW (Moulton et al. 2002; Peck et al. 2002). These programs are often focused on determining conditions and monitoring trends over time so fish community sampling is typically conducted at a single reach (Moulton et al. 2002). It is therefore necessary to evaluate the natural variability in fish community characteristics with multiple reaches sampled within stream segments using these standardized methods.

Most studies of sampling effort do not extensively sample multiple sites at each stream. Matthews (1990) found that increasing sampling sites more adequately represented the fish assemblage in the South Fork of Roanoke River in Virginia. In addition, Peterson and Rabeni (1995) determined that at least 24 samples were needed to obtain species richness values with a level of precision of 10% in Missouri streams and that spatial variation exceeded temporal variation. This suggests that more samples may be needed to assess the fish community structure in many streams due to variability in

habitat availability. For rare fishes high variability in catch rates leads to low precision and thus an unobtainable number of samples needed to detect trends (Paukert 2004). Even monitoring trends in common species may require more sampling than is feasible (Quist et al. 2006). Therefore the evaluation of the effort needed in wadable streams to assess community structure and monitor fish populations is important to both local and regional ecosystem management.

In this study we investigated the sampling effort needed to quantify fish assemblages in mid-sized (10-29 m mean width) wadable Great Plains streams. Our objectives were to (i) examine the species-area relationship within a stream segment (ii) determine the increment of sampling length (10, 20, 40, and 60 MSWs) and number of reaches needed to collect precise (75, 90, and 100%) estimates of segment species richness, (iii) determine the effect of sampling length on the number of reaches needed to detect 25, 50, and 75% changes in relative abundance of fish species, and (iv) identify the physical parameters and/or community structure that best explain the variation in sampling effort among streams.

Materials and Methods

Site description

Four streams in Nebraska and Kansas were sampled from 15 May to 30 June 2006, with individual streams sampled in five days or less to minimize temporal variation among samples within streams. Streams were selected to represent a variety of stream characteristics found in the Great Plains and were located in three of the 12 major basins (Kansas, Niobrara, and Platte) of the Missouri River. Streams were also selected on the

basis of access so that intensive longitudinal sampling (i.e. 20-30 km) could be conducted in a localized area to determine species present for the stream segment. The Niobrara River, a National Scenic River, is located in north central Nebraska and is a tributary to the Missouri River. Blue Creek is located in western Nebraska and is a tributary to the North Platte River. The North Loup River is a tributary to the Loup River which drains into the Platte River. The North Loup is located in central Nebraska Sandhills region. The West Branch Mill Creek is located in the Flint Hills region of eastern Kansas and is a tributary to the Kansas River. Table 1 provides a summary of the physical characteristics of each stream.

Data collection

Ten reaches were systematically sampled on each stream with a minimum of one km between reaches. Each reach was sampled at sampling lengths of 10, 20, 40 and 60 times the MSW using a towed, pulsed DC, electrofishing unit with two anode poles. The four MSW sampling lengths were adjacent to each other (i.e. the 10 MSW sampling length was included in the 20 MSW sampling length, etc.). Electrofishing was conducted by two people with anodes and two netters in a single upstream run in a zig-zag pattern with an emphasis to sample all available habitats (Lazorchak et al. 1998). Fish from each reach length were held in fish cages for identification after electrofishing for that entire reach (all MSW) was complete. Species were identified, enumerated, and released in the field. Unidentifiable and voucher specimens were preserved in 10% formalin and identified to species and enumerated in the laboratory.

Physical habitat was surveyed at six reaches on each stream using procedures adapted from the Environmental Protection Agency's (EPA) protocol for sampling wadable streams (Lazorchak et al. 1998). The only modification to these techniques was sampling reaches were characterized on six transects (reduced from 11) with five sections between transects. Wetted width (m) was measured at each transect and depth (m) was measured at three equally-spaced points along the transect (i.e. one-quarter, one-half, and three-quarters the wetted width distance) and divided by four to calculate transect mean depth (Arend 1999). Stream width and depth were calculated from the mean of transect depths and widths. Instream fish cover categories (filamentous algae, aquatic macrophytes, large woody debris, small woody debris, overhanging vegetation, undercut banks, and boulders) were estimated individually using five cover classes; "absent" (0%), "sparse" (< 10%), "moderate" (10 to 40%), "heavy" (40 to 75%), and "very heavy" (>75%) at each transect (Lazorchak et al. 1998). The midpoint of each percentage classes was used to determine mean percent cover for each reach. Coefficients of variation (CV) were calculated for each habitat parameter for each reach.

Species-area relationship

The cumulative number of species collected and sampled area were logarithmic (base 10) transformed and plotted for each stream and reach length (MSW) to determine the rate (z) at which species were accumulated (Ricklefs 2000). Linear regression was used to test significance of relationships and estimate the slope (z). Comparisons of z among species-area relationships for streams and MSW were made with analysis of covariance (ANCOVA) using area as a covariate. When ANCOVA was significant,

comparisons of slopes were conducted using pairwise comparisons with significance levels adjusted with a Bonferroni correction ($\alpha = 0.05/6$ or $\alpha = 0.008$).

Sampling effort simulations

To determine the number of reaches needed to detect 75, 90, and 100% of the collected species in all sampling, a Monte Carlo simulation that selected all possible combinations of reaches with replacement was used. Total segment species richness was calculated as all species encountered with all ten reaches for each stream. The probability of sampling precise estimates of the segment species richness were calculated by enumerating the number of 1 000 simulations that species richness met or exceeded the 75, 90, and 100% of the segment species richness. Simulation results were plotted for each stream and MSW. All simulations were run using all species collected and then again for common species (i.e., after removing species comprising <1% of the cumulative catch for each stream). Total sampling effort (cumulative number of MSWs) was determined for each combination of reach length and number of reaches required from simulation results.

Sample size estimation

The number of reaches needed to detect changes in catch per unit effort (CPUE, number of fish per 100 m of electrofishing) at various statistical power levels was determined using Simple Interactive Statistical Analysis (SISA, Uitenbroek 1997), which has been commonly used to determine needed sample sizes in fish studies (Allen et al. 1999; Tate et al. 2003; Paukert 2004). We estimated the number of reaches needed to

detect 25, 50 and 75% changes in CPUE at four levels of statistical power (i.e., 0.60, 0.70, 0.80, and 0.90) for each species that accounted for greater than 1% of the cumulative catch for each stream. The mean number of reaches needed was estimated for each stream and MSW sampling length. A significance level α equal to 0.05 was used for all sample size estimates. An analysis of variance (ANOVA) was used to determine if the mean number of sampled reaches required differed at 0.8 level of statistical power among streams and MSW separately. Comparisons among the number required reaches were conducted using pairwise comparisons with significance levels adjusted with a Bonferroni correction ($\alpha = 0.05/6$ or $\alpha = 0.008$).

Stream community structure

We evaluated the effect of longitudinal variability in fish communities on the sampling effort needed to estimate species richness and the ability detect changes in relative abundance using Jaccard's similarity coefficient (Krebs 1998). Linear regression was used to assess the relationship between the mean Jaccard's similarity coefficient and the number reaches between samples to assess the overall fish community variability for each stream. We also evaluated the ability of distance between reaches to predict species similarity with a Mantel test (Mantel 1967). We compared matrices of Jaccard's similarity and distance between sites with a Mantel test to determine the statistical significances and strength of the association (Mantel 1967). A significance level α equal to 0.05 was used for both the ANCOVA and Mantel test.

Habitat associations

A stepwise discriminant analysis was used to reduce the number of habitat variables and determine the subset of uncorrelated variables that best discriminated among streams from the set of nine habitat variable means CVs (see *Data Collection* above). A canonical discriminant analysis (CDA) was conducted with the new subset of significant habitat variables. Simple linear regression was used to asses the relationship between the number of reaches needed to detect 75, 90, and 100% of the total and common species richness and the mean canonical coefficients for all streams at 40 MSW.

Results

Total species richness ranged from 13 to 32 in the 20.1-28.4 km segments for the four streams sampled (Table 1). The number of individual fish sampled in each stream segment ranged from 2 660 in Blue Creek to 27 894 in the Niobrara River (Table 2). The number of rare species ranged from eight in the North Loup River to 17 in West Branch Mill Creek. These species cumulatively represented 1.8 to 3.7% of the total fish collected for each stream. The proportion of rare species in the North Loup River (42.1%) and West Branch Mill Creek (53.1%) was less than the Niobrara River (66.7%) and Blue Creek (69.2%) (Table 2). Individual species catch per unit effort (CPUE, fish per 100 m sampled) ranged from 0 to 81.1 ($\bar{x}$ = 4.6) for Blue Creek, from 0 to 259.2 ($\bar{x}$ = 15.5) for the Niobrara River, 0 to 83.6 ($\bar{x}$ = 5.6) for the North Loup River, and 0 to 536.0 ($\bar{x}$ = 14.0) for the West Branch Mill Creek across all sites. The Niobrara River was the shallowest (0.16 m) and widest (28.9 m) stream sampled with the least variability in width. Blue Creek was the narrowest (9.9 m) and deepest (0.45 m) stream with the least

variability in depth while the North Loup and West Branch Mill Creek were similar in mean width (11.0 m and 11.4 m). The West Branch Mill Creek exhibited the most variability in mean width and mean depth (Table 1). Mean percent cover of instream habitat variables and CVs varied considerably among streams (Table 1). For example, the Niobrara River and West Branch Mill Creek had greater CVs for small woody debris while Blue Creek and the North Loup River had higher CVs for undercut banks.

Species-area relationship

The rate of species accumulation (z) did not differ (ANCOVA; Ps > 0.05) among reach lengths for all streams so linear regression was used to estimate the slope of the relationship between species richness and area sampled for all reach lengths combined. Species richness was strongly correlated (r^2 range 0.68 – 0.93) with area sampled across all streams (Fig. 1). Slopes of species-area relationships varied among streams (ANCOVA; F=26.28; DF=3, 152; P<0.0001) and did not differ between Blue Creek and the Niobrara River (F=0.05; DF=1,2; P=0.817) nor West Branch Mill Creek and the North Loup River (F=4.73; DF=1,2; P=0.031) after a Bonferroni correction for multiple comparisons. Rates of species accumulation were greater for Blue Creek and Niobrara River (z's = 0.22) than West Branch Mill Creek (z = 0.12) and North Loup River (z = 0.07).

Sample effort simulations

The total sampling effort (number of reaches x MSW) required to obtain 75% of the species richness varied by stream and MSW sampled. Total sampling effort was

lowest in 10 MSW reaches (which required 6-8 sampled reaches) and ranged from 60 to 80 total MSW sampled (Fig. 2A). However, 75% of all species were only obtained within three of the four streams (i.e. 75% of all species were not collected with ten 10 MSW reaches in the Niobrara River). Increasing MSW typically decreased the number of reaches needed to collect 75% of the species. However, total effort increased. For example, only three sites at 40 MSW were needed to collect 75% of all species in the North Loup River, but this required a total effort of 120 (3 reaches x 40 MSW). To achieve the same number of species, four reaches at 20 MSW were needed, which is only 80 units of effort which was equal to eight reaches at 10 MSWs (Fig 2A). There was a greater total effort needed to collect 90% of the species (100-540 units of effort; Fig. 2A), but effort was still lowest with a higher number of shorter 10 MSW reaches. To collect 100% of the species, three streams needed all 10 sites at 60 MSW (Fig. 2A), the maximum sampling in our study. The West Branch of Mill Creek collected all species at 10 sites at 40 MSW. No combination of reaches at 10 or 20 MSW collected 100% of the species (Fig. 2A).

The number of reaches required to obtain common species decreased substantially for all streams (e.g., all common species were collected with just one 10 MSW reach in Blue Creek) (Fig. 2B). The number of reaches required to obtain 75% of the segment common species richness ranged from one to three 10 MSW reaches and total effort was less at 10 MSW for all streams (i.e. 10-30 units of effort). One to five 10 MSW reaches were required to obtain 90% of the common species for all streams (Fig. 2B). Total effort required to obtain 100% of common species ranged from 10-100 units of effort across all streams and reach lengths. Despite a greater number reaches, total effort was

minimized with shorter sampling lengths (i.e. 10 MSWs). Overall less effort was required to obtain common species than all species within all streams.

Sample size estimation

The mean number of reaches needed to detect a 25, 50 and 75% changes in CPUE of common species ranged from 63 to 994, 17 to 249, and 8 to 111 reaches, respectively across all levels of power for all streams (Fig. 3). The number of reaches needed varied among streams ($F=5.20$; $DF=3, 144$; $P=0.0019$) and MSW lengths ($F=4.10$; $DF=3, 144$; $P=0.0079$) for 25% changes in CPUE, among streams ($F=5.07$; $DF=3, 144$; $P=0.0023$) and MSW lengths ($F=4.10$; $DF=3, 144$; $P=0.0079$) for 50% changes in CPUE, and among streams ($F=4.74$; $DF=3, 144$; $P=0.0035$) and MSW lengths ($F=3.50$; $DF=3, 144$; $P=0.0171$) for 75% changes in CPUE. The number of reaches at 0.8 statistical power needed to detect 25, 50, and 75% changes in CPUE for the North Loup River, West Branch Mill Creek, and Blue Creek did not differ ($Ps > 0.008$) and were greater than that required to detect changes for the Niobrara River (Fig. 3). The number reaches needed to detect changes decreased as MSW increased. However, the number reaches required to detect 25 and 50% changes in CPUE did not differ ($P > 0.008$) between 20, 40, and 60 MSW reach lengths and were less than the number of reaches required for 10 MSW reach lengths (Fig. 3). Overall the numbers of reaches needed to detect changes was high. Even for common species at 60 MSW the mean number of reaches needed to detect a 75% change at 0.8 statistical power was still over 25 reaches in stream segments of 20-28 km.

Stream community structure

Similarity of the communities decreased as the number of reaches between sampling reaches increased for all streams (Fig. 4). The relationship between Jaccard's similarity coefficients and the number reaches between sampled reaches was more variable in Blue Creek ($r^2 = 0.10$) and Niobrara River ($r^2 = 0.11$) than in the North Loup River ($r^2 = 0.31$) and West Branch Mill Creek ($r^2 = 0.33$). There was no difference in the slopes among streams ($F = 1.72$; $DF = 3, 172$; $P = 0.16$), but slopes tended to be steeper for the North Loup River (-0.029) and West Branch Mill Creek (-0.024) than slopes of Blue Creek (-0.013) and the Niobrara River (-0.014). Results of the Mantel test were similar to regression results. A negative correlation of species similarity and distance between reaches was observed in all streams except the Niobrara river (Mantel test, $r = -0.26$, $P = 0.059$). A weak correlation existed in Blue Creek (Mantel test, $r = -0.37$, $P = 0.008$), while North Loup River (Mantel test, $r = -0.55$, $P = 0.0003$) and West Branch Mill Creek (Mantel test, $r = -0.66$, $P < 0.00001$) had a greater decrease in community similarity as distance between reaches increased. Greater community similarity existed among reaches of Blue Creek and the Niobrara River while the North Loup River and West Branch Mill Creek exhibited less similarity among all sampling reaches.

Habitat associations

The stepwise discriminant analysis of means and CVs of habitat parameters (Table 1) determined that mean width, mean depth, and percent cover of filamentous algae, depth CV, percent cover of filamentous algae CV, and percent cover of brushy debris CV discriminated among streams ($Ps < 0.05$). The first three canonical functions were significant ($Ps < 0.05$) (Fig. 5 and 6), with the first canonical axis accounting for

54.4% of total variability and was a gradient of percent cover of filamentous algae, CV of small woody debris percent cover, and CV of depth. Mean width and mean depth had the highest loadings on the second canonical axis and explained 34.4% of the total variability. This axis separated the Niobrara River, which was at least twice as wide and half as deep as other streams (Table 1). The CV of filamentous algae percent cover was the only variable with a high loading on the third axis, which explained 11.2% of the total variation, and separated the North Loup River and West Branch Mill Creek from streams with less variability in the percent cover of filamentous algae. The relationship of the mean standardized canonical coefficients of the first and second canonical axes and the number of sites needed to obtain 75, 90, and 100% of total (Fig. 5) and common species (Fig. 6) richness for each stream were not significant (Ps >0.10). The relationship of the mean canonical coefficients of the third canonical axis and number of reaches need to detect 75 and 90% of total species richness were significant (Ps = 0.09 and 0.04) and negative (βs = -1.43 and -0.85) indicating that as CV of filamentous algae increased the number of reaches needed to detect 75 and 90% of segment species richness decreased. After removing rare species the only the significant relationship was between the mean standardized canonical coefficients of the third canonical axis and the number of reaches needed to detect 75% of common species (Fig. 6; P = 0.04). This relationship was positive (β = 1.83) which indicates that by removing rare species less samples were needed with increased variation of filamentous algae cover.

Discussion

The species-area relationship has been well established over the last century (Arrhenius 1921; Gleason 1922; Williams 1964; Rosenzweig 1995) and has been the foundation of most sampling effort studies focused on determining the balance between inaccurately characterizing communities by under-sampling and cost prohibitive over-sampling. Although increasing sampling area and encountering more species may be the result of more effectively sampling the available species (Conner and McCoy 1979), increasing sampling area may also increase habitat diversity sampled, which can support a greater number of species (Williams 1964). Since these are not mutually exclusive, the indirect relationship between sample length (e.g. sample area) and stream characteristics (e.g. habitat) has been well studied to understand the effort needed to characterize communities. (e.g. Lyons 1992; Dauwalter and Pert 2003). However, these studies have only focused on individual sampling sites and the habitat characteristics within sites. Research focused on understanding the relationship between environmental factors at multiple scales (e.g. instream, riparian, watershed, basin, etc.) and fish communities are often conducted with individual reaches that are then used to characterize stream segments (Gido et al. 2006) or entire catchments (Diana et al. 2006; Heitke et al. 2006; Gido et al. 2006). This increases the need for accurate estimates of community attributes and to understand the sampling effort needed to characterize streams segments.

The species-area relationships of the four Great Plains streams indicate that the length of sampling reach had no effect on the rate of species accumulation for all streams. However, the rate of species accumulation differed among streams with Blue Creek and the Niobrara River having greater rates of increase than the North Loup River and West

Branch Mill Creek. This result was consistent with the effort required to obtain various levels of segment richness among streams. The Niobrara River and Blue Creek required consistently less effort than the North Loup River and West Branch Mill Creek.

Our results suggest that moderate levels (i.e. 75%) of segment species richness are only obtainable after sampling three to five reaches at the widely accepted protocol of 40 MSW sample lengths and more accurate levels (i.e. 90%) of segment species richness were only obtained after an extensive number of reaches (i.e. six to ten reaches at 40 MSW reach lengths) in relatively small stream segments of 20-28 km. However, the total effort required (cumulative MSWs) was consistently greater for longer reach lengths, suggesting an increased number of shorter reaches would characterize stream segments with less total effort. The effort required to obtain segment species richness was consistently higher in the Niobrara River and Blue Creek and consistently lower in the West Branch Mill Creek and the North Loup River, suggesting characteristics of stream (e.g. habitat complexity) affected sampling variability. Our results from Great Plains streams suggest that more homogeneous streams require more sampling effort to characterize fish community structure which is consistent with findings from Illinois and Virginia streams (Angermeier and Smogor 1995). The failure of all four stream simulations to reach asymptotic levels of 100% of species suggests that there are discontinuous distributions or low densities for some species (Angermeier and Smogor 1995), which was minimized after the removal of rare species. A species probability of occurrence in the available habitat is strongly correlated to the species relative abundance (Angermeier and Smogor 1995), suggesting that rare species are less likely to occupy all available habitats which decreases the chance being collected in a single sample.

After removal of rare species, the numbers of reaches required to obtain precise estimates of common species richness for the entire segment were consistently lower for the Niobrara River and Blue Creek and higher for the North Loup River and West Branch Mill Creek. The increased numbers of reaches needed for the Niobrara River and Blue Creek when all species were included was likely caused by rare species which were not sampled at a majority of reaches. Although total segment species richness may not be a feasible objective, the number of reaches needed decreased considerably when only common species were considered and would be much more reasonable to most managers and researchers of similar systems, if common species were the focus of the study objectives. Our findings suggest that standardized protocols may not accurately characterize fish assemblages and may fail to catch species of low relative abundance within an individual stream segment.

Patterns of fish assemblage shifts in longitudinal gradients have been well studied (e.g. Rahel and Hubert 1991; Edds 1993), suggesting fish assemblage similarity increases with decreased distances between samples and is related to the physical habitat that changes as streams increase in size (Platania 1991; Rahel and Hubert 1991). The Niobrara River and Blue Creek exhibited shallower and more variable trends in similarity across all reaches than the North Loup River and West Branch Mill Creek. Greater slopes in these relationships would indicate an increased rate of species turnover (or decreased similarity) across all sampling reaches. Greater variability in community similarity would indicate that reaches that should be more similar (e.g. closer sites) are not, and less similar reaches (e.g. farther apart) are more similar. This is likely due to species that were not consistently sampled at all reaches. Our results suggest that stream

segments which have less similar communities among all reaches (i.e. North Loup River and West Branch Mill Creek) needed less sampling events to reach the same proportion of species richness than streams that have more similar communities (i.e. Blue Creek and Niobrara River). Therefore, the random selection of reaches with less similar fish assemblages would represent a greater proportion of the streams total species richness for the entire segment (i.e. autosimilarity, Cao et al. 2002). Streams with higher species turnover should need fewer systematically sampled reaches to characterize a greater proportion of species in the segment than streams with more similar assemblages. Overall, our results of community similarity within streams suggest that given a fixed number of samples to characterize a stream segment, a greater distance between sampled reaches would represent a greater proportion of the segment community.

The increased number of reaches needed to obtain a given proportion of segment species richness in streams with higher variation in percent cover of filamentous algae is likely due to rare species discontinuous distributions and habitat selectivity (Angermeier and Smogor 1995). Since habitat complexity was lower in the streams that needed a greater number of reaches to obtain total segment species richness, rare species may be selective to specific habitat types thus resulting in low overall relative abundances and infrequent collection throughout sampling segments. Stream segments with greater habitat homogeneity present fewer opportunities for rare species to be collected, which in turn may increase the number of samples needed to collect all species present. Additionally, Angermeier and Schlosser (1989) found habitat complexity was correlated to species richness in Panama streams and not with streams in Minnesota and Illinois, but suggested the stability of streams in Panama may facilitate this relationship. Larger

streams with greater habitat heterogeneity and less temporal variation typically have stable fish communities (Schlosser 1987). The four mid-sized Great Plains streams sampled may be better suited for evaluating assemblage variability as related to habitat within stream segments.

A large number of samples would be needed to detect trends in the relative abundance of stream fishes in the four Great Plains streams sampled, which is similar to other studies in rivers and streams (Peterson and Rabeni 1995; Paukert 2004; Quist 2006). Unlike other studies of sample size estimation, we evaluated the effect of reach length and determined increasing reach length decreased the number of reaches needed to detect trends in relative abundance of fish species. However increasing reach length beyond the 20 MSW lengths did not decrease the number of reaches needed. The same pattern in the number of reaches needed to obtain varying levels of total segment species richness was observed in the estimated number of reaches needed to detect trends. Streams with greater habitat complexity and less overall community similarity required more reaches to detect trends in CPUE than streams with less habitat complexity and greater similarity of species composition. Since only common species were used in the sample size estimates, our results indicate that greater variability in CPUE may be the influenced by increased habitat complexity.

Although a number of studies have described the length of reach needed (number of MSWs) to assess stream fish communities, our results indicate that multiple reaches within a segment are needed to characterize stream fish communities. Additionally, the number of reaches required in Great Plains streams, with a towed barge electrofisher, may not be cost effective due to discontinuous species distribution or low species

abundance (40% greater of species represented less than 4% of the total number of individuals collected in each stream). However, 75% of all fish species were obtained with the least amount of total effort with five to ten 20 MSW reaches within all four streams. Monitoring changes in common species relative abundance in Great Plains streams may require an increase in sampling effort. Most importantly, the number of reaches needed to detect presences and monitor changes in relative abundances of stream fishes will ultimately depend on individual study objectives and the scale to which stream communities are to be characterized. Our results suggest that the sampling effort needed to characterize streams could be decreased by evaluating instream habitat characteristics. Furthermore, previous knowledge of stream community structure and species abundances may be helpful in the establishment of required sampling effort and to determine areas needing additional sampling effort to characterize a greater proportion of species present.

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Table 1. Species richness collected, total stream segment length, mean distance between reaches sampled (standard error in parenthesizes), mean stream width, mean stream depth, mean percent coverage of instream habitat measurements (CV in parenthesizes) from habitat surveys of four streams in Nebraska and Kansas sampled in summer 2006.

	Blue Creek		Niobrara River		North Loup River		West Branch Mill Creek	
Species richness	13		21		19		32	
Segment length (m)	28,416		20,125		25,232		25,719	
Mean dist. between sites (m)	3,157 (330)		2,236 (202)		2,804 (409)		2,858 (436)	
Mean width (m)	9.94	(27.1)	28.89	(19.2)	10.95	(23.8)	11.39	(43.7)
Mean depth (m)	0.45	(13.4)	0.16	(26.7)	0.33	(17.4)	0.36	(69.6)
Filamentous algae	6.39	(26.3)	2.15	(53.2)	0.54	(152.8)	21.54	(100.0)
Macrophytes	30.59	(56.9)	0.59	(130.4)	13.06	(74.0)	0.00	(0.0)
Large woody debris	0.00	(0.0)	0.03	(144.1)	0.00	(0.0)	0.33	(64.0)
Small woody debris	0.28	(64.7)	0.08	(203.7)	0.00	(0.0)	0.32	(200.3)
Overhanging vegetation	13.12	(76.3)	1.10	(48.6)	1.63	(46.3)	0.10	(122.5)
Undercut bank	0.06	(185.4)	0.03	(39.8)	0.06	(100.3)	0.12	(81.6)
Boulder	0.00	(0.0)	0.60	(82.0)	1.12	(128.8)	0.18	(124.1)

Table 2. Summary of species occurrences and percent abundance from four mid-sized streams of Nebraska and Kansas sampled using a towed electrofishing unit in summer 2006.

Common name	Scientific name	Blue Creek N=2,660	Niobrara river N=27,894	North Loup River N=5,273	West Branch Mill Creek N=23,600
Bigmouth shiner	*Notropis dorsalis*	0.226	25.790	7.225	
Black bullhead	*Ameiurus melas*			0.057	0.047
Black crappie	*Poxomis nigromaculatus*				
Blackside darter	*Percina maculata*				0.013
Bluegill	*Lepomis macrochirus*		0.004		0.318
Bluntnose minnow	*Pimephales notatus*				11.051
Brassy minnow	*Hybognathus hankinsoni*		0.090	5.385	
Brook stickleback	*Culaea inconstans*	0.301	0.004		
Brown trout	*Salmo trutta*	0.376			
Bullhead minnow	*Pimephales vigilax*				0.034
Central stoneroller	*Campostoma anomalum*		0.183		36.780
Channel catfish	*Ictalurus punctatus*			0.114	
Common carp	*Cyprinus carpio*		0.018	0.133	0.992
Common shiner	*Luxilus cornutus*				2.674
Creek chub	*Semotilus atromaculatus*	72.782	2.416	2.333	1.326
Fathead minnow	*Pimephales promelas*	0.226	1.391	0.778	0.068
Flathead catfish	*Pylodictis olivaris*				0.140
Flathead chub	*Platygobio gracilis*			5.765	
Golden redhorse	*Moxostoma erythrurum*				1.114
Green sunfish	*Lepomis cyanellus*	0.564		0.303	2.161
Johnny darter	*Etheostoma nigrum*				0.585
Largemouth bass	*Micropterus salmoides*		0.004		0.288
Logperch	*Percina caprodes*				0.216
Longear sunfish	*Lepomis megalotis*				3.106
Longnose dace	*Rhinichthys cataractae*	1.466	0.549	36.450	
Longnose gar	*Lepisosteus osseus*				0.068
Northern pike	*Esox lucius*		0.007	0.019	
Northern redbelly dace	*Phoxinus eos*			4.077	
Orangespotted sunfish	*Lepomis humilis*				0.008
Orangethroat darter	*Etheostoma spectabile*	11.767			5.174
Plains killifish	*Fundulus zebrinus*		0.004	0.645	
Plains topminnow	*Fundulus sciadicus*	0.902	0.484		
Rainbow trout	*Oncorhynchus mykiss*	0.038	0.004		
Red shiner	*Cyprinella lutrensis*		11.325	3.717	6.475
Redfin shiner	*Lythrurus umbratilis*				0.106
River carpsucker	*Carpiodes carpio*			0.910	0.030
Rock bass	*Ambloplites rupestris*		0.054		
Rosyface shiner	*Notropis rubellus*				9.737
Sand shiner	*Notropis stramineus*	0.414	47.580	15.172	1.479
Shorthead redhorse	*Moxostoma macrolepidotum*		3.090	1.346	
Slender madtom	*Noturus exilis*				2.415
Stonecat	*Noturus flavus*	0.639	0.312	5.860	0.004
Suckermouth minnow	*Phenacobius mirabilis*				6.869
Southern redbelly dace	*Phoxinus erythrogaster*				1.742
Topeka shiner	*Notropis topeka*				0.017
White sucker	*Catostomus commersonii*	10.301	6.586	9.710	4.742
Yellow bullhead	*Ameiurus natalis*				0.225
Yellow perch	*Perca flavescens*		0.108		

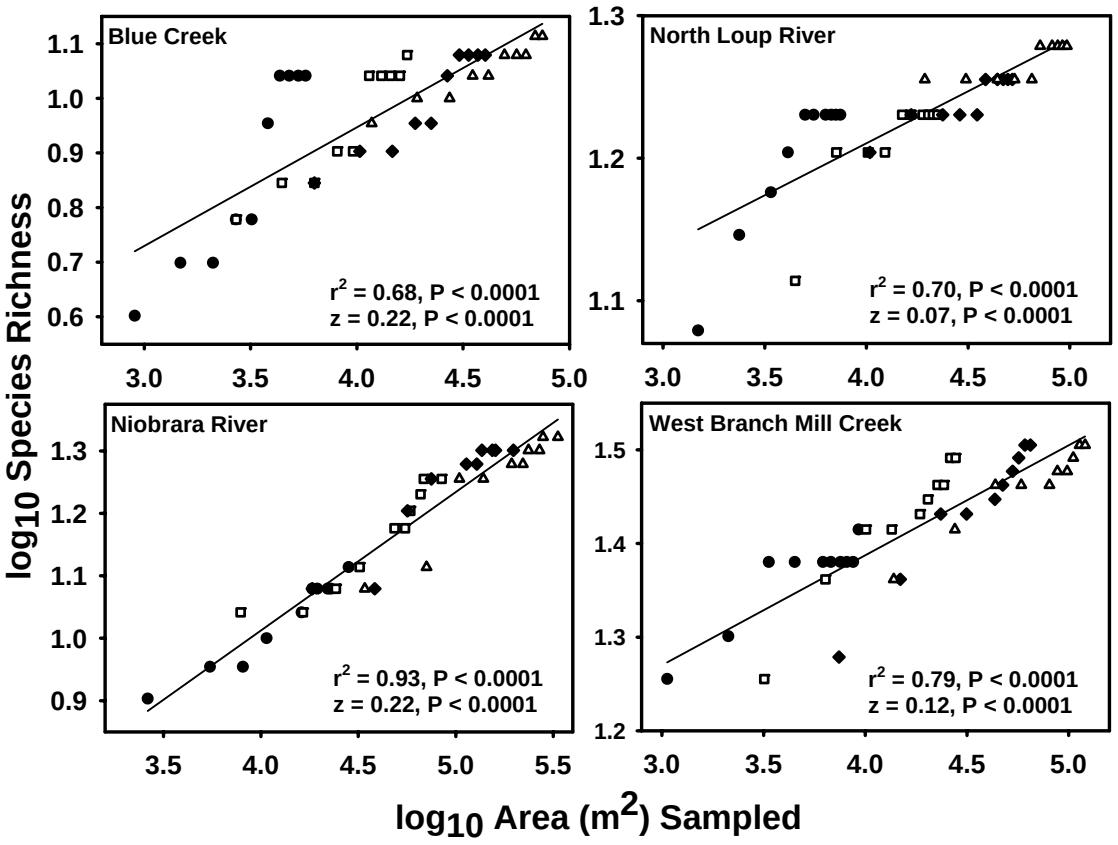

Fig. 1. Species-area relationships for four mid-sized streams in Nebraska and Kansas sampled in May-June 2006. Symbols represent length of sampled reaches for 10 (filled circles), 20 (open squares), 40 (filled diamonds), and 60 (open triangles) mean stream widths. Lines represent linear regressions and *z* is the slope of the line. See text for discussion of analyses of covariance.

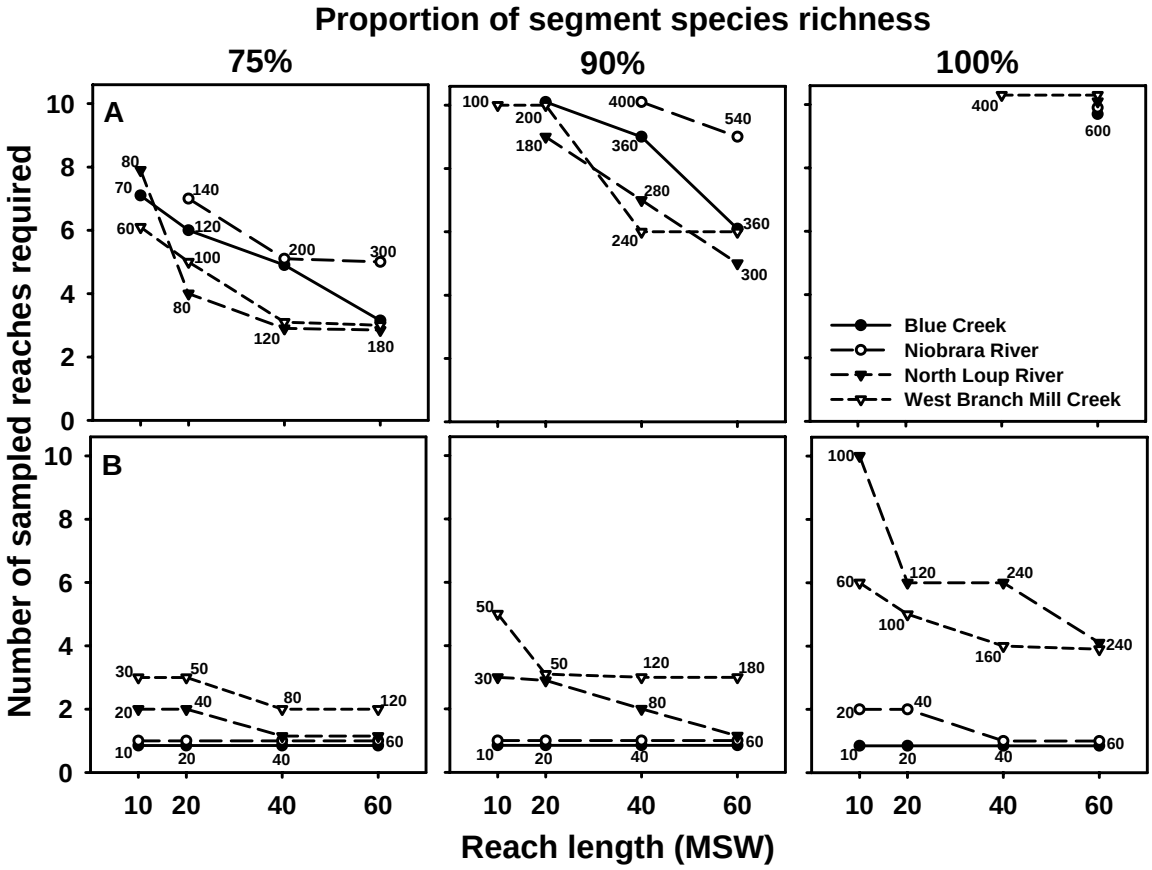

Fig. 2. Number of sampled reaches required to detect 75, 90 and 100% of segment species richness for four sampling lengths (10, 20, 40 and 60 times the mean stream width, MSW) of four wadable streams in Nebraska and Kansas based on 1,000 Monte Carlo simulations. Values are based on all species (panel A) and after the removal of rare species (panel B) (species accounting for < 1% of the cumulative catch for a stream). Total effort (number of reaches x MSW) indicated adjacent to symbols. Missing values indicate the number of reaches required was greater than our maximum number of reaches. The numbers of reaches required represent whole numbers and symbols are offset to improve clarity.

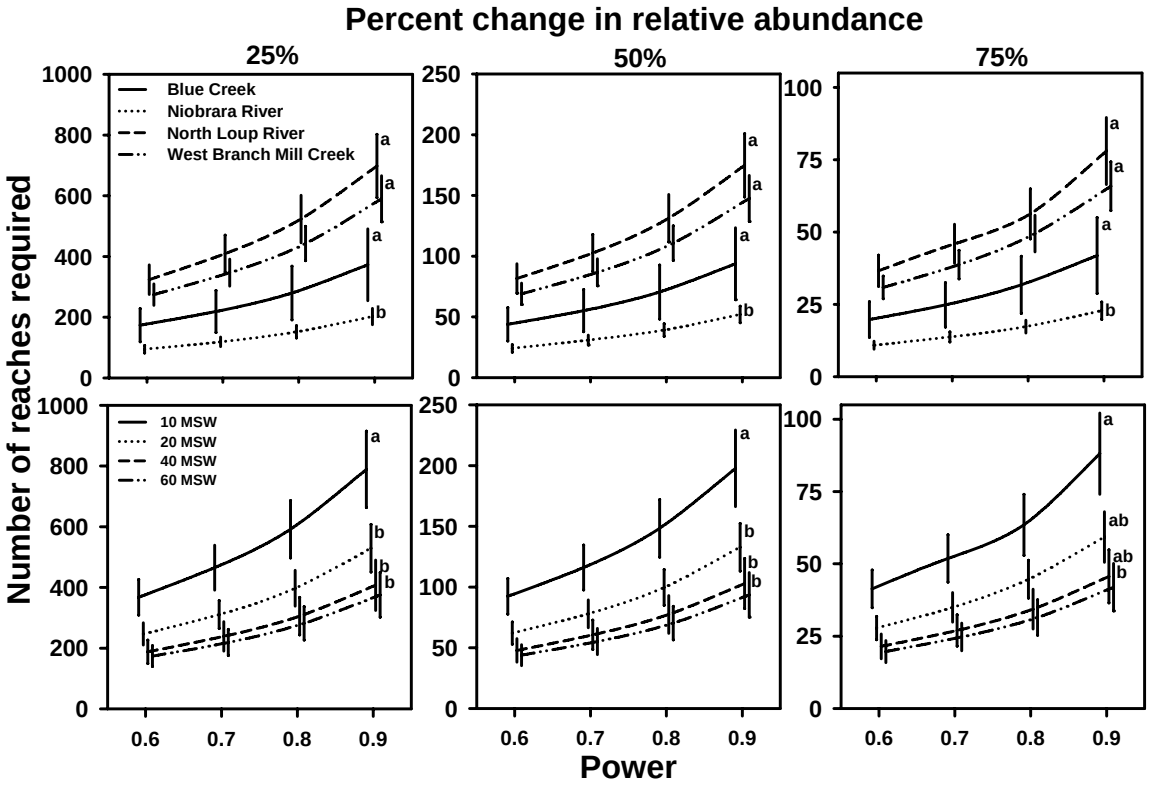

Fig. 3. Mean number of reaches required to detect a 25, 50, and 75% change in catch per unit effort (fish/100 m) of common fish species (after removal of rare species, < 1% cumulative catch) for four streams of Nebraska and Kansas (top) and four reach sampling lengths (mean stream width, MSW; bottom) at different levels of statistical power. Bars represent one standard error. The mean number of reaches needed with the same letter did not differ among streams (top) or MSW (bottom) (ANOVA; P >0.008) at 0.8 level of statistical power.

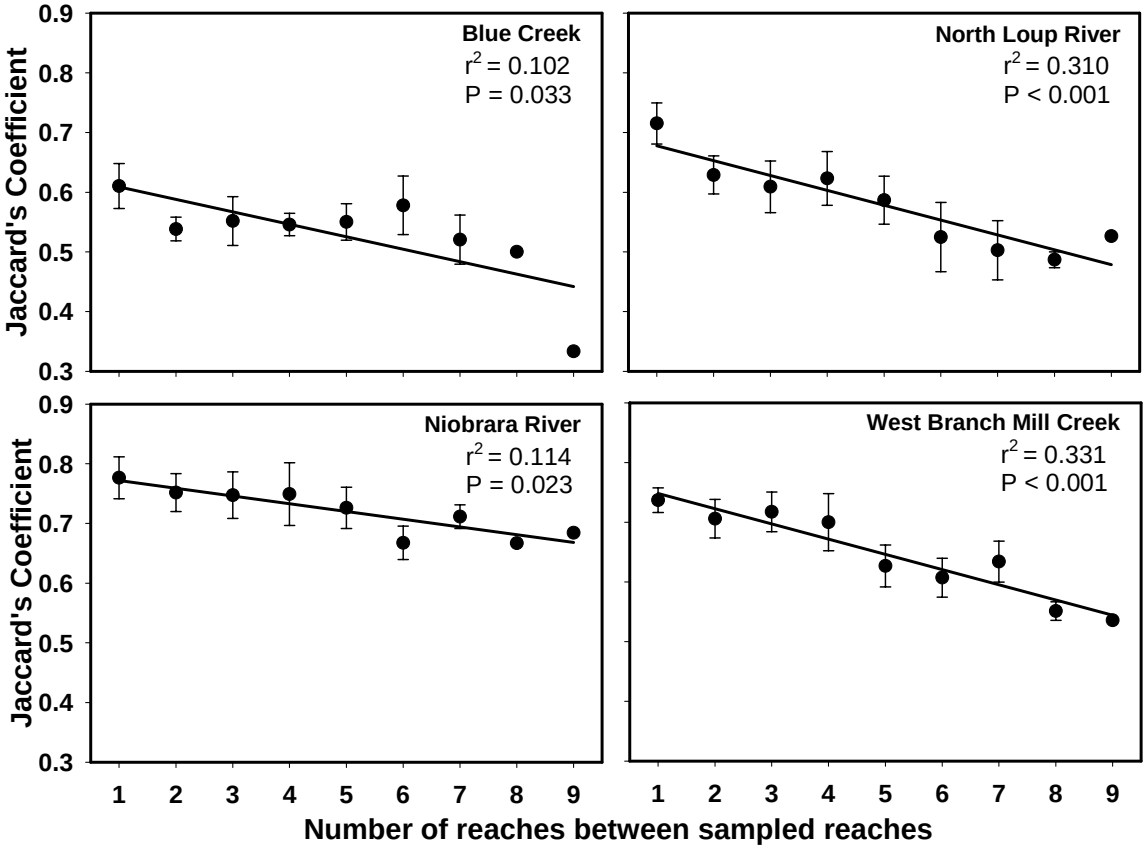

Fig. 4. Relationship between number reaches between samples and mean Jaccard's similarity coefficients against number of sites for four mid-sized streams in Nebraska and Kansas sampled with a pulsed DC towed electrofisher. Error bars represent 1 standard error.

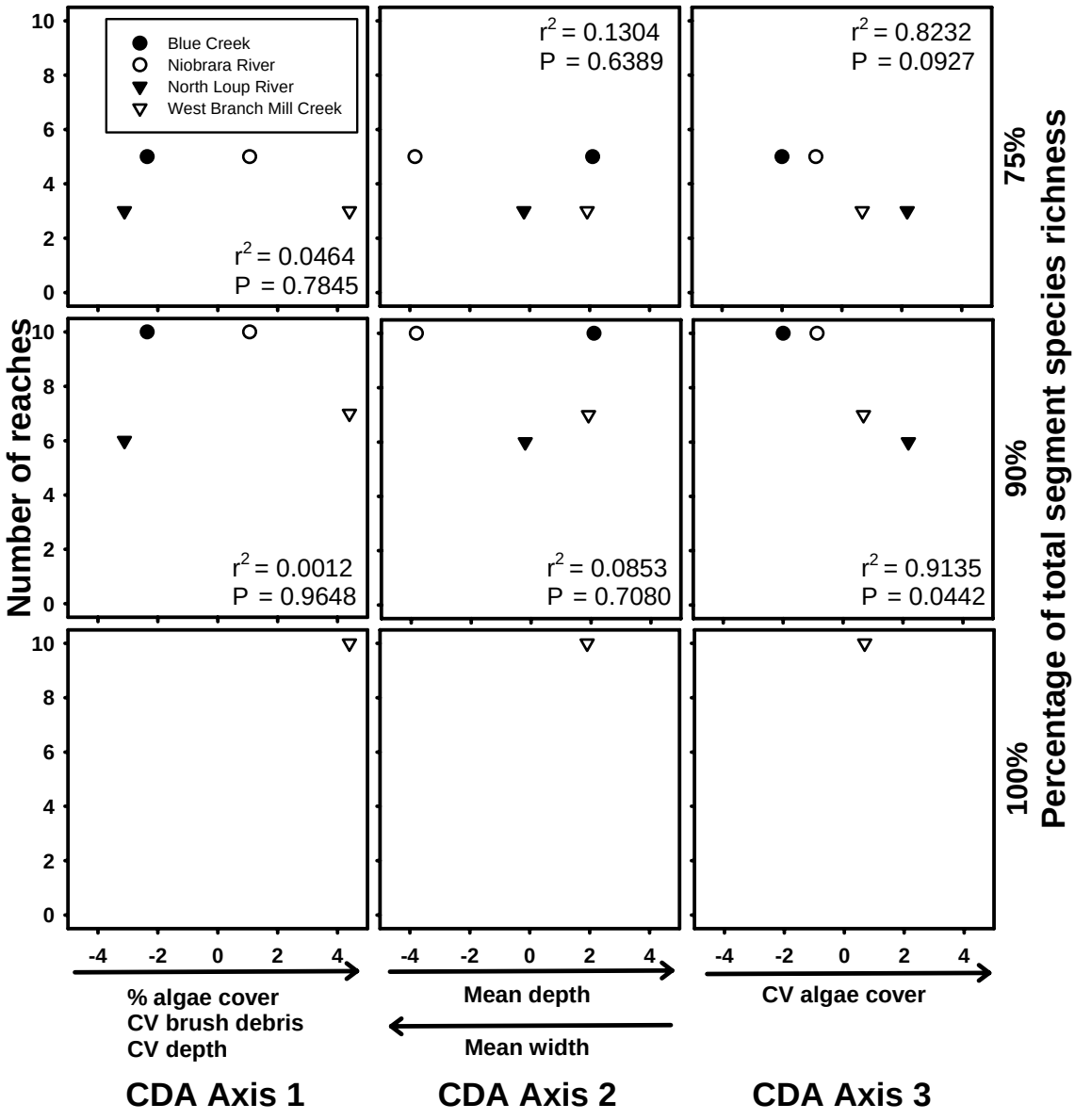

Fig. 5. Relationship between the number of reaches needed to detect 75, 90, 100% of segment species richness with a 95% probability and the mean standardized canonical coefficients from the canonical discriminant analysis (CDA) of 24 habitat surveys from four mid-sized streams in Nebraska and Kansas. Note: The number of reaches required to detect 100% of total segment richness was greater than 10 (unknown) for all streams except West Branch Mill Creek.

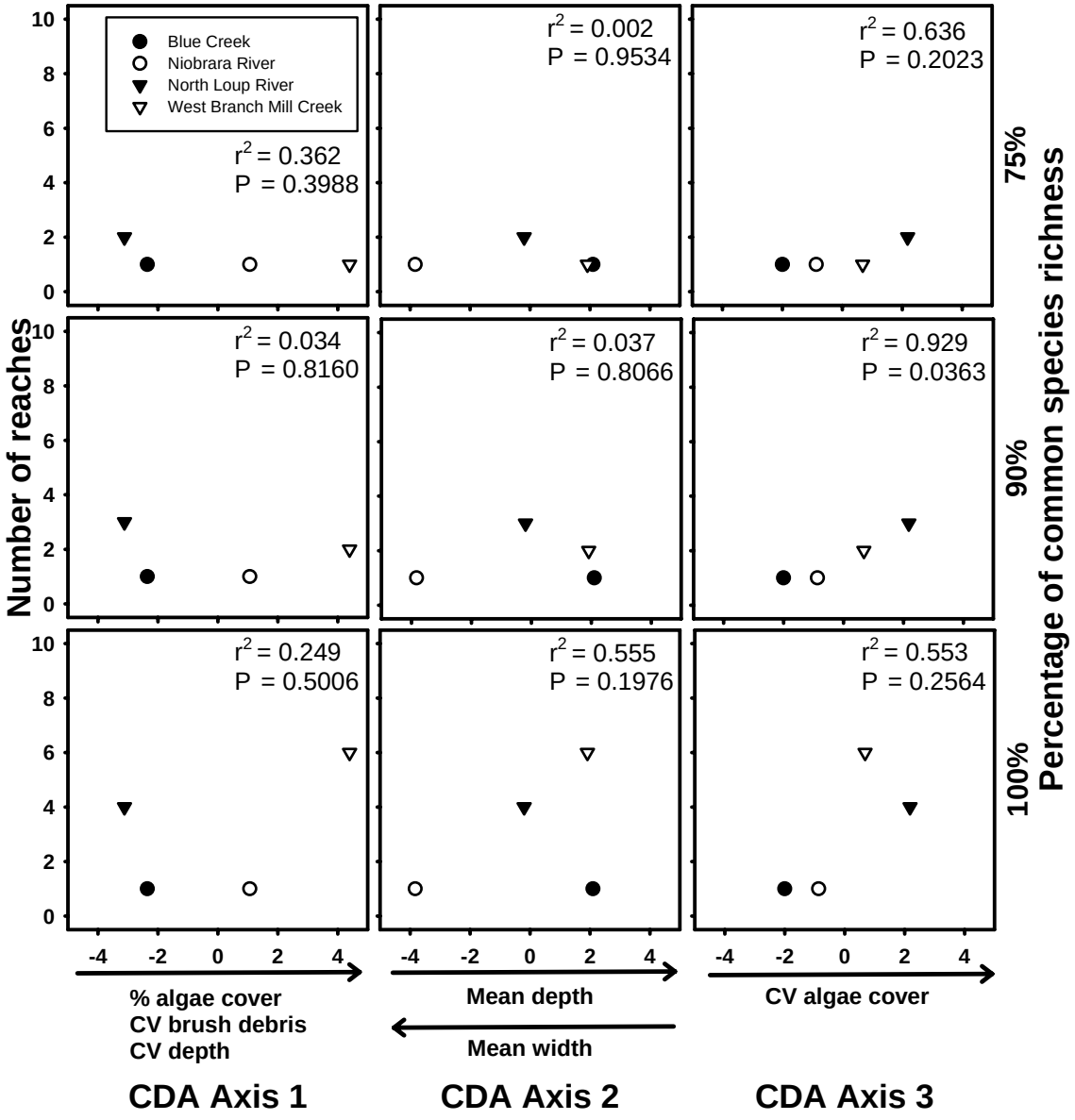

Fig. 6. Relationship between the number of reaches needed to detect 75, 90, 100% of common segment species richness (species accounting for less than 1% of the cumulative catch for a stream segment removed) with a 95% probability and the mean standardized canonical coefficients from the canonical discriminant analysis (CDA) of 24 habitat surveys from four mid-sized streams in Nebraska and Kansas.

Chapter 2

Habitat relationships with fish assemblages in least disturbed Great Plains regions

Abstract.—Effects of local environmental influences on the structure of fish assemblages were evaluated from 159 sites in two regions of the Great Plains with limited anthropogenic disturbance. These least disturbed regions offered an opportunity to evaluate the structure and natural variation of streams and fish assemblages within the Great Plains. We used canonical correspondence analyses to determine the influence of environmental conditions on species abundances, species occurrences, and assemblage characteristics. Analysis of regions separately indicated that similar environmental factors structured streams and fish assemblages, despite differences in environmental conditions and species composition between regions. Variance in fish abundance and assemblage characteristic data from both regions was best explained by metrics of stream size and associated metrics (width, depth, conductivity, instream cover). Our results provide a framework and reference for least disturbed conditions and assemblage structure in North American prairie streams.

Introduction

Understanding of fish-habitat relationships is essential to management and conservation of stream fishes (Kessler and Thorp 1993; Wildhaber et al. 2000). However, stream fish assemblages are structured by habitat at various spatial scales. At local scales, instream physical habitat can influence community structure and function (Gorman and Karr 1978; Schlosser 1982; Angermeier and Karr 1984; Quist and Guy

2001), while broader scales influences (e.g. geology and climate) can be important determinants of species distributions (Mathews and Robison 1988; Marsh-Matthews and Matthews 2000). Since fish assemblages are strongly influenced by physical habitat they have often been used to quantify the effects of disturbance on the environment (Karr 1981). Disturbance to physical habitat (e.g. riparian vegetation alteration, impoundments; Gorman and Karr 1978; Jones et al. 1999; Marchetti and Moyle 2000; Quist et al. 2003), and water quality (e.g. pollution, sedimentation; Tsai 1973; Rabeni and Smale 1995; Bonner and Wilde 2002) have resulted in fish assemblage shifts, decreased native species diversity, community homogenization, range reduction and extinction.

Disturbances to stream ecosystems can be assessed by sampling fish assemblages (Karr 1981). Fish assemblage structures such as guilds (e.g. feeding, reproduction, behavioral, etc.) can be used determine habitat degradation associated with non-point source pollution (Moyle 1994). Furthermore, the effects of disturbances can be long-lasting and possibly permanent despite reduction in the intensity of disturbance (Harding et al. 1998). In some regions, such as the Great Plains, only after anthropogenic influence occurred were most accounts of the fishes documented, leaving pre-settlement conditions poorly documented (Matthews 1988; Fausch and Bestgen 1997). Despite the difficulty in quantifying baseline conditions (because of the limited number of unmodified systems and a long history of disturbance) determining areas of least impact (i.e. reference) is important to designating areas in need of conservation (Hughes et al. 1986), quantifying intensity of disturbance (Mebane 2001) and determining natural variation in fish communities (Karr et al. 1986; Schlosser 1990; Smogor and Angermeier 2001). Additionally, least impacted systems are important to assessing restoration efforts

(Hughes et al. 1986) and validating metrics of fish communities (index of biotic integrity IBI, Smogor and Angermeier 2001).

With losses of native terrestrial vegetation of up to 98.0% in area, the Great Plains are one of the most endangered terrestrial ecosystems in the North America (Samson and Knopf 1994) and prairie streams are even more endangered due to watershed fragmentation (Dodds et al. 2004). Prairie stream fish are at risk because of alterations to land and water use, primarily driven by agricultural practices, but also physical modification (e.g., impoundment for flood control and irrigation, channelization, and riparian vegetation alteration), pollution, siltation, and the introduction of non-native species (Cross and Moss 1987; Pflieger and Grace 1987; Matthews 1988; Dodds et al. 2004; Gido et al. 2004). However, the intensity of anthropogenic disturbances is not evenly distributed across the Great Plains. Eastern regions have experienced greater changes to native terrestrial vegetation (e.g. Indiana, Illinois, and Iowa have had 99.9% declines in native tallgrass prairie) (Samson and Knopf 1994), while northern and western regions tend to have less anthropogenic disturbances (e.g. increased rangeland, less urbanization, lower human density) (Bramblett et al. 2005) and may be useful as reference. It is therefore important to understand how stream systems with minimal environmental disturbance structure fish assemblages within regions.

Our objective was to determine the relationship of fish community structure in response to environmental variation in two geographically separate regions of the Great Plains with limited disturbance by 1) evaluating the structuring of stream sites by environmental variables within regions 2) determining the environmental variables responsible for structuring fish communities for each region and 3) determine if similar

environmental features are related to community structure across a large scale of least disturbed areas. Because these regions are relatively unimpacted (see below) we hypothesized local environmental variables that are not directly related to anthropogenic disturbance will be most important to structuring fish assemblages. We predict that habitat variables of greatest importance will be similar between regions due to relatively homogenous and similar landscapes of each region.

Study Area

The Southwestern Tablelands located within south central Kansas (hereafter referred to as Red Hills) and the Sand Hills (located primarily in North Central Nebraska) EPA Level III ecoregions (Omernik 1987) are relatively unimpacted regions with low human population density and rangeland as the dominant land use (Chapman et al. 2001). The prevalence of rangeland within these regions retains a landcover more similar to the native vegetation (i.e. mixed grass prairie) of both regions as compared more pervasive land uses (e.g. rowcrop agriculture). Landcover within the Sand Hills is 88.8% grassland with only 3.6% rowcrop agriculture, while 50.9% of the Red Hills is grassland and 42.3% is rowcrop agriculture. However the Sand Hills are relatively homogeneous, while the Red Hills can be divided into two distinct areas, the Flat Tablelands and Cimarron Breaks. Within the Flat Tablelands crop production is greater than that of Cimmaron Breaks (Chapman et al. 2001). After the removal of the Flat Tablelands region from the Red Hills, 77.5% of the remaining region is grassland with 16.8% rowcrop agriculture. Streams within the Sand Hills and Red Hills are similar and have predominantly sand substrates and tributary streams that are spring fed by the Ogallala Aquifer (Chapman et

al. 2001). However, prairie streams in the northern Great Plains (i.e. Sand Hills) typically have more stable hydrologic regimes than those of southern Great Plains (Oklahoma and Kansas) (Mathews 1988) and have experience less change in water table level of the Ogallala Aquifer.

Methods

Data collection.—Fish were sampled with a backpack or tote barge electrofisher in a single upstream pass at all sites 159 during the summer of 1996-2005 in two Great Plains regions (Figure 1). Electrofishing was supplemented by seining when sites did not contain excessive instream vegetation, boulders, and/or woody debris. 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 (Lazorchak et al. 1998). Prior to sampling, block nets were established at the upstream and downstream ends to prevent fish movement out of the site. Streams with high discharges and/or excessive widths prevented the use of block nets at all streams. 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 variables were measured at all sites with a modification of the Environmental Protection Agency's (EPA) protocol for sampling wadable streams (Lazorchak et al. 1998). The only variation to EPA protocol was the reduction of 11 evenly-spaced transects to six for all Sand Hills and 12% of Red Hills sites. Six to 11 evenly spaced transects for each sampling length was used to survey physical habitat at each site. 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 measurements was calculated. Canopy cover was measured using a spherical densiometer at six locations at each transect and the percent cover for each transect was averaged for each stream (Lazorchak et al. 1998). Substrate was measured at five equally-spaced intervals at each transect. Substrate classes included fine (<0.06 mm), sand (0.06 to 2 mm), gravel fine (2 to 16 mm), gravel course (16 to 64 mm), cobble (64 to 250 mm). The total percentage of each substrate class was calculated for each site. Instream fish cover (filamentous algae, aquatic macrophytes, woody debris, overhanging vegetation and undercut banks) was estimated using a rank (0-4) of five cover classes; absent (0%), sparse (0-10%), moderate (10 to 40%), dense (40 to 75%) and very dense (>75%) (Lazorchak et al. 1998). Mean rank was then averaged across all transects to obtain a value for all instream fish cover categories for each site. Riparian human influence (i.e. rowcrop agriculture, rangeland, and rip-rap) was visually estimated at each transect on each bank. Observations were categorized into four ranks (i.e. 0-3): absent, on bank (adjacent to water), within 10 m, > 10 m. The mean of ranks was averaged across all transects for each site.

Prior to fish sampling and physical habitat measurements, *in situ* water chemistry was measured at the downstream transect. Dissolved oxygen (mg/L), conductivity (μS/cm), water temperature (C) and turbidity (NTUs) were measured using a handheld meter. Twenty evenly-spaced measurements across the transect were taken at 60% depth with a Marsh McBirney Flo-Mate 2000 or Pygmy flow meter to calculate total stream discharge.

Statistical analyses.—Principal component analyses (PCA) were performed on the 22 environmental variables to reduce dimensionality of the data and identify patterns of sites structuring within the Red Hills and Sand Hills (Johnson 1998). All variables were $\log_{10}(x + 1)$ transformed before analysis to better meet the assumptions of normality. Only principal components with eigenvalues greater than two were interpreted from PCA on the correlation matrix of the environmental variables (Ferre 1995). Variable loadings with absolute values greater than 0.25 were considered important in structuring streams within each region (Chatfield and Collins 1980). We computed Pearson's correlation coefficients for environmental variable loadings for interpreted axes between regions to determine if similar gradients structured streams between regions. A multiple analysis of variance (MANOVA) was conducted to compare all environmental variables between regions. If the MANOVA was significant then individual one-way ANOVAs were use to identify the variables that differed between regions (Johnson 1998).

Fish assemblage data were summarized into two data sets for each region to quantify patterns in relationships between the Red Hills and Sand Hills. Relative abundance of fish species (individuals/100 m) and fish assemblage characteristics (comprised of 13 metrics) were calculated for all sites. Previous studies have demonstrated the utility of fish assemblage characteristics for disturbance (Karr 1981; Fausch et al. 1984; Smogor and Angermeier 2001). The assemblage metrics computed for each site were: total fish abundance (total individuals/100 m), species richness, native cyprinids, native centrachids, and native benthic invertivores, proportion of top carnivores, invertivores, omnivores, intolerant, tolerant, simple lithophils, introduced

individuals, and Shannon's diversity index. Metrics included in the assemblage characteristics data set were adapted from similar metrics from regions similar to this study (Bazata 2005; Lydy et al. 2000; Bramblett and Fausch 1991). Fish species that did not occur at greater than five percent of sites within each region, and sites with a total abundance of less than five individuals per 100 m sampled were removed from all analyses. Since fish community structure may differ between regions a MANOVA was conducted to compare fish assemblage characteristics between regions. If the MANOVA was significant then individual one-way ANOVAs were used to identify which metrics differed between regions (Johnson 1998).

Canonical correspondence analysis (CCA) was used to identify the relationships of environmental variables with fish assemblage (abundance and assemblage characteristics) data sets for the Red Hills and Sand Hills separately. A stepwise forward selection and Monte Carlo permutation test (1,000 random permutations) was used to determine environmental variables that significantly ($P < 0.05$) explained variation in fish assemblage data sets and partial CCAs were used to determine the individual variable variance explained after the removal of variables with inflation factors greater than 10 (ter Braak and Smilauer 2002). Pearson's correlation of percent variances explained by environmental variables between regions was used to determine if assemblage structuring was similar for both regions in the Great Plains. The influence of environmental variables on fish assemblages in both regions were evaluated with combined CCAs for abundance and assemblage characteristics data sets using region as a covariable.

We computed Pearson's correlation coefficients between assemblage characteristics (native cyprinid richness, native benthic invertivore richness proportion

introduced individuals, proportion intolerant individuals) and CPUE of two introduced species (common carp and largemouth bass) to determine the effects of introduced species on native fish structure.

We computed Pearson's correlation coefficients between assemblage characteristics (native cyprinid richness, native benthic invertivore richness, percentage of introduced individuals, percentage intolerant individuals) and CPUE of two introduced species (common carp and largemouth bass) to determine the effects of introduced species on native fish structure. Assemblage characteristic richnesses and species CPUE were $\log_{10}(x + 1)$ transformed and percentages were arcsin transformed. All sites with zero for both metrics to be correlated were removed before analysis.

Results

The PCA of Red Hills sites produced three axes that cumulatively explained 45.5% of the environmental variation in sites (Table 1). The first axis had high loadings for instream cover (filamentous algae, macrophytes, and overhanging vegetation), stream size (mean width and discharge), turbidity, and substrate composition (percent fine and sand). The second axis had high loadings for instream cover of large woody debris, undercut banks, canopy cover, and conductivity. The third axes had high loadings for percent cover of small woody debris, gravel substrate composition (fine and course), and adjacent rangeland landuse. The PCA of Sand Hills sites produced three axes that explained 41.7% and high loadings for all three axes were similar to that of the Red Hills PCA. The first axis included high loadings for instream cover (macrophytes and small/large woody debris), canopy cover, stream size (mean width and discharge) and

adjacent rangeland landuse. The second axis produced high loadings for dissolved oxygen and substrate composition (sand and course gravel). The third axis had high loadings for mean depth and presence of rip-rap.

The PC scores for both the Sand Hills and Red Hills were correlated for both axis 1 ($r = 0.493$, $P = 0.02$, $N = 22$) and axis 2 ($r = 0.492$, $P = 0.02$, $N = 22$) suggesting similar environmental variables structured sites separately within the Sand Hills and Red Hills regions. However, environmental variables differed between regions (Wilk's lambda = 0.079; DF = 22,136; $P < 0.0001$). Nine variables (percent macrophyte cover, large/small woody debris, canopy cover, mean width, discharge, percent sand substrate, percent course gravel substrate, and adjacent rangeland landuse) of 22 total environmental variables had high loadings on at least one of the principal component axes interpreted for both the Red Hills and Sand Hills. The primary gradients included stream size (mean width, mean depth, discharge) and instream cover (e.g. macrophytes, woody debris, etc.) for both regions. In contrast, water temperature, percent cobble substrate, and adjacent rowcrop agriculture did not have high loadings on any of the first three axes for either region (Table 1).

Sixty one fish species were collected from 92 Red Hills and 67 Sand Hills sites: 23 from the Red Hills and 33 from the Sand Hills (Table 2). Seventeen species were common between the two regions, of which nine species occurred at greater than 10% of each region's sites. However, six species were unique to the Red Hills and 16 species were unique to the Sand Hills. Many of the region-specific species were common and abundant in their region (Table 2). Five species that were ubiquitous (> 50% of sites) in the Red Hills included Arkansas darter *Etheostoma cragini* (a Kansas state threatened

species; Haslouer et al. 2005), green sunfish *Lepomis cyanellus*, plains killifish *Fundulus zebrinus*, sand shiner *Notropis stramineus*, central stoneroller *Campostoma anomalum*, and red shiner *Cyprinella lutrensis*. Species that occurred at greater than 50% of the sites in the Sand Hills included white sucker *Catostomus commersonii*, creek chub *Semotilus atromaculatus*, longnose dace *Rhinichthys cataractae*, plains topminnow *F. sciadicus* (a Nebraska Tier I at risk species; Schneider et al. 2005), bigmouth shiner *N. dorsalis*, fathead minnow *Pimephales promelas*, and sand shiner.

A total of 39,722 fish were collected from Red Hills sites and 41,253 fish from Sand Hills sites. Total catch for sites ranged from 12 to 3,215 individuals for the Red Hills and 15 to 6,139 for Sand Hills sites (Table 3). Fish assemblage characteristics varied between regions (Wilk's lambda = 0.317; DF = 13,145; $P < 0.0001$). The number of native cyprinids, number of benthic invertivores, proportion of tolerant individuals, proportion of lithophilic individuals, and Shannon's diversity was higher in the Sand Hills sites, while the number of native centrachid species and proportion of intolerant individuals was higher in the Red Hills sites (Table 3). All other assemblage characteristics (total fish abundance, species richness, proportion of top carnivores, proportion of invertivores, proportion of omnivores, and the proportion of introduced individuals) did not differ between regions. Overall, the mean proportion of introduced species was low (i.e. < 5%) in both regions (Table 3).

Fish abundance.—The forward selection procedure retained 8 of the 19 environmental variables (Ps < 0.05) in the CCA of Red Hills sites and fish abundance data (Table 4). Axis 1 (33.4% of total variance) represented a gradient of substrate and

stream depths, which separated species with associations towards increased mean depth and fine substrate (western mosquitofish *Gambusia affinis*, fathead minnow, gizzard shad *Dorosoma cepedianum*, and common carp *Cyprinus carpio* from other species (Figure 2). Axis 2 (25.3% of total variance) represented a gradient of stream size (mean width and conductivity) and instream cover (filamentous algae, and undercut banks). This axis contrasted species associated with narrower streams having greater instream cover (e.g. Arkansas darter, central stoneroller, green sunfish) from species associated with wider streams and less instream cover (e.g. plains minnow, emerald shiner *N. atherinoides*, sand shiner, Figure 2). Nearly 40% of the Red Hills fish abundance variance explained by all environmental variables was explained by percent fine (13.0%) and sand substrates (10.1%), and mean width (8.5%) and conductivity (7.5%) which was far greater than the cumulative variance explained by the five remaining forwarded selected variables (14.1%) (Table 4).

Nine variables significantly explained variation of Sand Hills fish abundance data (Table 4). Axis 1 (30.9% of total variation) represented a gradient of conductivity, instream cover (i.e. macrophytes), and stream depth (Figure 2). Deeper streams with higher macrophyte coverage and lower conductivity were associated with orangethroat darter, creek chub, brook stickleback, and plains topminnow. Shallower streams with higher conductivity and less instream cover contained sites with increased abundance of bluntnose minnow *P. notatus*, common carp Johnny darter, river shiner *N. blennius*, and sand shiner. Axis 2 (18.9% of total variation) represented a gradient of stream width, instream cover (i.e. undercut banks), and substrate. Species associated with wider streams with less instream cover and cobble substrate included yellow perch *Perca*

flavescens, plains minnow, and shorthead redhorse *Moxostoma macrolepidotum*, while narrower streams with larger substrate and increased undercut bank coverage included species associations of Iowa darter *E. exile*, longnose dace, northern redbelly dace, and brassy minnow *H. hankinsoni* (Figure 2). The most important variable in structuring Sand Hills fish abundance was macrophyte cover (12.3% of variance explained) followed by conductivity (11.6%) and mean depth (11.2%) (Table 4).

Fish assemblage characteristics.—Eight environmental variables were selected as contributing to fish assemblage characteristics of the Red Hills (Table 4). Axis 1 (38.0% of total variance) represented a gradient of stream depth and instream cover (i.e. undercut banks), while Axis 2 (18.9% of total variance) represented a gradient of stream size (mean width and discharge), conductivity, substrate, and instream cover (overhanging vegetation and macrophytes). Patterns of fish assemblage characteristics indicated that intolerant species were associated with narrower streams that had greater instream cover, while simple lithophilic spawners were associated with wider streams and higher discharges (Figure 3). The proximity of top carnivore to introduced species from the CCA ordination (Figure 3) indicated these characteristics were shared by the same species. Only two top carnivore species (channel catfish and largemouth bass) existed in the Redhills and the largemouth bass was also an introduced species. Overall, species richness was higher in wider streams and total abundance was greater in shallower streams (Figure 3). Native centrachid richness and proportion of tolerant individuals was higher in deeper and presumably more stable streams compared to that of shallower streams. Of the eight variables selected, discharge explained the greatest variance

(11.0% of variance explained by all variables) while mean width explained 9.8% and macrophytes cover explained 8.4% (Table 4).

Eight environmental variables significantly explained variation of the Sand Hills fish assemblage characteristics data (Table 4). Axis 1 (50.2% of total variance) was influenced by the presence of rip-rap and represented a gradient of substrate (i.e. cobble to fine), while Axis 2 (26.2% of total variance) represented a gradient of stream size (i.e. mean width) conductivity, and dissolved oxygen. Total fish abundance was associated with wider streams while proportion of intolerant, simple lithophilic, and invertivore individuals was associated with narrower streams with lower conductivity and discharges. Proportion of top carnivores to introduced individuals was associated with the presence of rip-rap and cobble substrate. Rip-rap (10.6%) and cobble (8.0%) substrate explained the greatest amount of variance of the Sand Hills assemblage characteristics (Table 4). The proximity of top carnivore and introduced individuals indicated that top carnivores are predominately introduced within the Sand Hills. Within the Sand Hills streams, northern pike *Esox lucius* and largemouth bass, are non-native species and top carnivores.

Overall, the Red Hills and Sand Hills sites exhibited patterns of assemblage environmental associations along gradients of stream size, instream cover, and substrate with similar variables and total variation explained (Table 4). However, percent variance explained by all environmental variables were not related for fish abundances ($r = 0.356$, $P = 0.13$, $N = 19$) nor assemblage characteristics ($r = 0.016$, $P = 0.94$, $N = 21$) between regions. Mean width was included in all four CCAs and explained 4.6 to 9.8% of the variance explained in the fish assemblage data of both regions (Table 4), whereas

conductivity, mean depth, undercut banks, and percent fine substrate where include in three of the four CCAs (Table 4). Percent macrophyte cover, discharge, dissolved oxygen, sand substrate, cobble substrate, and rangeland land use were included in two of four CCAs (Table 4). Five variables (large woody debris, small woody debris, water temperature, course gravel substrate, and adjacent rowcrop landuse) did not significantly explain variation in any of the fish assemblage data sets (Table 4).

Combined analyses.—The ordination of the sample scores for the CCA of combined fish abundance data indicated a distinct separation between regions. The variance of fish abundance explained by the first two axes (41.9%) was lower than that explained by the similar axes of the CCAs of both regions separately (58.7 and 69.8%) (Table 4). However, 14 of 22 environmental variables significantly explained variation in the combined fish abundance data. Three variables (percent fine substrate, conductivity, and adjacent rangeland landuse) cumulatively explained (20.2%) of the variance in fish abundances in both regions while the remaining 11 variables cumulatively explained (35.0%) (Table 4). Axis 1, which was associated with conductivity and rangeland landuse separated Red Hills and Sand Hills sites (Figure 4). Conductivity and adjacent rangeland landuse were higher in the Red Hills, however percent fine substrate did not differ between regions (Table 3). Despite some overlap in the species present in both regions (Table 2), fish abundances were structured differently in relation to similar environmental variables.

The cumulative variance explained (68.2%) by the first two axes of the CCA of combined assemblage characteristics was similar to CCAs for regions separately (69.8

and 76.4%) (Table 4). Eleven variables contributied significantly to the variance explained, but only mean stream depth explained greater than 5% of the variance. Overhanging vegetation (4.2%) and canopy cover (4.2%) (Table 4), which differed between regions (Table 2) had the next highest variances explained. Despite differing environmental variables and assemblage characteristics (Table 2) a greater overlap in site scores than the combine fish abundance CCA ordination (Figure 4) was observed.

Native cyprinid richness was negatively related to the percentage of introduced species in the Red Hills and Sand Hills regions ($r = -0.370$, $P < 0.001$, $N = 142$), but was not related to CPUE of common carp ($r = -0.138$, $P = 0.105$, $N = 139$) nor largemouth bass ($r = -0.058$, $P = 0.496$, $N = 141$) (Figure 5). Similarly, native benthic invertivore richness was not related to CPUE of common carp ($r = 0.127$, $P = 0.118$, $N = 153$) nor largemouth bass ($r = 0.107$, $P = 0.189$, $N = 143$) (Figure 5). However, the percentage of intolerant individuals was negatively related to CPUE of common carp ($r = -0.219$, $P = 0.011$, $N = 135$) and largemouth bass ($r = -0.221$, $P = 0.009$, $N = 137$) (Figure 5). When common carp or largemouth bass abundance was above approximately 1 fish/100 m, the presence of intolerant individuals was rare and only exceeded 50% when common carp and largemouth bass very low or zero.

Discussion

Local fish assemblages are structured by environmental influences at multiple spatial scales (Schlosser 1982; Schlosser 1987; Marsh-Matthews and Matthews 2000) that can vary regionally (Hoeinghaus et al. 2007). We found consistent environmental relationships with fish assemblage structure in two geographically separated regions of

the Great Plains, despite large differences in the habitat and fish communities. For example, mean conductivity was on average eight times higher in the Red Hills compared to the Sand Hills. Additionally, our results indicate that patterns of assemblage structure were influenced by similar environmental factors that structured assemblage characteristics concurrently. Variation in assemblages in both regions was explained best by stream size (i.e. depth, width, and discharge) and metrics linked to stream size (e.g. smaller streams had increased percentage of instream cover; larger streams had greater conductivity, etc,). Taylor et al. (1993) found conductivity and stream width to be the most important variables in structuring sites and fish assemblages within streams in the Red River drainage of Oklahoma, which has a similar fish assemblage to the Red Hills (12 of 17 fish species shared). Our study also indicated conductivity was strongly associated with fish assemblage structure, but in regions north and west of Taylor et al. (1993) study, suggesting that conductivity may be important in structuring fish assemblages at larger spatial scales. Although the mean values for 15 of the 22 environmental variables used in our study differed between the Red Hills and Sand Hills, (often up to three fold or higher differences), fish communities were structured by similar environmental variables. This separation in environmental conditions between regions suggests regional geology was important to the variables of our study, but sites within regions structured similarly to environmental variables despite these differences.

Fish communities differed between the Red Hills and Sand Hills, but the total fish abundance and species richness collected at sites between regions did not differ. Great Plains streams have relatively low species diversity compared to warmwater streams of other regions (Pflieger 1997), as a result of the widely fluctuating environmental

conditions that characterize prairie streams. Therefore, Great Plains fishes are adapted to harsh conditions and often trophic and substratum generalists (Poff and Allan 1995). However, the proportion of omnivores in both of our study regions did not differ and was relatively low ($\bar{x} < 20\%$) compared to the Arkansas River in Colorado (Bramblett and Fausch 1991). Prairie stream fish assemblages are often tolerant due to highly variable environmental conditions, such as flow regime (Bramblett et al. 2005). However, we found the proportion of tolerant individuals greater in the more hydrologically stable Sand Hills streams, whereas the proportion of intolerant individuals was greater in the Red Hills. Results from our assemblage characteristic analysis suggests differing fish assemblages were structured by similar environmental variables (e.g. proportion of intolerant individuals in both regions was associated with narrower streams) in two separate regions of the Great Plains.

Stream size was the most important factor in structuring assemblages in our study which is consistent with Schlosser (1987). However, stream habitat and fish assemblages throughout the Great Plains are not uniform (Matthews 1988), which may influence how fish assemblages at small spatial scales respond to environmental variation. We found that substrate composition and instream cover were also important in structuring fish assemblages in both regions, but were likely indirectly associated with stream size. Large streams of the region are typically broad, shallow, often braided with sandy substrates, and increased dissolved solids (Matthews 1988). Large streams often have low habitat diversity (Bramblett and Fausch 1991) which is in contrast to the Schlosser's (1987) conceptual model of increased habitat heterogeneity with increased stream size.

For example, in narrower streams canopy cover was often higher, therefore increasing percent cover of woody debris relative to that of larger streams.

Prairie streams of the Great Plains are unique systems that support unique fishes. We found several species of regional conservation concern in high abundance in both regions. Many of these species were strongly associated with the environmental factors of our study. For example, the presence and abundance of the Arkansas darter (Kansas state threatened; Haslouer et al. 2005), which was collected at greater than 60% of the sites in the Red Hills, was strongly associated with narrower streams containing greater instream cover and lower conductivity. The plains topminnow is an endemic species to the Great Plains whose conservation status is listed as vulnerable, critically imperiled, or extinct in seven of the nine states once found (National Heritage Network 2007). Our study collected the plains topminnow at greater than 60% of sites sampled and our analysis indicated the species strongly associated with small streams containing increased macrophyte cover. The presence of rare and sometimes intolerant species and the low proportion of introduced individuals in both regions supports that these streams are of least disturbance or healthy systems (Bramblett and Fausch 1991, Lydy et al. 2000, Shearer and Berry 2002, Bramblett et al. 2005).

The local environmental conditions of streams are linked to the landscape at multiple spatial scales. Our results indicate that local factors were important in structuring fish assemblages in two regions of the Great Plains, but we did not directly test landscape or large scale factors in this study. Large scale terrestrial and geographic features are important to structuring assemblages (Roth et al. 1996, Marsh Matthews and Matthews 2000, Gido et al. 2006). However, fish assemblages have been found to be

more influenced by local factors in largely undisturbed catchments (Wang et al. 2006) or highly disturbed regions (Stauffer et al. 2000, Heikte et al. 2006). The strong local environmental influence of fish assemblage structure may be the result of similar homogenous landscapes of the Sand Hills and Red Hills regions.

Conservation of stream fishes is dependent on understanding influences at multiple scales and reference conditions (e.g. fish habitat relationships at least disturbed sites) are essential to assessing restoration efforts (Roni et al. 2005). Our results provide a framework of the how local habitat factors structure differing stream fish assemblages in two least disturbed Great Plains regions. Although few undisturbed regions may currently exist in the Great Plains, our study provides a comprehensive survey of conditions in areas of low anthropogenic disturbance and demonstrates the current variability in fish communities and environmental conditions of these two regions. The consistency of our results across a large region suggests the importance of local influences on communities and can be used by managers to assess conditions of streams in other Great Plains regions.

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Table 1. Principal component (PC) loadings from principal component analysis (PCA) of instream physical habitat structure, physiochemical, and adjacent land use environmental variables from 67 Sand Hill and 92 Red Hill sites collected in 19996-2005. Variable loadings with absolute values ≥ 0.25 in bold.

	Red Hills			Sand Hills		
Variable	PC1	PC2	PC3	PC1	PC2	PC3
Filamentous algae cover (mean rank)	**-0.31**	-0.11	0.08	-0.05	0.13	0.02
Macrophyte cover (mean rank)	**-0.38**	-0.09	**-0.27**	**-0.34**	0.30	0.02
Large woody debris cover (mean rank)	0.07	**0.25**	**0.31**	**0.40**	0.07	-0.10
Small woody debris cover (mean rank)	0.19	0.16	**0.29**	**0.30**	0.04	**-0.25**
Overhang vegetation cover (mean rank)	**-0.28**	0.00	-0.21	-0.04	0.22	-0.16
Undercut bank cover (mean rank)	-0.05	**0.30**	0.08	0.16	0.09	-0.17
Canopy cover (%)	-0.06	**0.45**	-0.13	**0.29**	0.23	-0.20
Mean depth (cm)	-0.01	0.18	-0.18	0.03	0.11	**0.44**
Mean width (m)	**0.39**	-0.12	0.03	**0.34**	-0.18	0.26
Discharge (m/sec)	**0.41**	0.08	-0.06	**0.29**	-0.15	**0.38**
Temperature (ºC)	0.07	-0.23	0.22	0.03	0.07	-0.23
Dissolved oxygen (mg/L)	0.17	0.24	-0.22	0.07	**0.33**	-0.06
Conductivity (µS/cm)	0.07	**-0.41**	0.15	0.17	-0.24	-0.21
Turbidity (NTUs)	**0.27**	0.09	**-0.26**	0.02	-0.11	0.07
Fine substrate (%)	**-0.25**	0.02	-0.08	-0.24	0.24	0.10
Sand substrate (%)	**0.32**	-0.16	-0.13	0.01	**-0.45**	-0.20
Fine gravel substrate (%)	-0.14	0.23	**0.29**	0.11	0.24	0.16
Course gravel substrate (%)	-0.07	0.23	**0.35**	0.19	**0.31**	**0.30**
Cobble substrate (%)	-0.03	0.16	0.21	0.20	0.22	0.00
Adjacent rowcrop landuse (presence/absence)	0.07	0.12	0.10	-0.01	-0.07	0.20
Adjacent rangeland landuse (presence/absence)	0.08	0.20	**-0.40**	**-0.36**	-0.08	-0.01
Rip-rap (presence/absence)	0.04	0.20	-0.04	0.05	0.24	**-0.32**
Eigenvalue	4.17	3.26	2.58	4.06	2.90	2.42
Percent variance explained	19.0	14.8	11.7	18.5	13.2	11.0
Cumulative variance explained	19.0	33.8	45.5	18.5	31.7	41.7

Table 2. Summary of species occurrences from 67 Sand Hill and 92 Red Hill sites collected from 1996-2005. Values are number of sites where species was collected.

Common Name	Scientific name	Abbreviation	NE	KS
Arkansas darter	*Etheostoma cragini*	arkdart	-	57
Bigmouth shiner	*Notropis dorsalis*	bigshnr	37	-
Black bullhead	*Ameiurus melas*	blkbull	12	20
Bluegill	*Lepomis macrochirus*	blugil	9	32
Blugill x green sunfish hybrid	*L. macrochirus x cyanellus*	blgxgrn	2	3
Bluntnose minnow	*Pimephales notatus*	blntnos	5	1
Brassy minnow	*Hybognathus hankinsoni*	brsymin	12	-
Brook stickleback	*Culaea inconstans*	brkstk	25	-
Brown trout	*Salmo trutta*	brntrt	1	-
Central stoneroller	*Campostoma anomalum*	centstn	5	49
Channel catfish	*Ictalurus punctatus*	chnlcat	6	27
Common carp	*Cyprinus carpio*	comcarp	15	37
Creek chub	*Semotilus atromaculatus*	crkchub	44	-
Emerald shiner	*N. atherinoides*	emshnr	-	21
Fathead minnow	*P. promelas*	fatmin	37	26
Flathead catfish	*Pylodictis olivaris*	flatcat	-	1
Flathead chub	*Platygobio gracilis*	flatchb	4	-
Finescale dace	*Phoxinus neogaeus*	findace	2	-
Freshwater drum	*Aplodinotus grunniens*	frshdrm	1	4
Gizzard shad	*Dorosoma cepedianum*	gizshad	-	8
Golden shiner	*Notemigonus crysoleucas*	goldshn	2	-
Golden redhorse	*Moxostoma erythrurum*	goldred	-	8
Grass pickerel	*Esox americanus*	graspik	9	-
Green sunfish	*L. cyanellus*	grnsun	26	56
Johnny darter	*E. nigrum*	jhndart	5	-
Iowa darter	*E. exile*	iowadrt	4	-
Largemouth bass	*Micropterus salmoides*	lrgbass	14	43
Longear sunfish	*L. megalotis*	longear	-	41
Longnose dace	*Rhinichthys cataractae*	lngdace	43	-
Longnose sucker	*Catostomus catostomus*	lngsker	1	-
Northern pike	*E. lucius*	nrtpike	6	-
Northern redbelly dace	*P. eos*	nredace	11	-
Orangespotted sunfish	*L. humilis*	orgspot	-	5
Orangethroat darter	*E. spectabile*	orgthrt	13	19
Pearl dace	*Margariscus margarita*	prldace	2	-
Plains killifish	*Fundulus zebrinus*	plnskil	-	56
Plains minnow	*H. placitus*	plnsmin	6	29
Plains topminnow	*Fundulus sciadicus*	plnstop	41	-
Pumpkinseed	*L. gibbosus*	pumpkin	5	-
Quillback	*Carpiodes cyprinus*	quilbck	4	-
Red River pupfish	*Cyprinodon rubrofluviatilis*	rdrvpup	-	1
Red River shiner	*N. bairdi*	rdrivsh	-	2
Rainbow trout	*Oncorhynchus mykiss*	rnbwtrt	3	-
Red shiner	*Cyprinella lutrensis*	redshnr	23	49
River carpsucker	*C. carpio*	rvrcarp	3	5
River shiner	*N. blennius*	rivshnr	11	-
Rock bass	*Ambloplites rupestris*	rckbass	1	-
Sand shiner	*N.stramineus*	sndshnr	35	51
Shorthead redhorse	*M. macrolepidotum*	shrtred	17	-
Shortnose gar	*Lepisosteus platostomus*	shrtgar	1	-
Silver chub	*Macrhybopsis storeriana*	slvrchb	1	-
Southern redbelly dace	*P. erythrogaster*	sredace	-	3
Suckermouth minnow	*Phenacobius mirabilis*	skrmth	-	24
Stonecat	*Noturus flavus*	stncat	33	-
Western silvery minnow	*H. argyritis*	wsilvmi	3	-
Walleye	*Sander vitreus*	walleye	-	1
Western mosquitofish	*Gambusia affinis*	wstrnmo	-	41
White crappie	*Pomoxis annularis*	whtcrap	1	1
White sucker	*Catostomus commersonii*	whtsuck	46	-
Yellow bullhead	*A. natalis*	yelbull	4	45
Yellow perch	*Perca flavescens*	yelprch	5	-

Table 3. Summary of fish assemblage characteristics and environmental variable means (standard deviations in parenthesizes) and ranges from 67 Sand Hills and 92 Red Hills sites collected from 1996-2005. Fish assemblage characteristics (multiple analysis of variance; Wilk's lambda = 0.317; DF = 13,145; P < 0.0001) and environmental variables (Wilk's lambda = 0.079; DF = 22,136; P < 0.0001) were different between regions. One-way analysis of variance F-statistics and P-values between region means is reported.

		Red Hills		Sand Hills			
Fish assemblage characteristics	**Code**	**Mean**	**Range**	**Mean**	**Range**	**F**	**P**
Total fish abundance (ind./ 100 m)	abund	199.9 (253.7)	6.1-2143.3	257.6 (515.4)	10.0-4124.7	0.87	0.353
# of fish species	rich	8.1 (5.2)	1.0-18.0	8.7 (3.2)	2.0-16.0	0.64	0.425
# of native cyprinid spp.	cyprin	2.8 (2.3)	0.0-7.0	4.3 (1.9)	0.0-8.0	19.2	<0.0001
# of native centrachid spp.	centra	1.4 (1.2)	0.0-4.0	0.5 (0.6)	0.0-2.0	35.1	<0.0001
# of native benthic invertivores spp.	bentinv	2.4 (1.5)	0.0-6.0	3.3 (1.6)	0.0-7.0	12.8	0.0005
% of top carnivore individuals	carniv	2.0 (5.3)	0.0-38.2	3.3 (11.5)	0.0-80.0	0.89	0.346
% of invertivore individuals	invert	67.3 (29.2)	0.0-100.0	59.5 (25.9)	11.0-99.6	2.99	0.086
% of omniviore individuals	omniv	18.5 (24.0)	0.0-100.0	21.1 (20.4)	0.0-82.9	0.53	0.469
% intolerant individuals	intoler	24.8 (37.3)	0.0-100.0	2.9 (5.6)	0.0-31.0	22.7	<0.0001
% of tolerant individuals	toler	18.1 (23.7)	0.0-99.5	34.5 (24.2)	0.0-82.9	18.2	<0.0001
% of simple lithophil individuals	litho	3.9 (10.4)	0.0-74.8	33.1 (30.9)	0.0-100.0	71.2	<0.0001
% of introduced individuals	intro	3.6 (11.3)	0.0-76.5	4.1 (11.6)	0.0-63.2	0.07	0.789
Shannon's diversity Index	H	1.0(0.6)	0.0-2.1	1.3(0.4)	0.1-2.1	5.82	0.017
Environmental variables							
Filamentous algae cover (mean rank)	-	0.60 (0.8)	0-3.27	0.72 (0.7)	0-3.00	1.91	0.169
Macrophyte cover (mean rank)	-	1.04 (0.9)	0-3.54	1.41 (0.8)	0-3.33	8.74	0.004
Large woody debris cover (mean rank)	-	0.10 (0.2)	0-1.73	0.20 (0.4)	0-1.50	3.82	0.052
Small woody debris cover (mean rank)	-	0.45 (0.3)	0-1.27	0.45 (0.5)	0-2.00	0.32	0.571
Overhang vegetation cover (mean rank)	-	1.30 (0.5)	0.55-3.09	1.14 (0.7)	0-4.0	5.72	0.018
Undercut bank cover (mean rank)	-	0.41 (0.5)	0-2.73	0.41 (0.5)	0-2.67	0.00	0.990
Canopy cover (%)	-	24.3 (22.6)	0-87.6	8.2 (0.1)	0-75.7	204.5	<0.0001
Mean depth (cm)	-	13.2 (9.4)	2.1-52.1	28.2 (12.9)	6.3-59.0	75.6	<0.0001
Mean width (m)	-	9.6 (9.9)	1.1-39.6	16.7 (25.0)	1.3-132.4	5.44	0.021
Discharge (m/sec)	-	22.8 (59.8)	0-480.7	77.0 (110.3)	0.3-508.6	32.8	<0.0001
Temperature (ºC)	-	21.9 (3.5)	13.8-35.5	22.0 (4.3)	11.5-33.9	0.04	0.842
Dissolved oxygen (mg/L)	-	5.7 (2.1)	0-12.2	5.5 (2.6)	0-12.4	1.26	0.2632
Conductivity (µS/cm)	-	1670 (1151)	309-4390	202 (56.7)	118-321	345.7	<0.0001
Turbidity (NTUs)	-	36.9 (103.8)	1.2-876.0	10.7 (8.3)	0.2-50.0	15.4	0.0001
Fine substrate (%)	-	13.1 (0.2)	0-92.7	8.8 (0.2)	0-100	2.08	0.151
Sand substrate (%)	-	71.2 (0.3)	5.5-100	83.3 (0.2)	0-100	8.74	0.004
Fine gravel substrate (%)	-	11.2 (0.1)	0-47.3	2.2 (0.1)	0-30.0	28.3	<0.0001
Course gravel substrate (%)	-	2.0 (0.05)	0-32.7	4.0 (0.1)	0-46.7	3.16	0.077
Cobble substrate (%)	-	0.1 (0.01)	0-5.5	0.7 (0.02)	0-10.0	6.05	0.015
Adj. rowcrop (presence/absence)	-	0.04 (0.2)	0-1	0.04 (0.3)	0-2	0.02	0.8873
Adj. rangeland (presence/absence)	-	1.82 (0.5)	0-2	1.51 (0.8)	0-2	9.01	0.0031
Rip-rap (presence/absence)	-	0.20 (0.2)	0-2	0.40 (0.8)	0-2	3.96	0.048

Table 4. Instream physical habitat structure, physiochemical, and adjacent land use environmental variables used in fish abundance, fish assemblage characteristic and combined canonical correspondence analyses (CCAs) for the Red Hills and Sand Hills sites sampled 1996-2005. Value indicates the percentage of variance explained by variable within each CCA. Variables in bold significantly ($P < 0.05$) correlated with fish assemblage data in forward selection of CCA. See Table 1 for units of environmental variables.

	Abundance		**Assemblage Characteristic**		**Combined regions**	
Variable	Red Hills	Sand Hills	Red Hills	Sand Hills	Abun-dance	Assem-blage
Filamentous algae cover	**4.2**	4.2	4.1	3.6	**2.3**	**1.0**
Macrophyte cover	9.1	**12.3**	**8.4**	3.4	**4.6**	3.1
Large woody debris cover	2.5	5.4	2.7	1.7	2.5	1.1
Small woody debris cover	5.5	7.7	2.0	1.2	**3.5**	1.0
Overhanging vegetation cover	6.9	4.2	**7.1**	1.6	**3.4**	**4.2**
Undercut bank cover	**2.4**	**4.1**	**3.6**	1.5	**1.6**	**1.3**
Canopy cover	4.1	3.7	2.5	4.8	**4.6**	**4.2**
Mean depth	**5.1**	**11.2**	**3.8**	2.6	**4.0**	**5.9**
Mean width	**8.5**	**7.3**	**9.8**	**4.6**	**3.3**	1.8
Discharge	-	-	**11.0**	**4.3**	2.6	2.7
Temperature	2.7	3.7	1.3	1.6	2.0	1.1
Dissolved oxygen	3.0	**2.4**	2.2	**5.7**	**2.2**	0.8
Conductivity	**7.5**	**11.6**	3.2	**4.0**	**7.4**	**2.3**
Turbidity	3.2	5.9	3.3	3.0	0.8	0.2
Fine substrate	**13.0**	6.8	**1.7**	**2.0**	**7.6**	**0.6**
Sand substrate	**10.1**	**3.5**	-	-	-	-
Fine gravel substrate	-	-	**1.8**	3.9	-	**2.9**
Course gravel substrate	-	-	0.6	2.3	1.4	1.2
Cobble substrate	1.6	**3.1**	0.6	**8.0**	**1.5**	**2.6**
Adjacent rowcrop landuse	1.1	1.3	0.5	0.5	0.5	0.2
Adjacent rangeland landuse	**2.4**	9.4	1.4	**2.8**	**5.2**	**2.9**
Rip-rap influence	1.4	**7.9**	2.0	**10.6**	**4.0**	**2.2**
Eigenvalues						
Axis 1	0.75	0.76	0.20	0.14	0.81	0.11
Axis 2	0.57	0.47	0.16	0.08	0.56	0.10
Cumulative % variance explained						
Axis 1 species	15.5	17.2	17.4	17.3	9.8	10.8
Axis 1 species + environmental var.	33.4	30.9	38.0	50.2	24.8	34.9
Axis 2 species	27.2	27.8	32.1	26.3	16.6	21.1
Axis 2 species + environmental var.	58.7	49.8	69.8	76.4	41.9	68.2

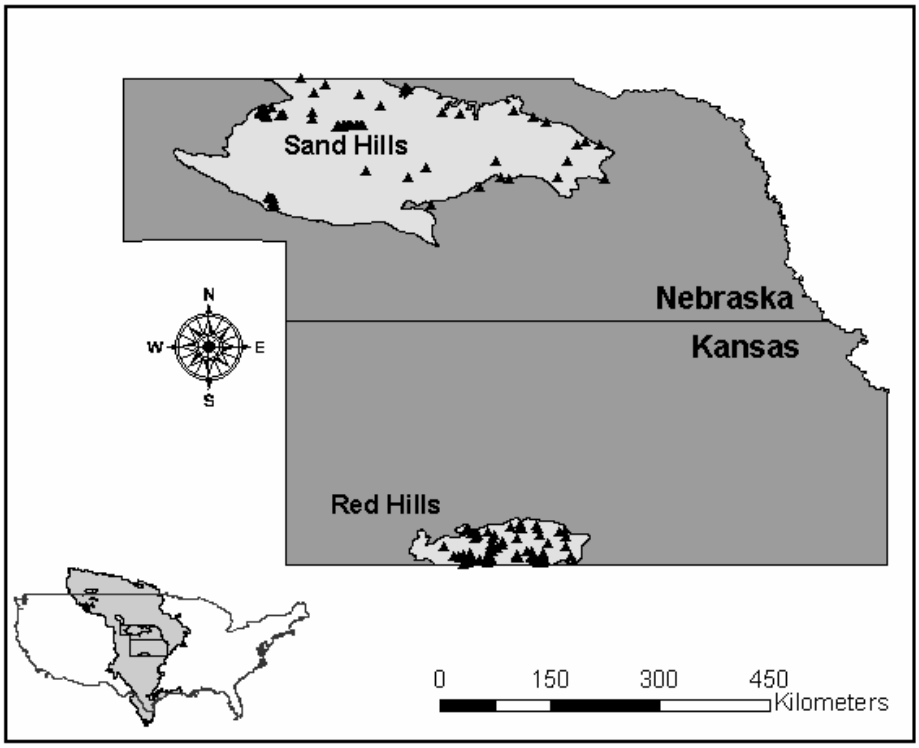

Figure 1. Locations of 67 Nebraska Sand Hills and 92 Kansas Red Hills sites (triangles) sampled from 1996-2005. Inset represents entire Great Plains region (Omernik 1987).

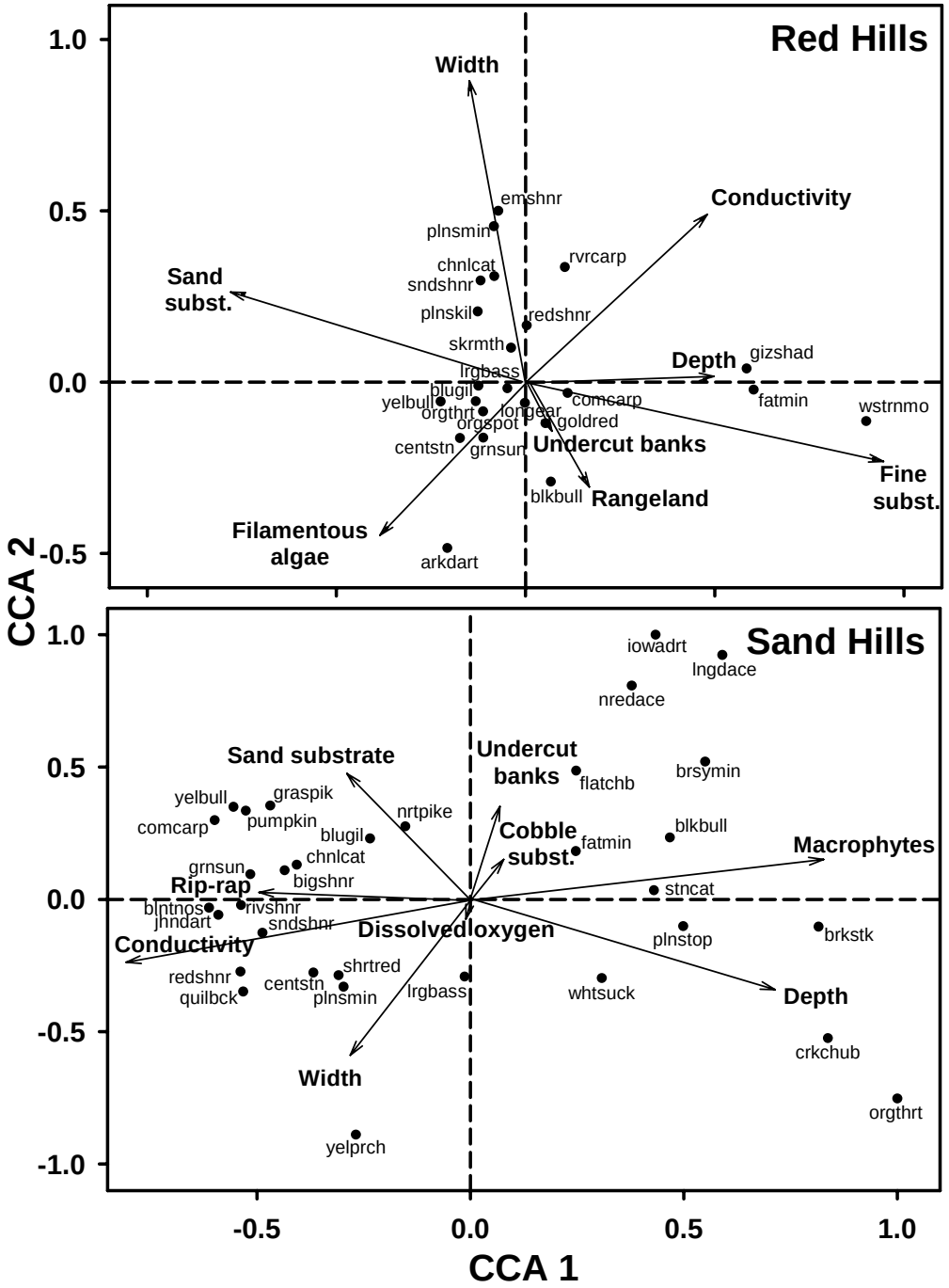

Figure 2. Species score ordination from canonical correspondence analysis (CCA) of stepwise forward selected environmental variables for fish abundance (individuals per 100 m) from 67 Sand Hills and 92 Red Hills sites. Abbreviation of fish species are listed in Table 1.

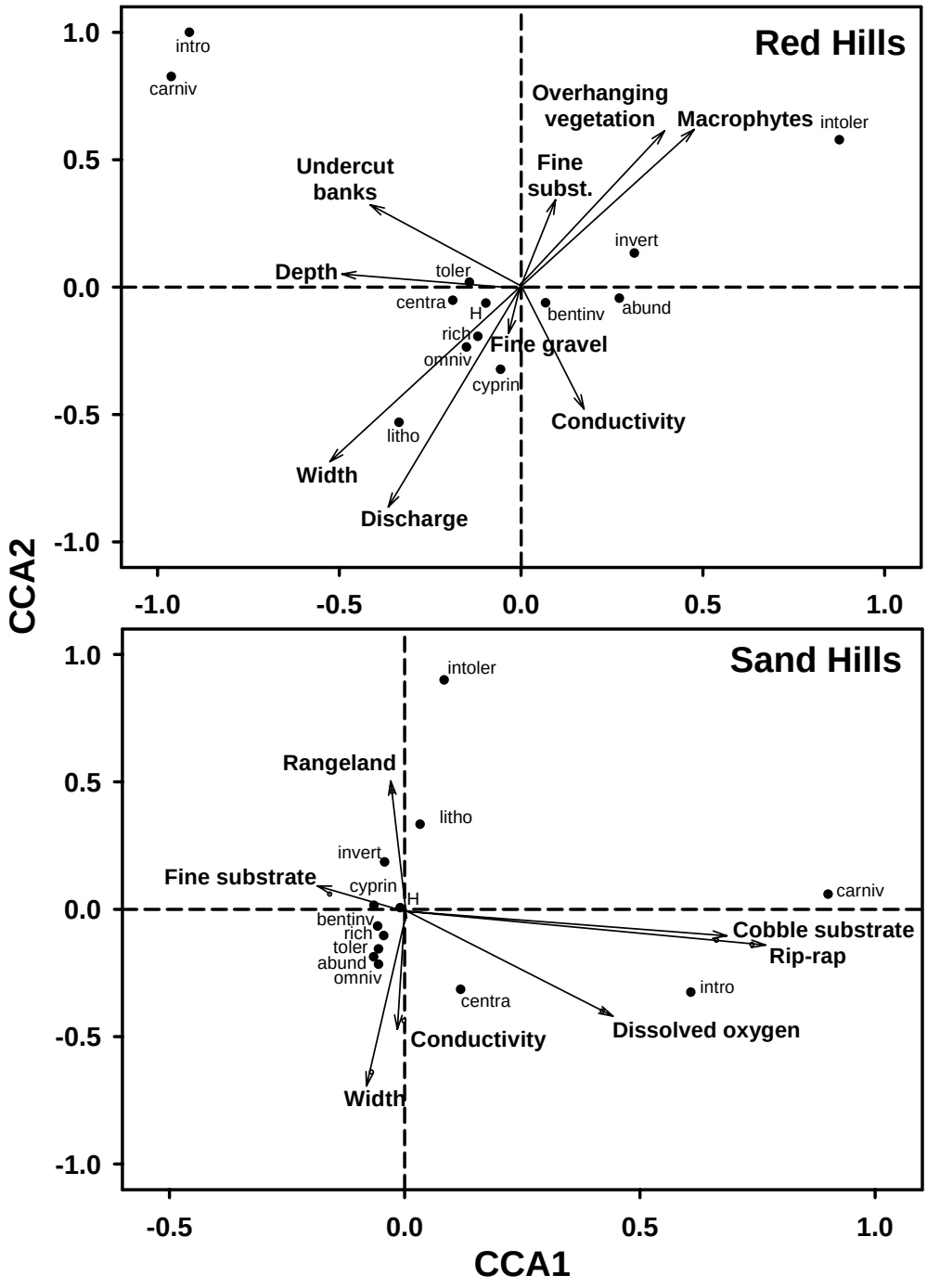

Figure 3. Species score ordination from canonical correspondence analysis (CCA) of stepwise forward selected environmental variables for fish assemblage characteristics from 67 Sand Hills and 92 Red Hills sites. Abbreviation of fish assemblage characteristics are listed in Table 2.

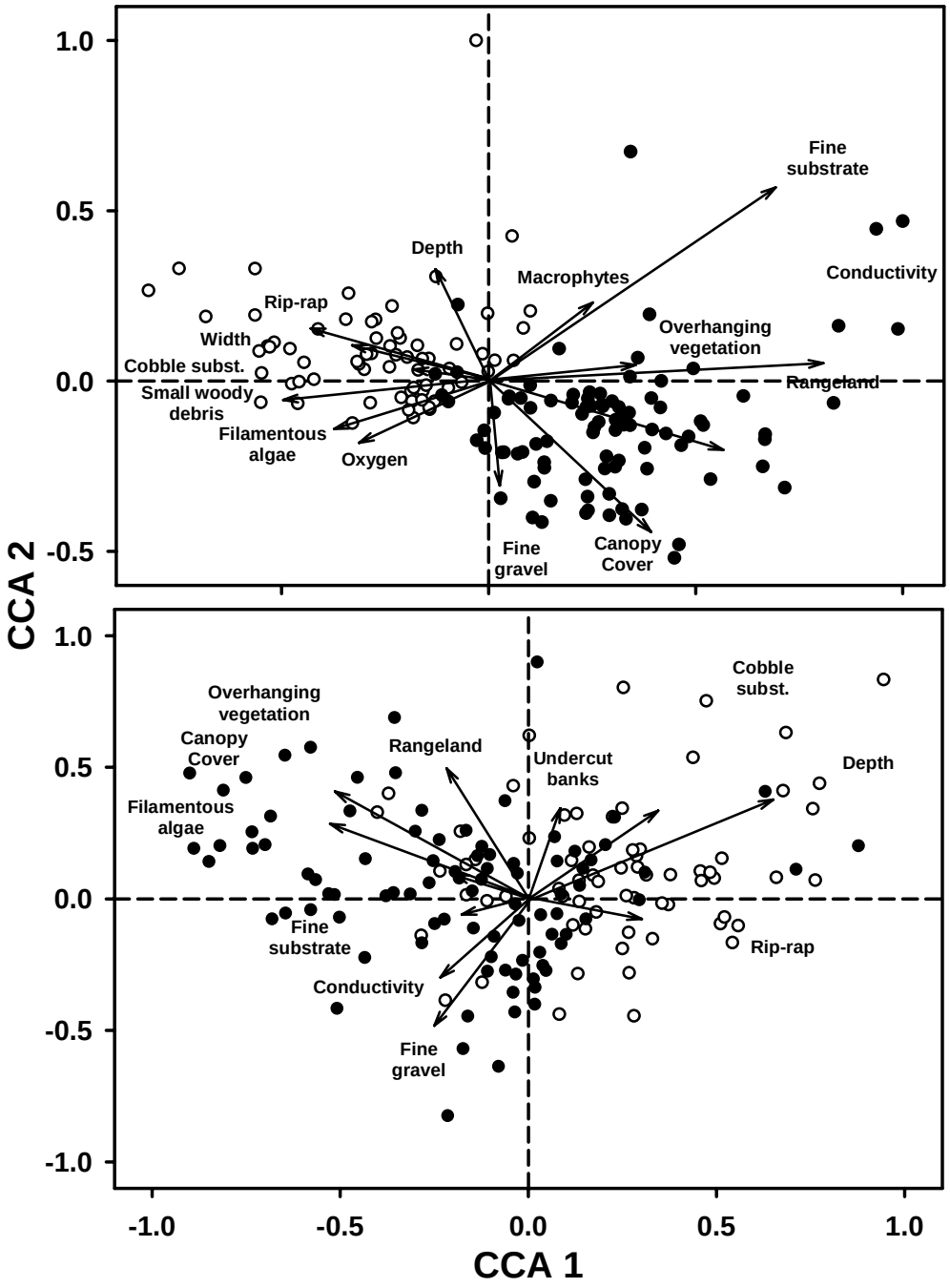

Figure 4. Sample score ordination of canonical correspondence analysis (CCA) of stepwise forward selected environmental variables for fish abundance (top) and assemblage characteristics (bottom) from 67 Sand Hils (open circles) and 92 Red Hills (closed circles) sites combined.

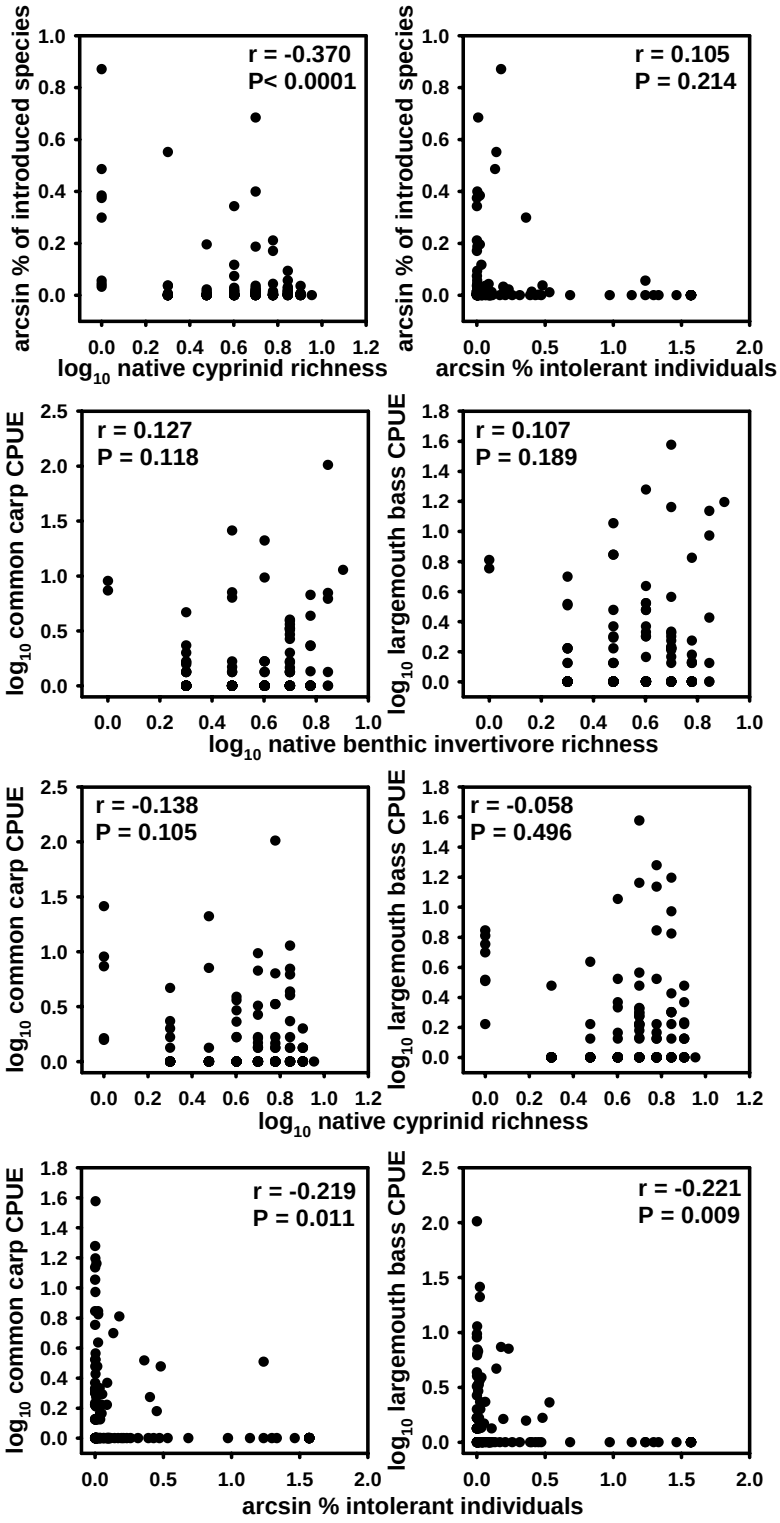

Figure 5. Correlations between assemblage characteristics for Red Hills and Sand Hills sites sampled summers 1999-2005.

Appendix A. Occurrence of fish species from 10 streams in Nebraska and Kansas Turner Properties sampled in summers 2005-2006.

Common name	Bear Creek	Blue Creek	Big Sandy Creek	Deer Creek	Horseshoe Drainage Ditch	Mud Creek	Niobrara River	North Loup River	Salt Fork Arkansas River	Snake river
Bigmouth shiner		X		X		X	X	X		
Black bullhead						X				
Bluegill							X[d]		X	
Bluntnose minnow							X[d]	X[d]	X	
Brassy minnow	X					X		X		
Brook stickleback[a]	X	X[d]		X		X	X	X		X
Brown trout		X[d]								
Central stoneroller			X				X		X	
Channel catfish								X[d]	X	
Common carp							X[d]	X		
Creek chub		X		X		X	X	X		X
Emerald shiner									X	
Fathead minnow		X		X		X	X	X	X	X
Flathead chub								X[d]		
Green sunfish		X[d]	X		X			X	X	
Green x longear sunfish			X							
Iowa darter							X	X		
Largemouth bass							X		X	
Longear sunfish			X						X	
Longnose dace		X		X		X	X	X		X
Northern pike							X[d]	X[d]		
Northern redbelly dace[b]						X		X		
Orangethroat darter[a]		X	X			X				
Plains killifish							X[d]		X	
Plains minnow[c]									X	
Plains topminnow[b]		X		X		X	X	X		X
Pumpkinseed					X					
Rainbow trout							X[d]			
Red shiner			X				X	X	X	
River carpsucker								X[d]		
River shiner						X		X		
Rock bass							X			
Sand shiner		X[d]	X				X	X[d]	X	X
Shorthead redhorse							X	X[d]		
Stonecat		X		X			X	X		
Suckermouth minnow	X								X	
Western mosquitofish			X						X	
White sucker		X		X			X	X		
Yellow bullhead									X	
Yellow perch							X			

[a]Nebraska Tier II At-risk species
[b]Nebraska Tier 1 At-risk species
[c]Kansas Species In Need of Conservation (SINC)
[d]Collected in summer of 2006

Appendix B. Catch per unit effort (fish/min of **electrofishing**) by species sampled from streams within Turner Properties for May through October 2005. Number in parenthesis represents standard error. N = number of sites.

			Species											
Ranch-Stream	N	Mean time electrofished (min)	Bigmouth shiner		Black bullhead		Bluegill		Bluntnose minnow		Brassy minnow		Brook stickleback	
Blue Creek Ranch														
Blue Creek	8	45.1 (3.8)	0.01	(0.01)	0	(0)	0	(0)	0	(0)	0	(0)	0.64	(0.81)
Deer Creek Ranch														
Deer Creek	5	24.9 (2.1)	0.60	(0.78)	0	(0)	0	(0)	0	(0)	0	(0)	0.02	(0.05)
Niobrara River	2	46.7 (18.8)	1.46	(1.4)	0	(0)	0	(0)	0	(0)	0	(0)	0.06	(0.04)
Snake River	3	16.8 (1.8)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0.09	(0.15)
McGinley Ranch														
Bear Creek	3	15.7 (1.4)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0.19	(0.12)
Horseshoe Drainage Ditch[a]	3	20.0 (0.2)	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])
Spikebox Ranch														
North Loup River	10	41.4 (12.3)	0.37	(0.63)	0.04	(0.07)	0	(0)	0	(0)	0.53	(1.09)	0.04	(0.08)
Mud Creek	4	27.1 (3.1)	0.05	(0.07)	0.03	(0.04)	0	(0)	0	(0)	0	(0)	0.16	(0.26)
Z-Bar Ranch														
Big Sandy Creek	1[b]	13.8 ([b])	0	([b])	0	([b])	0	([b])	0	([b])	0	([b])	0	([b])
Salt Fork Arkansas River	7	29.5 (7.6)	0	(0)	0	(0)	0.20	(0.14)	0.04	(0.1)	0	(0)	0	(0)

[a] No fish collected.

[b] Only one site electrofished due to high conductivity

Appendix B. Continued (see page 76 for heading)

Ranch-Stream	Species											
	Central stoneroller		Channel catfish		Common carp		Creek chub		Emerald shiner		Fathead minnow		Green sunfish	
Blue Creek Ranch												
Blue Creek	0	(0)	0	(0)	0	(0)	14.85	(5.88)	0	(0)	0.05	(0.11)	0.01	(0.01)
Deer Creek Ranch												
Deer Creek	0	(0)	0	(0)	0	(0)	0.26	(0.23)	0	(0)	0.40	(0.52)	0	(0)
Niobrara River	0.02	(0.02)	0	(0)	0	(0)	1.11	(0.93)	0	(0)	0.70	(0.57)	0	(0)
Snake River	0	(0)	0	(0)	0	(0)	0.55	(0.95)	0	(0)	1.34	(1.3)	0	(0)
McGinley Ranch												
Bear Creek	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0.61	(0.94)	0	(0)
Horseshoe Drainage Ditch[a]	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])
Spikebox Ranch												
North Loup River	0	(0)	0	(0)	0.01	(0.01)	0.71	(1.33)	0	(0)	0.09	(0.27)	0.01	(0.02)
Mud Creek	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0.69	(1.26)	0	(0)
Z-Bar Ranch												
Big Sandy Creek	0.80	([b])	0	([b])	0	([b])	0	([b])	0	([b])	0	([b])	0.29	([b])
Salt Fork Arkansas River	0.20	(0.31)	0.03	(0.041)	0	(0)	0	(0)	0.21	(0.14)	0.01	(0.03)	0.23	(0.15)

Appendix B. Continued (see page 76 for heading)

Ranch-Stream	Species											
	Iowa darter		Largemouth bass		Longear sunfish		Longnose dace		Northern redbelly dace		Orangethroat darter		Plains killifish	
Blue Creek Ranch														
Blue Creek	0	(0)	0	(0)	0	(0)	0.10	(0.12)	0	(0)	5.10	(1.34)	0	(0)
Deer Creek Ranch														
Deer Creek	0	(0)	0	(0)	0	(0)	9.50	(10.5)	0	(0)	0	(0)	0	(0)
Niobrara River	0.01	(0.02)	0.01	(0.012)	0	(0)	0.20	(0.16)	0	(0)	0	(0)	0	(0)
Snake River	0	(0)	0	(0)	0	(0)	0	(0)	0.84	(0.73)	0	(0)	0	(0)
McGinley Ranch														
Bear Creek	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Horseshoe Drainage Ditch[a]	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])
Spikebox Ranch														
North Loup River	0.01	(0.02)	0	(0)	0	(0)	2.66	(3.15)	0.78	(1.75)	0	(0)	0	(0)
Mud Creek	0	(0)	0	(0)	0	(0)	0.92	(1.07)	0.13	(0.13)	0.08	(0.11)	0	(0)
Z-Bar Ranch														
Big Sandy Creek	0	([b])	0	([b])	0	([b])	0	([b])	0	([b])	0	([b])	1.59	([b])
Salt Fork Arkansas River	0	(0)	0.09	(0.06)	0.08	(0.07)	0	(0)	0	(0)	0	(0)	2.08	(1.15)

Appendix B. Continued (see page 76 for heading)

Ranch-Stream	Species													
	Plains minnow		Plains topminnow		Red shiner		Shorthead redhorse		Stonecat		Suckermouth minnow		Western mosquitofish	
Blue Creek Ranch														
Blue Creek	0	(0)	0.50	(0.36)	0	(0)	0	(0)	0.06	(0.07)	0	(0)	0	(0)
Deer Creek Ranch														
Deer Creek	0	(0)	0.05	(0.1)	0	(0)	0	(0)	0.11	(0.09)	0	(0)	0	(0)
Niobrara River	0	(0)	0.69	(0.34)	0.99	(1.14)	0.47	(0.1)	0.06	(0.)	0	(0)	0	(0)
Snake River	0	(0)	1.10	(0.71)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
McGinley Ranch														
Bear Creek	0	(0)	0.02	(0.03)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Horseshoe Drainage Ditch[a]	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])
Spikebox Ranch														
North Loup River	0	(0)	0.43	(0.46)	0	(0)	0	(0)	0.06	(0.09)	0	(0)	0	(0)
Mud Creek	0	(0)	0.24	(0.39)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Z-Bar Ranch														
Big Sandy Creek	0	([b])	0	([b])	0.51	([b])	0	([b])	0	([b])	0	([b])	0	([b])
Salt Fork Arkansas River	3.65	(1.49)	0	(0)	0.84	(0.38)	0	(0)	0	(0)	0.03	(0.03)	0.16	(0.05)

Appendix B. Continued (see page 76 for heading)

Ranch-Stream	Species											
	White sucker		Yellow bullhead		Yellow perch		River shiner		Rock bass		Sand shiner	
Blue Creek Ranch												
Blue Creek	0.55	(0.34)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Deer Creek Ranch												
Deer Creek	0.07	(0.1)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Niobrara River	2.11	(1.)	0	(0)	0.36	(0.04)	0	(0)	0.01	(0.01)	5.80	(5.88)
Snake River	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0.02	(0.04)
McGinley Ranch												
Bear Creek	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Horseshoe Drainage Ditch[a]	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])
Spikebox Ranch												
North Loup River	0.29	(0.16)	0	(0)	0	(0)	0.02	(0.07)	0	(0)	0.02	(0.05)
Mud Creek	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Z-Bar Ranch												
Big Sandy Creek	0	([b])	0	([b])	0	([b])	0	([b])	0	([b])	1.59	([b])
Salt Fork Arkansas River	0	(0)	0.02	(0.02)	0	(0)	0	(0)	0	(0)	6.14	(5.12)

Appendix C. Catch per unit effort (fish/min of **seining**) by species sampled from streams within Turner Properties for May through October 2005. Number in parenthesis represents standard error. N = number of sites.

				Species												
Ranch-Stream	N	Mean time seined (min)		Bigmouth shiner		Black bullhead		Bluegill		Brassy minnow		Brook stickleback		Central stoneroller	
Blue Creek Ranch																
Blue Creek	8	7.38	(0.55)	0.03	(0.09)	0	(0)	0	(0)	0	(0)	0.02	(0.05)	0	(0)
Deer Creek Ranch																
Deer Creek	3[b]	5.66	(0.8)	11.14	(8.78)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Niobrara River	2	5.04	(0.48)	10.89	(2.46)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Snake River	2[b]	4.79	(1.54)	0	(0)	0	(0)	0	(0)	0	(0)	0.22	(0.07)	0	(0)
McGinley Ranch																
Bear Creek[a]	0[a]	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])
Horseshoe Drainage Ditch	3	8.57	(3.37)	0	(0)	0.07	(0.06)	0.09	(0.09)	0	(0)	0	(0)	0	(0)
Spikebox Ranch																
North Loup River	4[b]	5.88	(2.19)	6.09	(4.09)	0	(0)	0	(0)	0.20	(0.41)	0	(0)	0	(0)
Mud Creek	3[b]	4.70	(0.17)	9.90	(10.96)	0	(0)	0	(0)	0.22	(0.22)	0	(0)	0	(0)
Z-Bar Ranch																
Big Sandy Creek	4	5.30	(0.37)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	22.97	(16.38)
Salt Fork Arkansas River	7	6.81	(1.08)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0.05	(0.08)

[a] No collected.

[b] Due to extensive instream cover not all sites were seinable. See appendix A for number of sites electrofished

Appendix C. Continued (see page 81 for heading)

Ranch-Stream	Species											
	Creek chub		Emerald shiner		Fathead minnow		Grass pickerel		Green sunfish		Largemouth bass		Longear sunfish	
Blue Creek Ranch												
Blue Creek	0.71	(0.65)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Deer Creek Ranch												
Deer Creek	0.07	(0.12)	0	(0)	0.18	(0.17)	0	(0)	0	(0)	0	(0)	0	(0)
Niobrara River	0	(0)	0	(0)	0.11	(0.15)	0	(0)	0	(0)	0	(0)	0	(0)
Snake River	0.36	(0.26)	0	(0)	3.12	(3.27)	0	(0)	0	(0)	0	(0)	0	(0)
McGinley Ranch												
Bear Creek[a]	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])
Horseshoe Drainage Ditch	0	(0)	0	(0)	0	(0)	0.08	(0.15)	0.07	(0.06)	0	(0)	0	(0)
Spikebox Ranch												
North Loup River	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Mud Creek	0.42	(0.72)	0	(0)	0.91	(1.57)	0	(0)	0	(0)	0	(0)	0	(0)
Z-Bar Ranch												
Big Sandy Creek	0	(0)	0	(0)	0	(0)	0	(0)	2.56	(3.26)	0.14	(0.19)	2.13	(3.22)
Salt Fork Arkansas River	0	(0)	1.02	(0.99)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)

Appendix C. Continued (see page 81 for heading)

Ranch-Stream	Species													
	Longnose dace		Northern pike		Orangethroat darter		Plains killifish		Plains minnow		Plains topminnow		Pumpkinseed	
Blue Creek Ranch														
Blue Creek	0.46	(0.63)	0	(0)	0.05	(0.07)	0	(0)	0	(0)	0	(0)	0	(0)
Deer Creek Ranch														
Deer Creek	19.36	(7.76)	0	(0)	0	(0)	0	(0)	0	(0)	0.13	(0.11)	0	(0)
Niobrara River	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0.11	(0.15)	0	(0)
Snake River	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0.58	(0.33)	0	(0)
McGinley Ranch														
Bear Creek[a]	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])
Horseshoe Drainage Ditch	0	(0)	0.03	(0.05)	0	(0)	0	(0)	0	(0)	0	(0)	0.08	(0.15)
Spikebox Ranch														
North Loup River	2.91	(2.97)	0	(0)	0	(0)	0	(0)	0	(0)	0.14	(0.27)	0	(0)
Mud Creek	12.78	(11.95)	0	(0)	0.07	(0.12)	0	(0)	0	(0)	0	(0)	0	(0)
Z-Bar Ranch														
Big Sandy Creek	0	(0)	0	(0)	0.38	(0.54)	19.81	(22.65)	0	(0)	0	(0)	0	(0)
Salt Fork Arkansas River	0	(0)	0	(0)	0	(0)	26.79	(8.98)	3.17	(5.18)	0	(0)	0	(0)

Appendix C. Continued (see page 81 for heading)

Ranch-Stream	Species													
	Red shiner		River shiner		Sand shiner		Stonecat		Longear x Green Sunfish		Western mosquitofish		White sucker	
Blue Creek Ranch														
Blue Creek	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0.07	(0.1)
Deer Creek Ranch														
Deer Creek	0	(0)	0	(0)	0	(0)	0.12	(0.11)	0	(0)	0	(0)	0	(0)
Niobrara River	2.47	(1.92)	0	(0)	6.99	(5.16)	0	(0)	0	(0)	0	(0)	0	(0)
Snake River	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
McGinley Ranch														
Bear Creek[a]	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])	0	([a])
Horseshoe Drainage Ditch	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Spikebox Ranch														
North Loup River	1.48	(2.96)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0.10	(0.12)
Mud Creek	0	(0)	0.22	(0.22)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Z-Bar Ranch														
Big Sandy Creek	2.76	(1.65)	0	(0)	20.48	(23.12)	0	(0)	0.05	(0.097)	1.10	(1.26)	0	(0)
Salt Fork Arkansas River	3.05	(2.63)	0	(0)	27.06	(11.17)	0	(0)	0	(0)	0.08	(0.1)	0	(0)

Appendix D. Mean **physical habitat** and **cover** characteristics for streams within Turner Properties sampled during May through October 2005. Number in parenthesis represents standard error. N = number of sites

		Physiochemical variables													
Ranch-Stream	N	Mean depth (cm)		Mean width (m)		Discharge (m3/s)		Filamentous algae (%)		Macrophytes (%)		Large woody debris (%)		Brush/small woody debris (%)	
Blue Creek Ranch															
Blue Creek	8	44.7	(4.5)	9.8	(1.9)	2.01	(0.17)	18.1	(21.4)	50.0	(0)	0	(0)	1.0	(2.3)
Deer Creek Ranch															
Deer Creek	5	18.2	(5.4)	2.6	(0.9)	0.15	(0.1)	0.0	(0)	36.7	(22.8)	1.3	(3.)	6.0	(13.4)
Niobrara River	2	14.8	(3.4)	29.8	(13.7)	2.33	(0.45)	45.0	(0)	17.5	(10.6)	3.3	(2.4)	5.0	(4.7)
Snake River	3	14.0	(5.9)	1.7	(0.5)	0.04	(0.01)	0.0	(0)	28.3	(20.2)	0	0	2.2	(1.9)
McGinley Ranch															
Bear Creek	3	20.3	(2.4)	1.4	(0.2)	0.02	(0.01)	14.4	(13.6)	36.1	(42.8)	0	(0)	0	(0)
Horseshoe Drainage Ditch	3	34.2	(4.)	3.9	(2.4)	0.23	(0.07)	41.7	(38.2)	46.7	(39.4)	0	(0)	0	(0)
Spikebox Ranch															
North Loup River	10	30.5	(7.5)	8.2	(4.8)	1.23	(0.75)	3.8	(3.9)	40.0	(13.1)	0	(0)	5.3	(9.6)
Mud Creek	4	32.9	(8.8)	3.2	(0.4)	0.32	(0.15)	34.2	(35.2)	43.5	(20.2)	0	(0)	18.8	(23.9)
Z-Bar Ranch															
Big Sandy Creek	4	15.9	(9.3)	3.0	(1.1)	0.02	(0.01)	42.1	(15.8)	32.1	(30.9)	0.8	(1.)	5.0	(2.7)
Salt Fork Arkansas River	7	5.3	(0.8)	20.0	(5.7)	0.32	(0.03)	0.0	(0)	0	(0)	0.2	(1.7)	1.0	(7.9)

Appendix D. Continued (see page 85 for heading)

Ranch-Stream	Physiochemical variables															
	Overhanging vegitation (%)		Undercut banks (%)		Boulders (%)		Temperature (C^o)		Dissolved oxygen (mg/l)		Specific conductance (umhos/cm)		Turbidity (NTUs)	
Blue Creek Ranch														
Blue Creek	12.5	(7.1)	1.9	(1.4)	0	(0)	20.35	(2.7)	6.3	(0.7)	157.9	(4.0)	11.7	(2.3)
Deer Creek Ranch														
Deer Creek	27.0	(17.9)	8.0	(1.8)	0	(0)	20.52	(3.5)	6.6	(0.5)	119.5	(1.5)	13.7	(6.2)
Niobrara River	10.0	(0)	0	(0)	0.8	(1.2)	16.60	(7.2)	3.6	(0.5)	262.1	(11.9)	6.2	(2.2)
Snake River	25.0	(13.23)	1.7	(2.9)	0	(0)	24.83	(4.1)	4.2	(1.9)	287.8	(28.8)	6.9	(2.4)
McGinley Ranch														
Bear Creek	47.8	(14.4)	0.0	(0)	0	(0)	19.10	(5.7)	6.0	(0.9)	125.0	(2.6)	3.3	(0.95)
Horseshoe Drainage Ditch	23.9	(24.1)	11.7	(16.1)	0	(0)	17.00	(2.9)	6.0	(3.04)	403.3	(23.4)	4.6	(1.5)
Spikebox Ranch														
North Loup River	14.7	(8.4)	2.0	(2.3)	5.8	(7.7)	21.54	(2.8)	3.2	(1.5)	173.0	(6.3)	4.9	(1.4)
Mud Creek	38.3	(27.3)	1.3	(2.5)	0	(0)	24.78	(1.9)	4.3	(1.9)	238.5	(2.4)	9.1	(4.6)
Z-Bar Ranch														
Big Sandy Creek	14.2	(10.7)	8.8	(14.2)	0	(0)	23.48	(4.7)	2.7	(2.3)	2610.8	(349.8)	1.9	(0.4)
Salt Fork Arkansas River	1.1	(14.3)	0.1	(1.3)	0	(0)	28.39	(4.8)	3.5	(1.6)	3328.9	(224.0)	14.4	(3.7)

Appendix D. Continued (see page 85 for heading)

Ranch-Stream	Physiochemical variables									
	Alkalinity (mg/l CaCO3)		Chloride (mg/L)		Nitrate (mg/L)		Ammonia (mg/L)		Phosphate (mg/L)	
Blue Creek Ranch										
Blue Creek	72.0	(3.1)	0.7	(0.3)	1.2	(0.2)	0.01	(0.02)	0.7	(0.05)
Deer Creek Ranch										
Deer Creek	55.2	(1.9)	0.3	(0.2)	0.6	(0.3)	0.1	(0.03)	1.3	(0.3)
Niobrara River	122.5	(2.1)	1.4	(0.8)	1.0	(0.1)	0.1	(0.05)	0.7	(0.3)
Snake River	145.7	(13.3)	1.9	(0.4)	0.5	(0.2)	0.3	(0.1)	1.0	(0.5)
McGinley Ranch										
Bear Creek	67.1	(2.4)	0.4	(0.1)	0.5	(0.3)	0.2	(0.1)	1.2	(0.3)
Horseshoe Drainage Ditch	190.3	(22.3)	1.2	(0.9)	1.4	(0.4)	0.3	(0.1)	0.6	(0.2)
Spikebox Ranch										
North Loup River	79.7	(5.1)	0.7	(0.3)	0.6	(0.3)	0.1	(0.03)	0.8	(0.1)
Mud Creek	108.9	(3.7)	1.2	(0.2)	0.8	(0.3)	0.2	(0.05)	1.2	(0.3)
Z-Bar Ranch										
Big Sandy Creek	171.5	(24.6)	5.0	(1.3)	0.5	(0.4)	2.7	(0.03)	0.2	(0.03)
Salt Fork Arkansas River	136.1	(6.4)	24.5	(0)	0.6	(0.3)	0.05	(0.04)	0.3	(0.2)

Appendix E. Sampling locations within Turner Properties sampled during summers 2005-2006.

Ranch-Stream	Site #	Latitude (decimal degrees)	Longitude (decimal degrees)
Blue Creek Ranch			
Blue Creek	1	41.42850	-102.16725
Blue Creek	2	41.43999	-102.17196
Blue Creek	3	41.45701	-102.17965
Blue Creek	4	41.47143	-102.18512
Blue Creek	5	41.48082	-102.18748
Blue Creek	6	41.50446	-102.18954
Blue Creek	7	41.51606	-102.20811
Blue Creek	8	41.52234	-102.22738
Deer Creek Ranch			
Deer Creek	1	42.56213	-102.33984
Deer Creek	2	42.55253	-102.31208
Deer Creek	3	42.54514	-102.28751
Deer Creek	4	42.53431	-102.26665
Deer Creek	5	42.50888	-102.27018
Niobrara River	1	42.58422	102.34899
Niobrara River	2	42.62695	-102.24730
Niobrara River	3	42.61998	-102.31480
Niobrara River	4	42.58575	-102.34830
Niobrara River	5	42.62690	102.28306
Niobrara River	6	42.61357	102.32922
Niobrara River	7	42.59290	102.34200
Snake River	1	42.55031	-102.06680
Snake River	2	42.54383	-102.08786
Snake River	3	42.54077	-102.10466
McGinley Ranch			
Bear Creek	1	42.97534	-101.81908
Bear Creek	2	42.97940	-101.82707
Bear Creek	3	42.99044	-101.83793
Horseshoe Drainage Ditch	1	42.92189	-101.91360
Horseshoe Drainage Ditch	2	42.92935	-101.94229
Horseshoe Drainage Ditch	3	42.92495	-101.97593

Appendix E. Continued (see page 85 for heading).

Ranch-Stream	Site #	Latitude (decimal degrees)	Longitude (decimal degrees)
Spikebox Ranch			
North Loup River	1	42.41029	-101.07202
North Loup River	2	42.41167	-101.10199
North Loup River	3	42.41398	-101.13046
North Loup River	4	42.41664	-101.16721
North Loup River	5	42.41933	-101.21931
North Loup River	6	42.41855	-101.23115
North Loup River	7	42.41452	-101.26673
North Loup River	8	42.41388	-101.29194
North Loup River	9	42.40313	-101.31271
North Loup River	10	42.40178	-101.40088
Mud Creek	1	42.41673	-101.30149
Mud Creek	2	42.41867	-101.32755
Mud Creek	3	42.42260	-101.37782
Mud Creek	4	42.41838	-101.39080
Z-Bar Ranch			
Big Sandy Creek	1	37.06904	-98.85984
Big Sandy Creek	2	37.09225	-98.84985
Big Sandy Creek	3	37.11133	-98.85211
Big Sandy Creek	4	37.13127	-98.85034
Salt Fork Arkansas River	1	37.01656	-98.88445
Salt Fork Arkansas River	2	37.03517	-98.88847
Salt Fork Arkansas River	3	37.02579	-98.90923
Salt Fork Arkansas River	4	37.05488	-98.94315
Salt Fork Arkansas River	5	37.07023	-98.96430
Salt Fork Arkansas River	6	37.07361	-98.97897
Salt Fork Arkansas River	7	37.08654	-98.99403