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Historical Changes in Nebraska's Lotic Fish Assemblages: Implications of Anthropogenic Alterations

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ABSTRACT.—The plains of midwestern North America have undergone significant anthropogenic alterations following European settlement with consequent effects to lotic fish assemblage structure. We examined trends in fish assemblage structure and function in Nebraska's lotic systems using site-specific, presence-absence data from historical (1939–1940) and contemporary surveys (2003–2005; n = 183). Shifts in fish assemblage structure were characterized by declines of specialist species (*e.g.,* western silvery minnow *Hybognathus argyritis*) and increases in nonnative, sport, and generalist species (*e.g.,* common carp *Cyprinus carpio*). Our research illustrates differences between historical and contemporary surveys for both taxonomic and functional metrics. Changes in fish assemblage structure were correlated with a contemporary measure of anthropogenic alteration (Human Threat Index; HTI) and were most pronounced for large-scale threats (*i.e.*, watershed HTI, overall HTI). The HTI is a composite index of cumulative anthropogenic alterations experienced by a stream system and was used to investigate broad-scale implications of anthropogenic activity on fish assemblage structure. Fish assemblages among sites were more similar in contemporary surveys than in historical surveys, such changes might indicate a homogenization of the fish assemblages. Losses of native species and increases in introduced species have occurred in Nebraska's lotic systems across a broad temporal span and shifts are likely related to high levels of human perturbation.

INTRODUCTION

North America harbors one of the most diverse temperate freshwater fish assemblages in the world with estimates of over 1200 species (Briggs, 1986; Abell *et al.*, 2000). However, approximately 40% of fishes in North America are listed as endangered, threatened, or imperiled, with recent estimates of 61 species considered extinct or extirpated from their natural habitats (Jelks *et al.*, 2008). Despite high contemporary rates of fish imperilment, future extinction rates for fish and freshwater fauna are projected to further increase by 500% and 640%, respectively (Ricciardi and Rasmussen, 1999). Causes for freshwater fish imperilment in the United States have been reported as, but not limited to, water diversion (percentage of listed species whose decline was caused by a respective alteration; 73%),

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invasive species (49%), pollution (49%), and agricultural activities (41%; Miller-Reed and Czech, 2005; Hoagstrom *et al.*, 2011).

The loss of any native species from an ecosystem can have detrimental impacts on ecosystem stability and function. Increased native species biodiversity may provide important ecological services to systems (*e.g.,* greater productivity, increased biotic resistance, storage of genetic information, improved water quality; Hooper *et al.*, 2005; Balvanera *et al.*, 2006; Cardinale *et al.*, 2011). For example the decline of native marine biodiversity appears to coincide with an increased rate of fisheries collapse and reduced water quality, fisheries instability, and loss of recovery potential of marine fisheries (Worm *et al.*, 2006). Unfortunately, losses of freshwater biodiversity have been widely reported (Dudgeon *et al.*, 2006; Pittock *et al.*, 2008; Dudgeon, 2010) and largely appear to be a result of widespread anthropogenic alterations to aquatic systems (Allan and Flecker, 1993; Malmqvist and Rundle, 2002; Mooney *et al.*, 2009).

Stream systems generally harbor high habitat and biotic diversity relative to lentic systems (Pringle *et al.*, 1988). Unfortunately, many stream systems have been severely degraded (Allan and Flecker, 1993) and have experienced disproportionate losses of biodiversity compared to terrestrial systems (Jenkins, 2003). High levels of stream degradation have been attributed to their position in the landscape as a receiver of the cumulative anthropogenic disturbances in a watershed (Williamson *et al.*, 2008). Furthermore, declines in stream quality appear to be driven by human population growth and economic activities (Allan, 2004; King *et al.*, 2011; Limburg *et al.*, 2011). As a result, lotic systems have been declared one of the most anthropogenically-impacted ecosystems on the planet (Malmqvist and Rundle, 2002). In the United States, over a third of the stream length evaluated was considered impaired using various indices of biological integrity (IBI; *e.g.,* fish, amphibian; Paulsen *et al.*, 2008; Pont *et al.*, 2009). However, stream degradation is not consistent across regions in the United States because of differing anthropogenic factors.

The Interior Plains physiographic region (hereafter termed plains) of midwestern North America was historically a vast grassland complex but has suffered widespread land use alteration following European settlement (Matthews, 1988; Ricketts *et al.*, 1999). The plains are highly productive, with deep fertile soils, diverse flora and fauna communities, and a climate appropriate for many temperate grasses and cultivated crops (Samson and Knopf, 1994; Peterson and Cole, 1995). As a result of high system productivity, a majority of available land in the region was converted to agricultural production following European settlement (Peterson and Cole, 1995). For instance, 47% of available land in Kansas was converted from native vegetation to row crop agriculture across a 62 y period (Gido *et al.*, 2010). The plains are currently one of the most threatened ecosystems in North America; individual state or province losses in excess of 99% have been reported for the Great Plains (Samson and Knopf, 1994; Ricketts *et al.*, 1999; Samson *et al.*, 2004). Loss of remaining prairie systems continue, with over 90,000 km^2 lost in the Great Plains alone between 1982 and 1997 (Samson *et al.*, 2004). Lotic ecosystems in the Interior Plains have also experienced widespread alteration and degradation (Dodds *et al.*, 2004; Hoagstrom *et al.*, 2006; Paulsen *et al.*, 2006).

The plains region contains relatively high biodiversity. In the United States, approximately 26% of all freshwater fish species are found in the Great Plains (Rabeni, 1996; Page and Burr, 2011). Native plains fishes are commonly described as tolerant species, having evolved in harsh physicochemical conditions (*e.g.,* high temperature, low dissolved oxygen, variable turbidity levels, stream intermittency) and frequent disturbance (*e.g.,* drought, flooding; Matthews, 1988; Fausch and Bestgen, 1997; Dodds *et al.*, 2004). Unfortunately, several native cyprinids (*e.g., Hybognathus, Macrhybopsis, Notropis, Platygobio*) have experienced

declines in abundance and historical distributions throughout the plains region (Luttrell *et al.*, 1999; Fisher *et al.*, 2002; Quist *et al.*, 2004).

The goal of this study was to evaluate patterns of change from historical to contemporary fish assemblages at two spatial scales (ecoregion, statewide) using taxonomic and functional metrics of assemblage structure. Our specific objectives were to compare fish assemblage structure in Nebraska's lotic systems across a large temporal span (*i.e.*, 65 y) and determine if changes in fish assemblage composition are correlated with anthropogenic factors. We hypothesized that structural shifts in Nebraska's lotic fish assemblages have occurred across the temporal span of the study and are related to anthropogenic influences found in ecoregions with differing levels of human alteration.

METHODS

STUDY AREA

Nebraska is located in the Interior Plains physiographic region of North America. Most of the state is located in the Great Plains; the eastern quarter of the state is part of the Central Lowlands. Nebraska was historically dominated by a prairie ecosystem and has lost approximately 85% of its tallgrass and mixed-grass prairies since the middle of the 19th century (Samson and Knopf, 1994). Nebraska's land use is currently dominated by range and pasture (percent of state area; 50%) and cropland (*i.e.*, cultivated, noncultivated; 39%; Nebraska National Resources Inventory, 2007). The east-west land use gradient in Nebraska is cultivated row-crop agriculture common in eastern portions of the state, and ranching and limited row-crop production in the west (Baltensperger, 1985).

Sampling locations were distributed across all level III ecoregions of Nebraska to provide both regional and statewide estimates of fish assemblage structure (Fig. 1). Ecoregions consist of both terrestrial and aquatic habitats that contain similar abiotic and biotic components (Omernik, 1995; Reiners, 1995; McMahon *et al.*, 2001). Due to system similarities (*e.g.,* climate, hydrology, species composition) ecoregions are often used as spatial units for conservation efforts in terrestrial (Pike *et al.*, 1999) and aquatic systems (Van Sickle and Hughes, 2000; Dauwalter *et al.*, 2008). Use of an alternative spatial unit (hydrological unit codes) was investigated, and the taxonomic and functional trends were similar. An ecoregion approach, where spatial units were grouped based on similarities in both terrestrial and aquatic systems, is appropriate for an investigation of fish assemblage shifts in relation to land use changes. Therefore, we used level III ecoregions of Nebraska in comparisons of spatial changes in fish assemblage structure. Nebraska has six level III ecoregions: Central Great Plains (CGP), High Plains (HP), Nebraska Sandhills (NSH), Northwestern Great Plains (NWGP), Northwestern Glaciated Plains (NWGLP), and Western Corn Belt Plains (WCBP; McMahon *et al.*, 2001; Wiken *et al.*, 2011).

HUMAN THREAT INDEX

The anthropogenic impact at both ecoregion and stream segment spatial scales was estimated using the Human Threat Index (HTI). The HTI is a geospatially referenced synthesis of human threats facing aquatic ecosystems in EPA Region 7 (Diamond *et al.*, 2010). The HTI was created using 35 geospatial data layers of ''threats'' (*e.g.,* dams, channelized streams, altered land covers, waste treatment facilities) present in the region that were likely to influence the quality of aquatic systems. The index was created by quantifying and ranking individual threats for each stream segment against all other similarly sized segments in EPA Region 7 at both the local (*i.e.*, stream segment; stream segment catchment polygons were on average 2–3 km^2) and watershed scale (*i.e.*, the whole

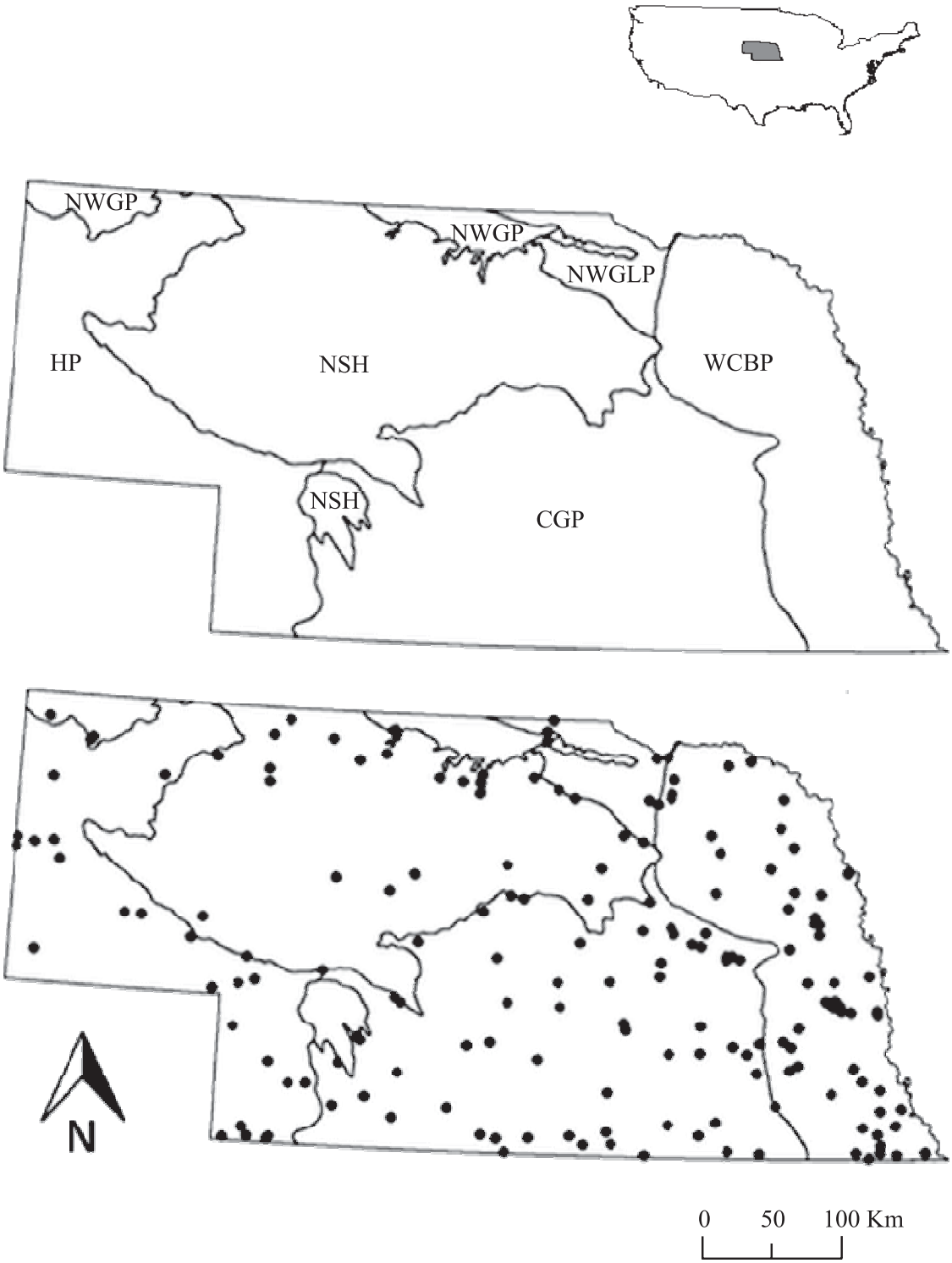

FIG. 1.—Historical and contemporary stream sampling locations across Nebraska (n = 183). Level III ecoregions of Nebraska are displayed. Nebraska's level III ecoregion acronyms include, Central Great Plains (CGP), High Plains (HP), Nebraska Sandhills (NSH), Northwestern Great Plains (NWGP), Northwestern Glaciated Plains (NWGLP), and Western Corn Belt Plains (WCBP)

upstream watershed). Ranked threats in EPA Region 7 were scaled from zero to 100, with larger values indicating increased perceived threat to stream segments. Scaled values of all individual, local-scale threats were summed and rescaled to create the local HTI (LHTI). The watershed HTI (WHTI) was created in a similar manner using watershed-scale threats. Overall HTI (OHTI) values were created by summing the local and watershed HTI values and rescaling the values from zero to 100. As part of the HTI project, every stream segment in EPA Region 7 was assigned three HTI values (*i.e.*, LHTI, WHTI, OHTI). Human Threat Index values were tested for an explanation of shifts in spatial and temporal fish assemblage structure. Detailed description of the development of the HTI is in Diamond *et al.*, (2010).

HISTORICAL VS. CONTEMPORARY COMPARISON OF NEBRASKA'S LOTIC FISH ASSEMBLAGES

Fish assemblage surveys occurred in wadeable lotic systems during two periods: historical (1939–1940) and contemporary (2003–2005). Historical data were collected at 215 wadeable sites across Nebraska by Raymond E. Johnson (University of Michigan) and predominately used seines (1.8, 3.0, 7.6 m length). Additional sampling techniques (*e.g.,* rotenone) were used occasionally to supplement seining efforts (Johnson, 1942). The primary objective of the historical surveys was to describe the distribution of Nebraska fishes, thus substantial sampling effort and a variety of techniques were used. Contemporary surveys were an attempt to resample all historical survey locations. Unfortunately, due to stream dewatering, 32 historical sites were unable to be sampled. As a result, contemporary surveys were completed at 183 historical sampling locations. Backpack and barge electrofishing techniques were used in contemporary surveys. Block nets were set upstream and downstream of sites prior to fish sampling whenever feasible. Sampling length was 40 times the mean wetted stream width, with a minimum of 150 m and maximum length of 300 m. If possible, fish were identified in the field and released. Unidentified fishes were preserved in 10% formalin and later identified in the laboratory. Vouchered specimens are stored at the University of Nebraska-Lincoln.

Due to differences in sampling techniques and gear efficiencies between historical and contemporary methods, presence-absence site data were used for statistical analysis. Only sites that were sampled as part of both historical and contemporary surveys were included in the analysis ($n = 183$). Species presence-absence data were used to construct both taxonomic and functional metrics for each time period and site. Information was compiled at ecoregion and statewide spatial scales. All species were used in analyses of historical changes in the percentage of species occurrences and species richness, as well as cluster analysis of fish assemblage structure. However, due to sensitivities associated with including rare species, only species present at more than 5% of the sites were included in the permutational multivariate analysis of variance [PERMANOVA]. Rare species (<5% of sites) were also excluded from correlation analyses using site dissimilarity, species turnover, and HTI values (Zar, 1999).

Due to the functional contributions of rare species to ecosystems, all species were included in the functional classification metrics. Functional classification categories included feeding guilds, tolerance, and reproductive guilds. In addition to traditional functional categories, metrics of species status were included to investigate trends of nonnative and sport fishes. Additional functional guilds were evaluated, but due to low species richness many functional guilds consisted of only one species and were equivalent to taxonomic analyses. Tolerance, feeding, and reproductive guilds were classified following Meador and Carlisle (2007) and Frimpong and Angermeier (2009). The generalist classification was assigned to a species considered both tolerant and omnivorous. Nonnative species were identified using historical distribution records compiled by Lee *et al.* (1981),

and sport fish species were classified using Nebraska Game and Parks Commission designations (Nebraska Game and Parks Commission, 2012a).

We calculated the proportion of sites with at least one species of each functional group present. Summed presences of individual species were compiled in a similar manner for the taxonomic analysis. The historical percentage of site occurrences was calculated for both taxonomic and functional metrics for ecoregions and for the entire state. The percentage change in occurrence rates was calculated by subtracting the historical from the contemporary percentage of sites occupied, divided by the historical percentage of occupied sites.

A paired Wilcoxon signed-rank test was used to compare differences in species richness and proportion of nonnative species from historical to contemporary surveys for ecoregions and statewide ($\alpha = 0.05$) using all 74 species. Similarity of ecoregion taxonomic structure was evaluated using a Jaccard dissimilarity measure and an unweighted pair-group clustering method (UPGMA; McCune and Grace, 2002). For the taxonomic cluster analysis, the presence or absence of each species for an individual ecoregion was used to minimize the influence of varying sample sizes among ecoregions. Cluster analysis was calculated with PC-ORD (MJM Software, Gleneden Beach, Oregon).

Differences in taxonomic and functional assemblage structure were evaluated using PERMANOVA because of advantages over a traditional multivariate analysis of variance (*e.g.,* lack of multivariate normality assumptions; Anderson, 2001). The PERMANOVA was used to model the effects of time (*e.g.,* historical versus contemporary) and ecoregion on fish assemblage composition using Jaccard's dissimilarity. Site was also included in the PERMANOVA model and because sites were revisited (*e.g.,* paired samples), permutations were restricted by site in all of the PERMANOVA analyses. The overall PERMANOVA model (site + time + ecoregion) was used to determine if the effect of ecoregion warranted separate analyses for each ecoregion. If the PERMANOVA was significant for individual ecoregions, Jaccard's dissimilarity was calculated for sites (historical versus contemporary) and the mean was estimated for taxonomic and functional datasets by ecoregion. The strength of correlation between assemblage shifts across the temporal span of the surveys and anthropogenic stressors were evaluated using Spearman's correlation coefficient between Jaccard's dissimilarity and HTI (*i.e.*, LHTI, WHTI, OHTI) of ecoregions (Zar, 1999). We used Spearman's correlation coefficient to compare species turnover and HTI. The PERMANOVA analyses used ADONIS with the Vegan package in R version 2.11 with a significance level of $\alpha = 0.05$ using 999 permutations (Oksanen *et al.*, 2011).

RESULTS

HUMAN THREAT INDEX

Human Threat Index values varied among Nebraska's level III ecoregions (Fig. 2). The CGP (mean ± SE; 35.7 ± 1.9) and WCBP ecoregions (47.9 ± 1.8) had the highest OHTI values. Nebraska Sandhills (13.4 ± 1.8) and NWGP (21.4 ± 4.3) had the lowest OHTI values of level III ecoregions of Nebraska. Trends in HTI values indicated that levels of anthropogenic alteration were not consistent across the state and the highest levels of disturbance occurred in the eastern portion of the state (*i.e.*, CGP, WCBP) with greater human population and more intensive agricultural practices.

HISTORICAL VS. CONTEMPORARY COMPARISON OF NEBRASKA'S LOTIC FISH ASSEMBLAGES

A total, of 74 species and 17 families, was detected during historical and contemporary surveys. Two federally-endangered species were sampled: Topeka shiner *Notropis topeka* and pallid sturgeon *Scaphirhynchus albus*. Additionally, two species listed as threatened or

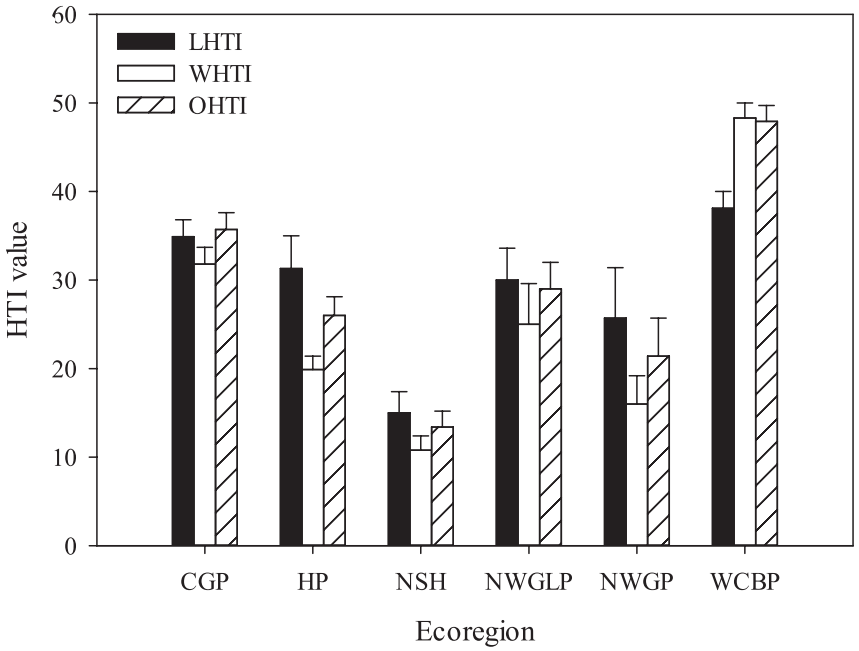

FIG. 2.—Local Human Threat Index (LHTI), watershed HTI (WHTI), and overall HTI (OHTI) values for Nebraska's level III ecoregions. Ecoregion acronyms are Central Great Plains (CGP), High Plains (HP), Nebraska Sandhills (NSH), Northwestern Great Plains (NWGP), Northwestern Glaciated Plains (NWGLP), and Western Corn Belt Plains (WCBP). Error bars represent one standard error of the mean

endangered by the State of Nebraska were also sampled: sturgeon chub *Macrhybopsis gelida* (Nebraska's conservation status; endangered) and finescale dace *Phoxinus neogaeus* (threatened). Eight species were absent from contemporary surveys and 10 species were not collected in historical samples (Table 1). Seven of the eight species absent from contemporary surveys were considered native and 40% of the species not collected in historical surveys were considered nonnative. Two species in the reduced species list, the western mosquitofish *Gambusia affinis* (contemporary percentage of site occurrences; 13%) and freshwater drum *Aplodinotus grunniens* (11%), were not found in historical samples and a change in occurrence could not be estimated.

The species with the greatest positive percentage of change from historical to contemporary surveys included shorthead redhorse *Moxostoma macrolepidotum* (historical percentage of occupied sites, percentage of change in site occurrence from historical to contemporary surveys; 4%, 443%), bluegill *Lepomis macrochirus* (5%, 410%), flathead catfish *Pylodictis olivaris* (3%, 380%), and shortnose gar *Lepisosteus platostomus* (3%, 360%; Table 2). Species with the greatest declines were western silvery minnow *Hybognathus argyritis* (11%, −60%), plains topminnow *Fundulus sciadicus* (16%, −50%), orangespotted sunfish *Lepomis humilis* (7%, −46%), and flathead chub *Platygobio gracilis* (28%, −45%). Groups with the greatest change from historical to contemporary studies included nonnative fishes (27%, 164%), piscivores (19%, 166%), sport fishes (33%, 103%), and lithophilic spawning species (55%, 36%).

Historical and contemporary fish assemblage surveys in Nebraska documented the presence of 64 and 66 species, respectively. Statewide mean site species richness increased by 55% and varied significantly (W = value of Wilcoxon signed-rank test, P-value; W = 848.5, $P < 0.01$) from historical (mean ± SE; 6.6 ± 0.27) to contemporary surveys (10.2 ± 0.33; Fig. 3). Species richness for the individual ecoregions differed significantly between

TABLE 1.—Extirpation and addition of species from a taxonomic assessment of historical (1939–1940) and contemporary (2003–2005) surveys in Nebraska streams. Numbers indicate the number of sites where the species was present. An asterisk indicates species that were considered rare (*i.e.*, found at <5% of the sites). Native (†) and nonnative (‡) species status are indicated following the species name

Species	Historical sampling (1939–1940)	Contemporary sampling (2003–2005)
Extirpations		
Goldfish *Carassius auratus*‡*	1	0
Common shiner *Luxilus cornutus*†*	4	0
Sturgeon chub *Macrhybopsis gelida*†*	5	0
Topeka shiner *Notropis topeka*†*	2	0
Mountain sucker *Catostomus platyrhynchus*†*	1	0
Bigmouth buffalo *Ictiobus cyprinellus*†*	3	0
Burbot *Lota lota*†*	1	0
Sauger *Sander canadensis*†*	2	0
Additions		
Pallid sturgeon *Scaphirhynchus albus*†*	0	2
Longnose gar *Lepisosteus osseus*†*	0	11
Gizzard shad *Dorosoma cepedianum*†*	0	16
Grass carp *Ctenopharyngodon idella*‡*	0	3
Spotfin shiner *Cyprinella spiloptera*‡*	0	1
Blue catfish *Ictalurus furcatus*†*	0	1
Northern pike *Esox lucius*‡*	0	7
Brook silverside *Labidesthes sicculus*†*	0	2
Western mosquitofish *Gambusia affinis*‡	0	23
Freshwater drum *Aplodinotus grunniens*†	0	21

historical and contemporary surveys in the CGP (W = value of Wilcoxon signed-rank test, P-value, historical mean ± historical SE, percentage change from historical to contemporary surveys; W = 1544, $P < 0.01$, 7.3 ± 0.56, 50%), NSH (W = 243.5, P = 0.01, 6.5 ± 0.49, 44%), and WCBP (W = 1594, $P < 0.01$, 6.0 ± 0.42, 85%) ecoregions. The mean proportion of nonnative species increased from historical to contemporary surveys by 57%. A significant difference in the statewide proportion of nonnative species was observed between historical and contemporary surveys (W = 8676, $P < 0.01$; Fig. 4). Proportions of nonnative species varied significantly between survey periods in the CGP (W = 1213.5, $P < 0.01$, 4.5 ± 1.31, 70%) and WCBP (W = 848.5, $P < 0.01$, 3.9 ± 1.02, 57%) ecoregions.

Cluster analysis of taxonomic structure using the least restrictive pairings separated the central, eastern, and western plains ecoregions (*i.e.*, CGP, HP, NSH, WCBP) and northwestern mixed grasslands (*i.e.*, NWGP, NWGLP; Fig. 5) using historical data. However, contemporary surveys grouped eastern and southern ecoregions (*i.e.*, CGP, WCBP), central and northern ecoregions (*i.e.*, HP, NSH, NWGP), and separated the NWGLP ecoregion from all others. Historically, the fish assemblage structure of the CGP ecoregion was more similar to the structure of HP and NSH ecoregions. The CGP assemblage structure was most similar to WCBP in contemporary surveys. Taxonomic ecoregion structure was more similar in contemporary surveys (maximum distance measure in cluster analysis; 0.51) than in historical surveys (0.61).

Overall taxonomic fish assemblage composition differed among ecoregions (PERMANOVA, *F*-test statistic, P-value; $F_{5,358} = 8.04$, $P < 0.01$) and time ($F_{1,358} = 14.6$, $P < 0.01$). Taxonomic assemblage composition differed between sampling events for the CCP ($F_{1,123} = 6.97$, $P < 0.01$), HP ($F_{1,43} = 1.58$, P = 0.02), NSH ($F_{1,43} = 2.19$, $P < 0.01$), NWGLP ($F_{1,9} = 1.63$,

TABLE 2.—Historical percentage of occurrence of both taxonomic and functional categories by ecoregion and statewide scale. Items in parentheses indicate percentage of change in occurrence from historical (1939–1940) to contemporary (2003–2005) surveys; NA indicates no percentage of change due to a lack of occurrence in the historical survey. An asterisk indicates species that were considered rare (*i.e.*, found at <5% of the sites). Native (†) and nonnative (‡) species status are indicated following the species name

	CGP	HP	NSH	NWGLP	NWGP	WCBP	Overall
	n = 63	n = 23	n = 23	n = 6	n = 9	n = 59	n = 183
Taxonomic							
Pallid sturgeon *Scaphirhynchus albus*†*	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)
Shovelnose sturgeon *Scaphirhynchus platorynchus*†*	0 (NA)	0 (NA)	0 (NA)	17 (−100)	0 (NA)	3 (100)	2 (33)
Shortnose gar *Lepisosteus platostomus*†	0 (NA)	0 (NA)	0 (NA)	17 (0)	0 (NA)	7 (375)	3 (360)
Longnose gar *Lepisosteus osseus*†*	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)
Goldeye *Hiodon alosoides*†*	2 (200)	0 (NA)	0 (NA)	50 (−100)	11 (−100)	7 (25)	5 (−11)
Gizzard shad *Dorosoma cepedianum*†*	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)
Central stoneroller *Campostoma anomalum*†	17 (64)	26 (100)	0 (NA)	0 (NA)	0 (NA)	3 (250)	10 (105)
Goldfish *Carassius auratus*‡*	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	2 (−100)	1 (−100)
Grass carp *Ctenopharyngodon idella*‡*	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)
Red shiner *Cyprinella lutrensis*†	67 (21)	43 (0)	42 (−10)	67 (0)	33 (0)	73 (28)	61 (18)
Spotfin shiner *Cyprinella spiloptera*‡*	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)
Common carp *Cyprinus carpio*‡	19 (258)	13 (233)	8 (300)	0 (NA)	11 (−100)	19 (164)	16 (221)
Western silvery minnow *Hybognathus argyritis*†	8 (0)	0 (NA)	0 (NA)	33 (−100)	11 (−100)	20 (−92)	11 (−60)
Plains minnow *Hybognathus placitus*†	43 (−37)	22 (−60)	4 (0)	17 (0)	0 (NA)	32 (−21)	29 (−30)
Brassy minnow *Hybognathus hankinsoni*†	24 (−27)	26 (−33)	42 (−50)	17 (−100)	0 (NA)	3 (100)	19 (−29)
Common shiner *Luxilus cornutus*†*	0 (NA)	13 (−100)	4 (−100)	0 (NA)	0 (NA)	0 (NA)	2 (−100)
Sturgeon chub *Macrhybopsis gelida*†*	6 (−100)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	2 (−100)	3 (−100)
Shoal chub *Macrhybopsis hyostoma*†	17 (−73)	4 (−100)	0 (NA)	0 (NA)	0 (NA)	3 (400)	8 (−7)
Silver chub *Macrhybopsis storeriana*†*	3 (100)	4 (−100)	4 (−100)	0 (NA)	0 (NA)	5 (100)	4 (43)
Pearl dace *Margariscus margarita*†*	2 (−100)	0 (NA)	8 (0)	0 (NA)	0 (NA)	0 (NA)	2 (−33)
Golden shiner *Notemigonus crysoleucas*†*	0 (NA)	0 (NA)	13 (−33)	0 (NA)	0 (NA)	3 (−50)	3 (−40)
Emerald shiner *Notropis atherinoides*†	13 (38)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	10 (117)	8 (79)
River shiner *Notropis blennius*†	24 (60)	13 (0)	8 (150)	0 (NA)	0 (NA)	17 (200)	16 (113)
Bigmouth shiner *Notropis dorsalis*†	22 (36)	35 (−38)	58 (−14)	50 (33)	33 (−33)	17 (120)	28 (23)
Sand shiner *Notropis stramineus*†	65 (34)	61 (14)	63 (7)	83 (−20)	44 (0)	71 (31)	66 (24)
Topeka shiner *Notropis topeka*†*	2 (−100)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	2 (−100)	1 (−100)

TABLE 2.—Continued

	CGP	HP	NSH	NWGLP	NWGP	WCBP	Overall
	n = 63	n = 23	n = 23	n = 6	n = 9	n = 59	n = 183
Suckermouth minnow *Phenacobius mirabilis*†	10 (150)	4 (0)	4 (−100)	0 (NA)	0 (NA)	22 (54)	11 (71)
Finescale dace *Phoxinus neogaeus*†*	2 (−100)	0 (NA)	4 (100)	0 (NA)	0 (NA)	0 (NA)	1 (0)
Bluntnose minnow *Pimephales notatus*†*	0 (NA)	0 (NA)	4 (300)	0 (NA)	0 (NA)	0 (NA)	1 (600)
Fathead minnow *Pimephales promelas*†	76 (21)	74 (−18)	42 (−10)	50 (0)	33 (33)	76 (7)	69 (8)
Flathead chub *Platygobio gracilis*†	38 (−63)	17 (−100)	8 (50)	83 (−20)	33 (−67)	22 (−15)	28 (−45)
Blacknose dace *Rhinichthys atratulus*†*	0 (NA)	0 (NA)	0 (NA)	17 (0)	0 (NA)	0 (NA)	1 (300)
Longnose dace *Rhinichthys cataractae*†	2 (100)	22 (0)	38 (56)	33 (50)	44 (0)	0 (NA)	11 (52)
Creek chub *Semotilus atromaculatus*†	22 (36)	48 (73)	38 (33)	50 (33)	22 (100)	29 (18)	31 (39)
River carpsucker *Carpiodes carpio*†	30 (37)	0 (NA)	0 (NA)	33 (50)	0 (NA)	25 (133)	20 (83)
Quillback *Carpiodes cyprinus*†	13 (75)	9 (−100)	8 (0)	0 (NA)	0 (NA)	5 (200)	8 (80)
Longnose sucker *Catostomus catostomus*†*	0 (NA)	9 (150)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	1 (200)
White sucker *Catostomus commersonii*†	22 (71)	57 (15)	25 (133)	17 (300)	33 (133)	5 (67)	22 (73)
Mountain sucker *Catostomus platyrhynchus*†*	0 (NA)	0 (NA)	0 (NA)	0 (NA)	11 (−100)	0 (NA)	1 (−100)
Bigmouth buffalo *Ictiobus cyprinellus*†*	5 (−100)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	2 (−100)
Shorthead redhorse *Moxostoma macrolepidotum*†	5 (300)	9 (50)	4 (500)	0 (NA)	0 (NA)	2 (800)	4 (443)
Black bullhead *Ameiurus melas* †	27 (0)	13 (−67)	13 (33)	33 (−100)	33 (−33)	37 (−36)	27 (−24)
Yellow bullhead *Ameiurus natalis*†	5 (167)	0 (NA)	8 (50)	0 (NA)	0 (NA)	2 (800)	3 (250)
Blue catfish *Ictalurus furcatus*†*	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)
Channel catfish *Ictalurus punctatus*†	32 (55)	9 (0)	4 (200)	33 (100)	11 (−100)	19 (291)	20 (124)
Tadpole madtom *Noturus gyrinus*†*	3 (50)	0 (NA)	13 (−100)	0 (NA)	0 (NA)	3 (−100)	4 (−57)
Stonecat *Noturus flavus*†	10 (50)	4 (300)	0 (NA)	0 (NA)	11 (0)	5 (367)	6 (264)
Flathead catfish *Pylodictis olivaris*†	5 (233)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	3 (550)	3 (380)
Grass pickerel *Esox americanus*†*	0 (NA)	0 (NA)	13 (133)	0 (NA)	0 (NA)	0 (NA)	2 (233)
Northern pike *Esox lucius*‡*	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)
Brown trout *Salmo trutta*‡*	0 (NA)	0 (NA)	4 (0)	0 (NA)	11 (300)	0 (NA)	1 (300)
Rainbow trout *Oncorhynchus mykiss*‡*	2 (−100)	9 (0)	4 (100)	0 (NA)	22 (0)	0 (NA)	3 (0)
Burbot *Lota lota*†*	0 (NA)	0 (NA)	0 (NA)	17 (−100)	0 (NA)	0 (NA)	1 (−100)
Plains killifish *Fundulus kansae*†	21 (38)	48 (0)	4 (−100)	0 (NA)	0 (NA)	7 (−75)	16 (3)
Plains topminnow *Fundulus sciadicus*†	13 (−75)	22 (−80)	58 (−36)	17 (−100)	22 (−100)	0 (NA)	16 (−50)
Western mosquitofish *Gambusia affinis*‡	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)
Brook silverside *Labidesthes sicculus*†*	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)
Brook stickleback *Culaea inconstans*†*	3 (50)	0 (NA)	4 (200)	0 (NA)	11 (−100)	0 (NA)	2 (175)

TABLE 2.—Continued

	CGP	HP	NSH	NWGLP	NWGP	WCBP	Overall
	n = 63	n = 23	n = 23	n = 6	n = 9	n = 59	n = 183
Rock bass *Ambloplites rupestris*‡*	0 (NA)	0 (NA)	0 (NA)	0 (NA)	11 (−100)	0 (NA)	1 (200)
Green sunfish *Lepomis cyanellus* †	13 (438)	4 (500)	33 (63)	0 (NA)	56 (−40)	17 (310)	17 (241)
Pumpkinseed sunfish *Lepomis gibbosus*†*	0 (NA)	0 (NA)	0 (NA)	0 (NA)	11 (100)	0 (NA)	1 (500)
Orangespotted sunfish *Lepomis humilis*†	6 (−25)	4 (−100)	0 (NA)	17 (−100)	0 (NA)	12 (−43)	7 (−46)
Bluegill *Lepomis macrochirus*‡	5 (567)	4 (100)	13 (67)	17 (200)	22 (100)	0 (NA)	5 (410)
Smallmouth bass *Micropterus dolomieu*‡*	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	2 (−100)	1 (100)
Largemouth bass *Micropterus salmoides*‡	6 (300)	4 (200)	13 (167)	17 (−100)	22 (50)	0 (NA)	6 (245)
Black crappie *Pomoxis nigromaculatus*‡*	2 (200)	0 (NA)	4 (−100)	17 (−100)	11 (−100)	0 (NA)	2 (−25)
White crappie *Pomoxis annularis*‡*	5 (100)	4 (−100)	0 (NA)	0 (NA)	0 (NA)	5 (−67)	4 (14)
Orangethroat darter *Etheostoma spectabile*†	8 (60)	13 (167)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	4 (125)
Iowa darter *Etheostoma exile*†*	5 (−67)	4 (100)	4 (0)	0 (NA)	0 (NA)	0 (NA)	3 (60)
Johnny darter *Etheostoma nigrum*†*	2 (200)	0 (NA)	4 (400)	0 (NA)	0 (NA)	0 (NA)	1 (650)
Yellow perch *Perca flavescens*‡*	2 (−100)	9 (−100)	0 (NA)	0 (NA)	0 (NA)	2 (−100)	2 (−75)
Sauger *Sander canadensis*†*	0 (NA)	0 (NA)	0 (NA)	17 (−100)	0 (NA)	2 (−100)	1 (−100)
Walleye *Sander vitreus*‡*	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	2 (0)	1 (200)
Freshwater drum *Aplodinotus grunniens*†	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)	0 (NA)
Functional							
Insectivores	89 (1)	91 (1)	96 (−1)	100 (0)	89 (1)	86 (4)	90 (6)
Omnivores	100 (0)	91 (1)	96 (−1)	100 (0)	78 (1)	98 (1)	97 (2)
Piscivores	17 (57)	13 (17)	29 (17)	33 (0)	33 (11)	15 (63)	19 (166)
Generalists	97 (1)	96 (0)	92 (−1)	100 (0)	67 (2)	95 (2)	94 (4)
Lithophilic	48 (13)	83 (−1)	58 (5)	83 (1)	67 (2)	47 (17)	55 (36)
Tolerant	97 (1)	96 (0)	92 (−1)	100 (0)	67 (2)	95 (2)	94 (4)
Nonnative	27 (60)	30 (16)	29 (16)	17 (6)	44 (8)	24 (58)	27 (164)
Sport	43 (26)	26 (5)	25 (7)	33 (5)	44 (7)	27 (54)	33 (103)

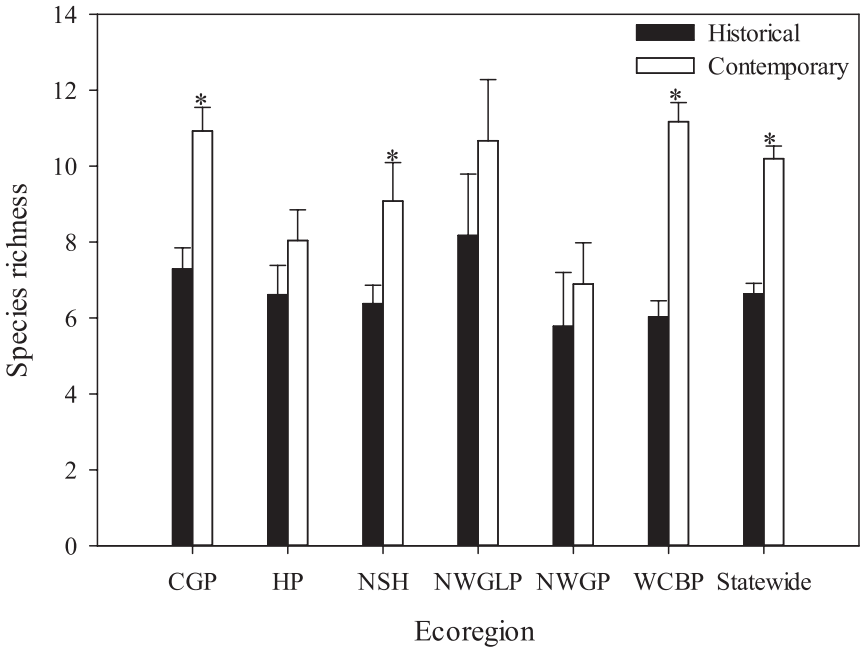

FIG. 3.—Taxonomic species richness by ecoregion and statewide spatial scale from historical (1939–1940) to contemporary (2003–2005) surveys of Nebraska streams. Ecoregion acronyms are Central Great Plains (CGP), High Plains (HP), Nebraska Sandhills (NSH), Northwestern Great Plains (NWGP), Northwestern Glaciated Plains (NWGLP), and Western Corn Belt Plains (WCBP). Bars indicate one standard error of the sample mean. An asterisk indicates statistically significant differences between historical and contemporary samples using a paired Wilcoxon signed-rank test (a = 0.05)

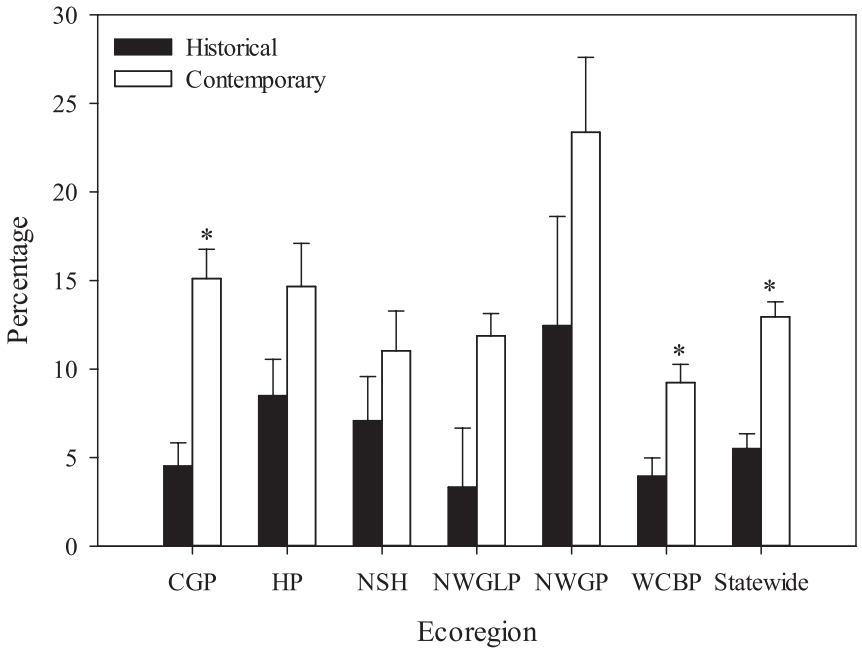

FIG. 4.—Mean proportion of species classified as nonnative by ecoregion and statewide spatial scale from historical (1939–1940) to contemporary (2003–2005) surveys of Nebraska streams. Ecoregion acronyms are Central Great Plains (CGP), High Plains (HP), Nebraska Sandhills (NSH), Northwestern Great Plains (NWGP), Northwestern Glaciated Plains (NWGLP), and Western Corn Belt Plains (WCBP). Bars indicate one standard error of the sample mean. An asterisk indicates statistically significant differences between historical and contemporary samples using a paired Wilcoxon signed-rank test (a = 0.05)

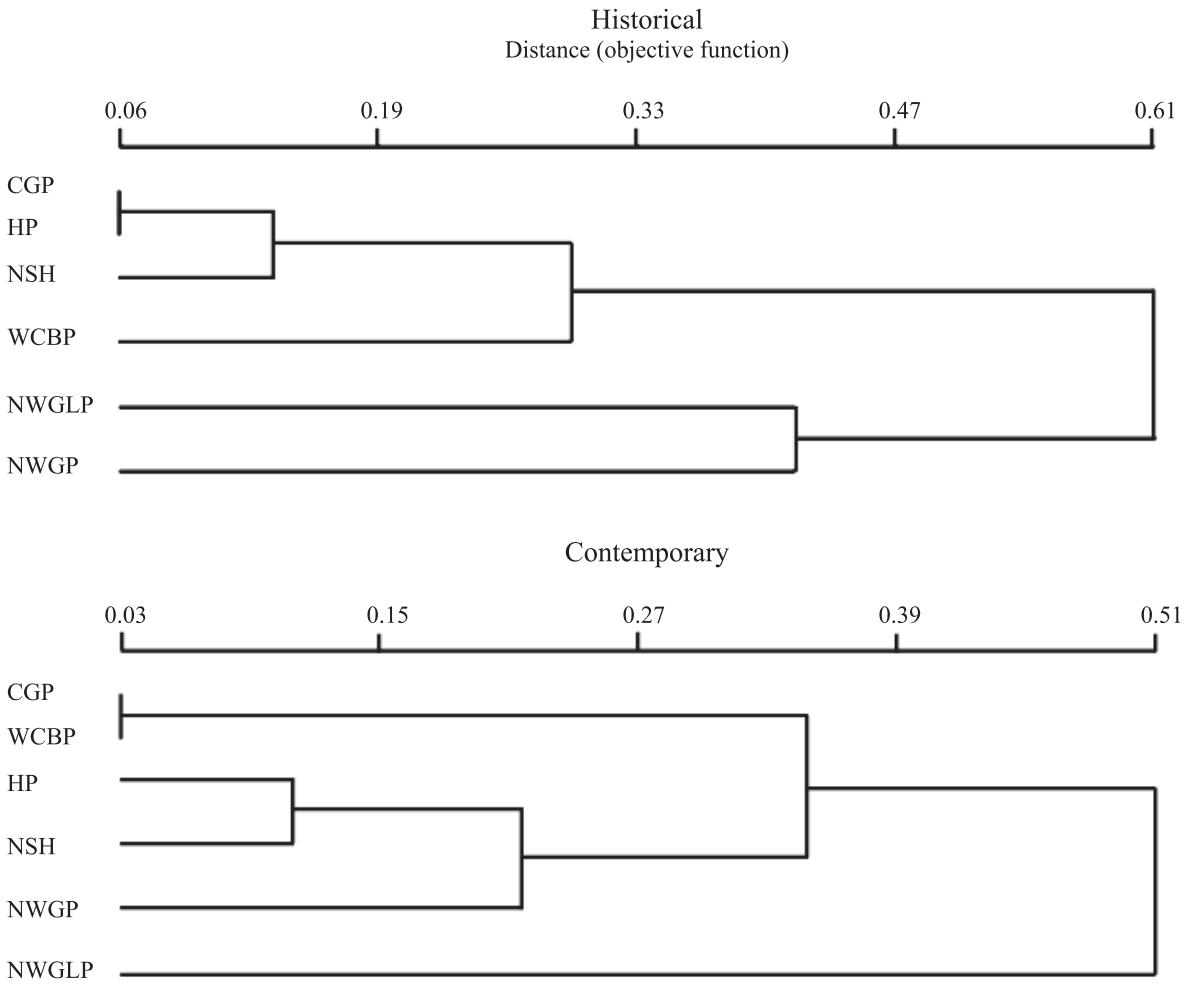

FIG. 5.—Historical (1939–1940) and contemporary (2003–2005) dendrograms of Nebraska's level III ecoregions, based on Jaccard dissimilarity measures comparing taxonomic presence-absence data. Nebraska's level III ecoregion acronyms include, Central Great Plains (CGP), High Plains (HP), Nebraska Sandhills (NSH), Northwestern Great Plains (NWGP), Northwestern Glaciated Plains (NWGLP), and Western Corn Belt Plains (WCBP)

P = 0.03), and WCBP ecoregions ($F_{1,115}$ = 10.3, P < 0.01). No difference in taxonomic structure was observed for the NWGP ecoregion ($F_{1,15}$ = 1.24, P = 0.28). Overall functional fish assemblage composition differed between ecoregions ($F_{5,358}$ = 3.94, P < 0.01) and time ($F_{1,358}$ = 31.5, P < 0.01). Functional fish assemblage composition differed between sampling events for CGP ($F_{1,123}$ = 13.4, P < 0.01), NSH ($F_{1,43}$ = 4.53, P < 0.01), NWGP ($F_{1,15}$ = 14.6, P = 0.04), and WCBP ecoregions ($F_{1,115}$ = 19.3, P < 0.01), but not for the HP ($F_{1,43}$ = 1.43, P = 0.08) and NWGLP ecoregions ($F_{1,9}$ = 2.24, P = 0.06). No correlation was found between mean HTI and mean dissimilarity of taxonomic composition or mean HTI and mean dissimilarity of functional composition data (Fig. 6). Ecoregion species turnover values were correlated to all of the HTI measures ($\rho \geq 0.83$; Fig. 6).

DISCUSSION

Nebraska's lotic fish assemblage structure has undergone a transition in the past 60 y. We identified declines in the occurrences of specialist species (*e.g.*, western silvery minnow,

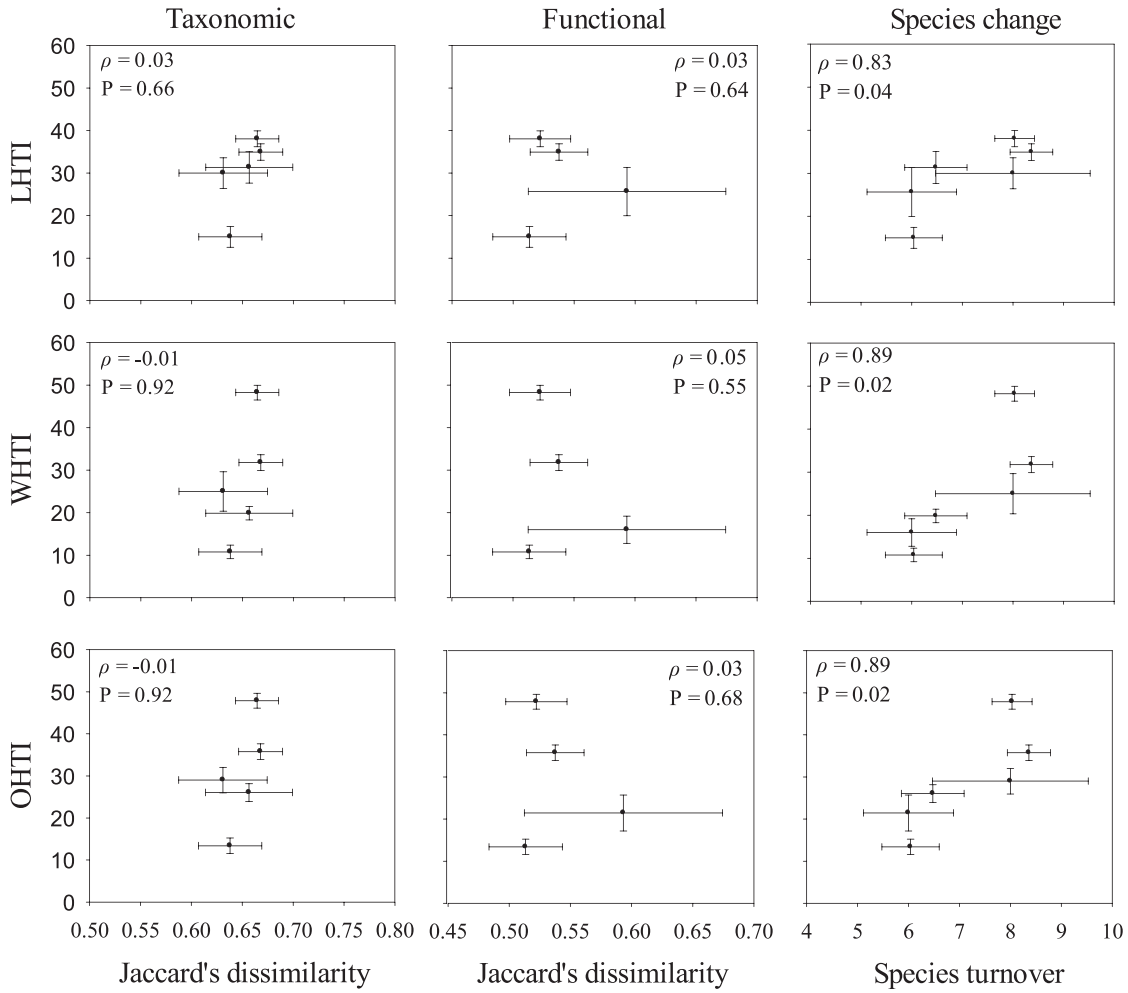

FIG. 6.—Spearman's rank correlation between a measure of anthropogenic alteration (*i.e.*, Human Threat Index; HTI) and mean dissimilarity of paired sites for ecoregions using both taxonomic and functional metrics for fishes sampled in Nebraska streams. Mean ecoregion species turnover values were also compared to the HTI. Error bars represent one standard error of the mean; horizontal bars are for variables on the x-axis. Human Threat Index (HTI) acronyms are local HTI (LHTI), watershed HTI (WHTI), and overall HTI (OHTI) values for Nebraska's level III ecoregions

plains topminnow, flathead chub) and increased occurrence of sport, nonnative, and generalist species (*e.g.,* bluegill, common carp, largemouth bass *Micropterus salmoides*). Despite a noticeable shift in the fish assemblage structure over 60 y, our comparison is likely a conservative description of fish assemblage change because alteration to the region likely occurred prior to the historical surveys used in this analysis, and our comparisons were based on occurrence rather than abundances. The historical data from this study were based on lotic systems that had already experienced anthropogenic alterations for 60 to 70 y. Due to widespread alterations of the system prior to the study, the most sensitive species likely experienced declines or extirpations prior to sampling. Unfortunately, accurate records of fish assemblages prior to European settlement do not exist. Nonetheless, our study provides insight into shifts of Nebraska's lotic fish assemblages.

Sampling techniques and effort varied by sampling period. Similar issues with differences in sampling techniques and effort among sampling periods are common in studies examining temporal trends using historical and contemporary data (Patton *et al.*, 1998;

Hoagstrom *et al.*, 2007; Jacquemin and Pyron, 2011). Despite differences in sampling techniques, research in small lotic systems of the Great Plains indicate that electrofishing and seine techniques sample similar fish assemblages (Patton *et al.*, 2000). A comparison of sampling techniques (*i.e.*, seine, benthic trawl, boat-mounted electrofishing) in Iowa's large nonwadeable rivers indicated that seines and electrofishing (the two gears used in our study) generally sampled the same fishes (Neebling and Quist, 2011). While not ideal, we used presence-absence data to minimize potential effects of gear biases on our analysis. Historical and contemporary fish assemblages can be effectively compared with presence-absence data given a large sample size to provide conservative estimates of temporal trends (Winemiller *et al.*, 2008; Jacquemin and Pyron, 2011). Ultimately, if sampling techniques are more effective in contemporary surveys, then a comparison of a less effective historical technique would yield a conservative estimate of species decline. Thus, our data provide important inferences regarding species declines. Inferences involving species increases, particularly small increases, should be interpreted with the understanding that increased sampling efficiency in recent surveys may contribute to the observed patterns.

Results of this research corroborate findings of declines among native fishes throughout the region (Cross and Moss, 1987; Patton *et al.*, 1998; Quist *et al.*, 2004; Hoagstrom *et al.*, 2011; Pasbrig *et al.*, 2012). Declining species include, but are not limited to, members of the *Hybognathus* genus (*i.e.*, brassy minnow *H. hankinsoni*, plains minnow *H. placitus*, western silvery minnow), *Macrhybopsis* genus (*i.e.*, sturgeon chub, silver chub *M. storeriana*, shoal chub *M. hyostoma*, peppered chub *M. tetranema*) and other native species such as pearl dace *Margariscus margarita*, flathead chub, plains topminnow, and finescale dace. Loss of native plains fishes is particularly troubling since most species are adapted to harsh environmental conditions characteristic of the region including high flow variability, high sediment loads, variable temperatures, and low dissolved oxygen (Matthews, 1988; Fausch and Bestgen, 1997; Dodds *et al.*, 2004). Declines of many native plains fishes are attributed to construction of impoundments and subsequent aquatic habitat alterations of altered substrate composition and decreased turbidity levels (Quist *et al.*, 2004; Quist *et al.*, 2005; Hoagstrom *et al.*, 2007). Bonner and Wilde (2002) suggested that decreased turbidity conditions may favor sight-feeding minnows such as emerald shiners *Notropis atherinoides*, over species with adaptations for the historically turbid environment of plains rivers and streams, such as the peppered chub. Altered hydrologic regimes also appear to negatively affect spawning success and recruitment of many native Great Plains fishes (Durham and Wilde, 2006; Falke *et al.*, 2010; Perkin and Gido, 2011). Biotic interactions with nonnative piscivores are implicated in the declines of imperiled species (Wenke *et al.*, 1993; Quist *et al.*, 2004; Hoagstrom *et al.*, 2007). Hoagstrom *et al.*, (2007) proposed predation by a nonnative piscivore, smallmouth bass *Micropterus dolomieu*, as a potential mechanism for the decline of native species in the upper Cheyenne River in South Dakota. Similar results were found by Quist *et al.*, (2004) for rivers in Wyoming. Competitive interactions with nonnative species could negatively affect imperiled species of the region. For instance, western mosquitofish, a nonnative species in Nebraska, was introduced into the Republican River in Nebraska in 1975 and restricts the reproductive success of the native plains topminnow in Nebraska (Lynch, 1988; Haas, 2005).

Although many species apparently declined since the historical surveys, some species and functional groups appear to have increased in frequency of occurrence in Nebraska's lotic systems. Coincidentally, increases in the proportion of sport fishes, nonnative, and generalist species across all ecoregions could be the direct result of anthropogenic influences. Increased frequency of occurrence of nonnative species is likely the result of

active propagation and stocking efforts. More specifically, several nonnative species are currently (*e.g.,* rainbow trout), or were historically (*e.g.,* common carp) propagated and stocked for angling opportunities in Nebraska (Livingston *et al.*, 1884). Some nonnative species can have negative effects on aquatic systems or greatly benefit from degraded conditions. For instance, common carp can degrade water quality by removing macrophytes and altering macroinvertebrate communities (Schrage and Downing, 2004; Miller and Crowl, 2006). As a result, common carp influenced the decline of native fishes (Koehn, 2004; Jackson *et al.*, 2010; Weber and Brown, 2011). However, some tolerant species (*e.g.,* green sunfish *Lepomis cyanellus*) may have increased in distribution as a result of degraded habitat conditions. More specifically, due to the degradation of aquatic systems (*e.g.,* low dissolved oxygen, altered natural flow regimes) native intolerant species may decline leaving a niche that can only be occupied by species with a high tolerance to poor water quality conditions. Trends of tolerant species additions or increases in relative abundance following land use change and stream degradation have been widely reported (Weaver and Garman, 1994; Anderson *et al.*, 1995; Onorato *et al.*, 2000; Gido *et al.*, 2010). For instance, additions of native tolerant species (*i.e.*, creek chub *Semotilus atromaculatus*, brown bullhead *Ameiurus nebulosus*) were reported following stream degradation in Virginia (Weaver and Garman, 1994).

Construction of dams alters native lotic fish assemblages (Martinez *et al.*, 1994). The number of impoundments with dams taller than 1.8 m in Nebraska has increased from 76 to 2368 since the 1930s (National Inventory of Dams, 2011). Additionally, over 27,000 small private impoundments were created in Nebraska prior to 1968 (Nebraska Game and Parks Commission, 2012b). Impoundments result in habitat fragmentation of lotic systems. Such fragmentation disrupts the reproductive success of native cyprinids and may be responsible for species extirpations (*e.g.,* speckled chub *Macrhybopsis aestivalis*, chub shiner *Notropis potteri*; Winston *et al.*, 1991; Wilde and Ostrand, 1999; Perkin and Gido, 2011). In addition, impoundments are commonly stocked with sport (*e.g.,* channel catfish, largemouth bass, bluegill; Blaser and Eades, 2006) and prey species (*e.g.,* gizzard shad *Dorosoma cepedianum*; Ward *et al.*, 2007). Many of these species emigrate from impoundments to connected lotic systems (Falke and Gido, 2006). Introduced lentic-specialist species may have a variety of interactions (*i.e.*, predation, competition) with native Great Plains lotic fishes. Relatively little is known regarding the potential cumulative effects of lentic fish introductions on native fishes in lotic systems. Additions of introduced lentic species appear to shift fish assemblage structure. For instance, a transition in fish assemblage structure has been reported in the Laramie River of Wyoming with pre-impoundment assemblages structured by catostomids and cyprinids, and post-impoundment assemblages characterized by nonnative sport fishes (*e.g.,* rainbow trout, smallmouth bass, walleye *Sander vitreus*) and few native cyprinids (*e.g.,* suckermouth minnow *Phenacobius mirabilis*; Quist *et al.*, 2005).

Because aquatic systems are commonly influenced by a variety of threats at various spatial scales, we used the HTI as a surrogate measure of anthropogenic alteration (Hoagstrom *et al.*, 2011). High HTI values should be associated with elevated levels of human alteration and likely have the largest cumulative effects on aquatic ecosystems. Results of the cluster analysis indicated ecoregions with the two highest HTI values (*i.e.*, CGP, WCBP) had the most similar fish assemblage structure in contemporary surveys. Correlation analyses indicated a relationship between HTI and mean species turnover. These trends were most pronounced on the watershed level and with a combined metric of anthropogenic alteration (OHTI). Moreover, it appears that large- (*i.e.*, watershed), rather than local-scale (*i.e.*, in-stream) threats may have had a greater influence on the shifts of fish assemblages in

Nebraska's lotic systems. Large-scale threats have been attributed to the decline of native fishes across the Interior Plains region of North America (Anderson *et al.*, 1995; Hoagstrom *et al.*, 2011). Specifically, declines of many plains fishes have been related to dewatering, habitat fragmentation, and other forms of anthropogenic alteration (Hoagstrom *et al.*, 2006, Hoagstrom *et al.*, 2011). The highest turnover rates of fish assemblages in Nebraska were located in eastern and southern ecoregions (CGP, WCBP). Hoagstrom *et al.,* (2006) found similar results with the loss of numerous species in eastern drainages of South Dakota (land use in eastern South Dakota is characterized by high levels of intensive row crop agriculture relative to other portions of South Dakota; Fodness, 1994). Results of our study, coupled with similar studies in the region, indicate that large-scale alterations have resulted in changes in fish assemblages and ultimately, the homogenization of fish assemblages.

Increases in similarity of Nebraska's fish assemblages from historic to contemporary surveys and the loss of native specialist species and increases of generalist, nonnative, and sport fish species characterize the process of biotic homogenization (Rahel, 2002). Changes in Nebraska's lotic fish assemblages are not surprising given the widespread shifts in land use and water quality following European settlement. However, such shifts could have many negative effects on assemblage structure and function (*e.g.,* decreased system stability, reduced ecosystem function) and may be irreversible (Olden *et al.*, 2004). Consequently, biotic homogenization must be understood so realistic conservation goals can be set and ecosystems can be managed. Our study provides conservative estimates of species declines and suggests that continued monitoring of Nebraska's lotic systems is essential for evaluating species status and developing conservation strategies.

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APPENDIX—Coordinates of sites that were surveyed in historical (1939–1940) and contemporary (2003–2005) surveys. Site coordinates are provided in the Universal Transverse Mercator (UTM) coordinate system

System name	Site ID	Easting	Northing	UTM zone
Republican River	0	577721	4430109	14
Western Sarpy Ditch	1	727283	4547350	14
Western Sarpy Ditch	3	727962	4545376	14
Platte River	4	734471	4541975	14
Papillion Creek	5	748404	4571993	14
Logan Creek	6	709046	4633597	14
Combination Ditch	7	729422	4650135	14
Logan Creek	12	676121	4682629	14
Union Creek	14	626978	4631774	14
Platte River	15	636604	4584178	14
Clear Creek	17	639354	4582787	14
Wahoo Creek	18	721080	4548033	14
Clear Creek	20	720061	4551759	14
Wahoo Creek	21	716427	4551020	14
Platte River	24	721218	4566579	14
Platte River	25	709052	4586489	14
Maple Creek	27	708545	4602347	14
Elkhorn River	28	704440	4610992	14
Logan Creek	29	704592	4616099	14
Elkhorn River	32	688739	4634258	14
Elkhorn River	34	668946	4651459	14
North Fork Elkhorn River	35	629549	4662785	14
North Fork Elkhorn River	36	621870	4675746	14
Bazile Creek	37	589845	4703879	14
Bazile Creek	38	590251	4707189	14
Spring Creek	39	590325	4707028	14
Bazile Creek	40	591819	4718029	14
Verdigre Creek	41	572794	4701827	14
Bazile Creek	42	586314	4734175	14
Niobrara River	43	577807	4733541	14
Steel Creek	45	577707	4733640	14
Ponca Creek	46	496628	4759875	14
Keya Paha River	47	491855	4751533	14
Niobrara River	48	492124	4744929	14
Long Pine Creek	50	441137	4702803	14
Long Pine Creek	51	441711	4710663	14
Bone Creek	52	428268	4711128	14
Snake River	55	348070	4725583	14
Plum Creek	56	409449	4713666	14
Long Pine Creek	57	442973	4716781	14
Calamus	58	464554	4648275	14
North Loup River	59	468278	4625176	14
Cedar River	60	527560	4624283	14
Beaver Creek	61	575991	4623726	14
Shell Creek	62	620137	4601286	14
Wood River	64	558952	4530378	14
Middle Loup River	65	546648	4561562	14
Middle Loup River	66	506505	4559612	14

APPENDIX—Continued

System name	Site ID	Easting	Northing	UTM zone
Mud Creek	68	457844	4577764	14
Middle Loup River	69	446696	4612584	14
Middle Loup River	70	392547	4639653	14
Dismal River	71	373127	4626375	14
Dismal River	72	330994	4635378	14
South Loup River	74	396931	4587692	14
South Loup River	77	508737	4541428	14
Lincoln Creek	78	618220	4529710	14
Platte River	79	560641	4524995	14
Little Blue River	81	547136	4478113	14
Spring Creek	83	546719	4478364	14
Platte River	85	492544	4500703	14
Buffalo Creek	86	454600	4512852	14
Platte River	87	384886	4541663	14
Irrigation Ditch	88	379972	4545234	14
North Platte River	89	322649	4564366	14
Medicine Creek	92	351796	4513745	14
Medicine Creek	93	353961	4511786	14
Red Willow Creek	94	337271	4494145	14
Stinking Water Creek	96	312424	4477331	14
Frenchman River	97	298776	4477126	14
Republican River	98	284606	4435877	14
Rock Creek	100	267663	4435866	14
Rock Creek	102	263517	4443154	14
Spring Creek	104	282815	4493722	14
South Platte River	105	270576	4555566	14
Lonergan Creek	106	263843	4573376	14
North Platte River	107	721597	4584435	13
Blue Creek	109	722319	4608940	13
Cedar Creek	111	681583	4598826	13
Greenwood Creek	113	668149	4598774	13
Lodgepole Creek	116	600713	4564639	13
Tubbs Spring	118	608046	4648847	13
Akers Draw	119	593510	4646177	13
Horse Creek	121	579336	4641379	13
Winters Creek	122	614010	4634918	13
Niobrara River	124	603137	4697315	13
Sowbelly Creek	126	596504	4742795	13
White River	127	631240	4730101	13
White River	128	629696	4727360	13
Niobrara River	133	689416	4705895	13
Arkansas Flats Creek	136	277621	4707661	14
Snake River	137	277669	4717067	14
Niobrara River	138	279984	4743542	14
Elkhorn River	140	502386	4707215	14
Haines Branch	142	689895	4516170	14
Middle Creek	144	683220	4520358	14
Little Salt Creek	147	695249	4530338	14
West Fork Big Blue River	149	644261	4514995	14
Platte River	201	641721	4584591	14

APPENDIX—Continued

System name	Site ID	Easting	Northing	UTM zone
Clear Creek	202	644317	4582227	14
Clear Creek	205	636621	4581320	14
Western Sarpy Ditch	206	727107	4548808	14
Silver Creek	208	715160	4551955	14
Weeping Water	212	247545	4519138	15
Four mile Creek	213	251030	4543315	15
Platte River	214	251078	4545730	15
Little Nemaha River	217	738725	4501616	14
South Fork Little Nemaha	218	745475	4491567	14
Little Nemaha River	219	251209	4484842	15
Little Nemaha River	220	267697	4470193	15
Wine Branch	221	287663	4454823.2	15
Big Nemaha River	223	284976	4435699	15
Big Muddy Creek	224	285000	4436577	15
Rock Creek	225	263248	4435324	15
Fourmile Creek	226	249238	4436861	15
South Fork Big Nemaha	227	249924	4443024	15
North Fork Big Nemaha River	228	722152	4481192	14
Big Muddy Creek	229	250589	4469098	15
Big Muddy Creek	230	263129	4457475	15
Long Branch	231	755248	4451428	14
North Fork Big Nemaha River	232	747484	4456081	14
South Fork Big Nemaha	233	752794	4433278	14
Turkey Creek	234	743177	4436676	14
Cub Creek	235	685036	4459210	14
Turkey Creek	236	679055	4470588	14
Salt Creek	238	689273	4498549	14
Salt Creek	239	695630	4501876	14
Turkey Creek	240	663464	4495066	14
Little Blue River	242	595401	4453666	14
Spring Creek	243	619816	4446136	14
Big Sandy Creek	244	632374	4456703	14
Rose Creek	245	642446	4435053	14
Little Blue River	246	667320	4433805	14
Platte River	247	685992	4591346	14
Willow Creek	249	551315	4437885	14
Elm Creek	250	547393	4446171	14
Republican River	251	529909	4436611	14
Thompson Creek	252	519425	4442944	14
Turkey Creek	253	487223	4440869	14
Republican River	254	461627	4439166	14
Prairie Dog Creek	255	468646	4429298	14
Sappa Creek	256	449480	4441729	14
Muddy Creek	258	422574	4461934	14
Republican River	259	379374	4453339	14
Medicine Creek	260	383663	4487934	14
Red Willow Creek	261	358631	4468437	14
Frenchman River	263	332946	4461282	14
North Fork Republican River	264	283033	4434677	14
North Fork Republican River	265	247978	4435161	14

APPENDIX—Continued

System name	Site ID	Easting	Northing	UTM zone
South Platte River	266	741206	4547802	13
South Platte River	267	257596	4552263	14
North Platte River	273	579979	4648457	13
North Fork Big Blue River	274	664495	4517751	14
Platte River	277	437057	4509672	14
Niobrara River	280	728920	4724509	13
Bear Creek	281	293245	4755075	14
Niobrara River	282	326190	4741372	14
Niobrara River	283	375695	4745534	14
Minnechaduza Creek	284	374792	4749667	14
Minnechaduza Cheek	285	372290	4748687	14
Elkhorn River	288	515447	4701529	14
Beaver Creek	289	537379	4647585	14
Merriman Creek	290	579601	4698786	14
West Bow Creek	293	633852	4729408	14
Bow Creek	294	650873	4733221	14
Sand Creek	295	700900	4565643	14
West Fork Big Blue River	296	655683	4509519	14
West Fork Big Blue River	297	618662	4509287	14
West Fork Big Blue River	298	594304	4507782	14
Loup River	301	616892	4590914	14
Looking Glass Creek	302	610005	4592377	14
Loup River	303	585887	4577749	14
Prairie Creek	304	585377	4566784	14
South Loup River	306	467984	4543633	14
North Loup River	307	477990	4623200	14
North Loup River	308	523215	4590680	14
Cedar River	309	571185	4601576	14
Bogus Creek	310	595268	4599864	14
Beaver Creek	311	593212	4604229	14
Elkhorn River	312	569725	4668958	14
Cache Creek	313	554673	4673505	14
Elkhorn River	315	708318	4610452	14
Logan Creek	316	686984	4668832	14
South Creek	319	677971	4705763	14