Consequences of Stream Fragmentation and Climate Change for Rare Great Plains Fishes

Joshuah S. Perkin and Keith B. Gido
Division of Biology, Kansas State University, 116 Ackert Hall, Manhattan, Kansas 66506
Phone: (785) 532-5088, Fax: (785) 532-6653, Emails: jperkin@ksu.edu; kgido@ksu.edu

Eric Johnson
Kansas Department of Wildlife and Parks, 512 SE 25th Ave., Pratt, KS 67124
Phone: (620)-672-0798, Email: ericj@wp.state.ks.us

Vernon M. Tabor
United States Fish and Wildlife Service Ecological Services Field Office, 2609 Anderson
Avenue, Manhattan, KS 66502
Phone: (785) 539-3474, Email: vernon_tabor@fws.gov

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Executive Summary

Rate of global biodiversity loss increased significantly during the 20th century associated with human environmental alterations. Specifically, mismanagement of freshwater resources contributed to historical and contemporary loss of stream-dwelling fish diversity and will likely play a role in determining the persistence of species in the future. We present a mechanistic pathway by which human alteration of streams has caused the decline of a unique reproductive guild of Great Plains stream-dwelling fishes, and suggest how future climate change might exacerbate these declines. Stream fragmentation related to impoundments, diversion dams and stream dewatering are consequences of increasing demand for freshwater resources and have effectively created a mosaic of large river fragments throughout the Great Plains of central North America. We analyzed community composition, species population status, fragment size and flow regime components for 60 stream fragments spanning the latitudinal range of the contiguous United States. Stream fragment lengths were a strong predictor of conservation status among pelagic-spawning cyprinid populations, explaining 71% of cumulative extirpations. Mean fragment lengths were least for extirpated (140 ± 55 km) and declining (205 ± 65 km) populations and highest for stable (425 ± 185 km) populations. Similarly, components of flow regimes within fragments associated with magnitude of discharge explained 29% of variation among daily streamflow values, and extirpations were positively correlated ($r = 0.36$, $P = 0.02$) with declining discharges. Future climate change scenarios project stream fragments in the southern Great Plains may lose up to 12% of their discharge before 2060, while stream fragments in the northern Great Plains may gain up to 5%. Continued human demand for water resources combined with reduced availability in the southern Great Plains will likely contribute to increased need for fragmentation (e.g., impoundments) and cause further disparity from natural flow regimes. Conservation measures that restore connectivity of river fragments and natural flow regimes will likely benefit pelagic-spawning cyprinids. Moreover, maintenance of long stream fragments will ensure that stable populations of these species do not undergo further declines.

Background

Human impact upon global ecosystems has escalated since the beginning of the Industrial Revolution, to the extent that human-induced shifts in biotic, sedimentary and geochemical properties may merit the advent of a new epoch: *the Anthropocene* (Crutzen 2002). Whereas this term is still in its infancy (see Zalasiewicz et al. 2008), there is little question that humans have drastically altered global biological and ecological processes (Vitousek et al. 1997). For example, alterations to the hydrologic cycle in the form of groundwater depletion and impoundment of surface waters have compromised the connectivity of these ecosystems (e.g., riparian and floodplain regions), thereby creating heightened concern for a suite of environments dependent upon freshwater (Schroter et al. 2005). Furthermore, global changes in climate stemming from carbon dioxide emissions are projected to increase variation in temperature and precipitation, which are strongly linked to the functioning of freshwater environments (Milly et al. 2005). Collectively, these alterations have imperiled freshwater organisms worldwide, most notably organisms dependent upon streams and rivers for long-term persistence (Lytle and Poff 2004, Dudgeon et al. 2006).

Human-induced habitat fragmentation and loss associated with stream regulation has contributed to declines in the abundance and distribution of native stream-dwelling faunas (Fahrig 2003, Helfman 2007). Within the contiguous United States, 85% of rivers are fragmented by impoundments that disrupt organism movement and alter streamflow (Hughes *et al*., 2005). These disturbances are thought to have contributed to the imperiled status of about 40% of North American freshwater and diadromous fishes (Jelks et al. 2008). Declines in these species are associated with the splitting up of riverscapes (*sensu* Fausch et al. 2002), which generally include alteration of species-specific spatial dynamics (e.g., dispersal, Haro et al. 2000), increased isolation among populations (e.g., evolutionarily significant units, Schick and Lindley 2007), altered geomorphic processes that create and maintain instream habitat (e.g., flow regime, Poff et al. 1997), altered autecological processes (e.g., reproductive cues, Taylor and Miller 1990; range of environments within physiological limits, Matthews 1987) and reduction in amount of contiguous habitat (e.g., longitudinal stream length, Dudley and Platania 2007; Figure 1). Among these principal factors, alteration to flow regime is commonly implicated in the decline of stream-dwelling fish populations, and a growing body of literature suggests flow regime is a major component required for maintaining integrity within stream fish communities (e.g. Baxter 1977, Poff et al. 1997, Marchetti and Moyle 2001, Lytle and Poff 2004, Propst and Gido 2004, Taylor et al. 2008, Gido et al. 2010). For example, magnitude of floods and high flow pulses that maintain instream habitat are reduced following impoundment (Richter et al. 1996, Perkin and Bonner 2010) and depending upon reservoir management, downstream reaches of impounded streams may experience reductions in mean annual flow and an increase in number of days with zero flow (Bonner and Wilde 2000). Similarly, agricultural and municipal water withdrawals may act in concert with stream regulation to reduce streamflow and contribute to fish extirpations by increasing zero flow days and stream desiccations (Gido et al. 2010). Accordingly, components of a flow regime including flow duration, frequency, magnitude, rate of change and timing (see Poff et al. 1997) are generally regarded as important in terms of conservation delivery for rare and declining stream-associated species (Propst and Gido 2004, Durham and Wilde 2006, Poff and Zimmerman 2010). However, the additive impacts of stream fragmentation, reduced streamflow and associated consequences of future climate change require further research in favor of enhancing conservation of rare and declining species (e.g., Dudley and Platania 2007, Schick and Lindley 2007, Rahel and Olden 2008).

Impacts of stream fragmentation will be exacerbated under future climate change scenarios. Streamflow derived from runoff within a river basin is generally regarded as the difference between precipitation and evapotranspiration, and represents a measure of freshwater resource availability within a region (Milly et al. 2005). Consequently, changes in precipitation and evapotranspiration hold potential for altering freshwater resources available for human use and maintenance of ecological processes within riverine ecosystems. Future climate change scenarios predict increases in temperature and evapotranspiration rates over much of the global surface (IPCC 2007). Similarly, climate change models indicate many regions will experience a loss in magnitude of annual precipitation, which will contribute to reduced streamflow availability. Within the continental United States, these components will collectively contribute to reduced streamflow availability within the central and southern plains region, east of the Rocky Mountains (Milly et al. 2005). Climate impacts may be particularly detrimental in this region because of the predominant eastward drainage direction of most large streams and relatively little hydrologic connectivity to northern wetter and cooler regions (Matthews and Zimmerman 1990). Consequently, a future climate characterized by prolonged periods of little precipitation bordered by severe deluge events will likely exacerbate cotemporary impacts associated with stream fragmentation (Covich et al. 1997). Specifically, alteration of streamflows associated with reduced discharge availability will negatively influence stream-dwelling fishes (Rahel and Olden 2008; Figure 1) by altering spatial dynamics of stream-dwelling fishes, isolating populations, and reducing availability of large sized patches of habitat (Winston et al. 1991, Luttrell et al. 1999, Luttrell et al. 2002, Falke and Gido 2006).

Reproductive ecology is a particularly insightful method of explaining the decline and disappearance of many fluvial-dependant fishes in fragmented rivers. A growing body of literature exists for the guild of small-bodied cyprinids that produce semi-buoyant, non-adhesive eggs within pelagic zones of large flowing streams (e.g., Moore 1944, Platania and Altenbach 1998, Dudley and Platania 2007, Durham and Wilde 2008a, Wilde and Durham 2008). These so-called pelagic-spawning cyprinids represent 25-40% of imperiled species within ecoregions of the Great Plains (Jelks et al. 2008) and have precipitously declined since at least the 1950s when species belonging to this guild dominated vertebrate communities within Great Plains prairie rivers (Cross and Moss 1987, Fausch and Bestgen 1997, Gido et al. 2010). During the span of about the last 60 years, some pelagic-spawning cyprinids became extirpated from as much as 80% (Arkansas River shiner, *Notropis girardi*, Wilde 2002) to 90% (peppered chub *Macrhybopsis tetranema*, Luttrell et al. 1999) of their historical range. Reduced reproductive success is commonly invoked to explain pelagic-spawning cyprinid declines because of the unique spatial dynamics involved with their life histories (Durham and Wilde 2006). Pelagic-spawning cyprinids dispense gametes into pelagic zones of flowing streams. Immediately following spawning, water enters the chorion membrane and fills the perivitelline space of eggs, causing eggs to swell and become semi-buoyant. These semi-buoyant eggs remain suspended within the water column and drift for 24-28 hours before hatching, after which pre-larvae develop as they drift for an additional 2-3 days, presumably becoming displaced great distances (e.g., up to 140km) downstream from parent localities (Moore 1944, Bottrell et al. 1964, Platania and Altencach 1998). Stockpiling of reproductively active adults below barriers suggests migration during reproduction, which provides a mechanism for recolonization of upstream reaches following downstream drift of pre-larva (Cross 1950, Platania and Altenbach 1998, Bonner 2000, Hoagstrom et al. 2010b). Stream fragmentation therefore carries the potential to negatively impact the spatial dynamics of pelagic-spawning cyprinids via interruption of

dispersal across two planes of space (i.e., in downstream and upstream directions) and time (i.e., during pre-larval and adult life stages; Dudley and Platania 2007).

Objectives Summary

Historical and contemporary patterns in stream fragmentation combined with projected changes in climate present a substantial conservation challenge for pelagic-spawning cyprinids in the Great Plains. Quantifying fragmentation and determining threshold values for the longitudinal stream length necessary for imperiled species persistence will ultimately benefit management plans by providing information on the probability of long-term success of decisions (e.g., successful repatriation, Luttrell 1997, Luttrell et al. 2002). Projecting potential success of management plans before actions are taken aids in developing landscape-scale (or riverscape-scale) strategies that maximize effectiveness while keeping costs down. Additionally, quantifying specific aspects of a flow regime that are necessary for the long-term persistence of imperiled species aids not only in contemporary conservation planning (e.g., management of human-mediated flow regimes) but predicting potential effects of climate change. The goal of this study was to provide information that can be used in decision-support tools to enhance conservation delivery for pelagic-spawning cyprinid communities throughout the Great Plains Landscape Conservation Cooperative (GPLCC) area. Specific objectives of this project included: 1) documenting the extent to which Great Plains riverscapes inhabited by this guild of fishes are fragmented; 2) evaluating threshold values for riverscape fragmentation that correspond with declines or extirpations of each species; 3) quantifying specific parameters of flow regime that are necessary for conservation and recovery of declining communities; 4) prioritizing regions in need of increased connectivity or where climate change is most likely to negatively impact extant populations.

Study Area

The North American plains are a semi-arid region dominated by grassland, prairie and steppe biomes that span approximately 20˚ of latitude from the Alberta Province of Canada south to the Rio Grande Basin of northern Mexico. Within the middle lying United States, the Rocky Mountains define the western extent of the plains and contribute to a gradual eastward slope toward the Mississippi River, which defines the eastern edge of the plains. Consequently, most large-order plains prairie rivers flow from west to east within three major basins: the Missouri River, Arkansas River and Red River basins (Matthews and Zimmerman 1990). These river basins occur in two major plains regions, the Great Plains and Osage Plains (collectively referred to as the Great Plains hereafter) and span the majority of 10 states: Montana, North Dakota, South Dakota, Wyoming, Nebraska, Colorado, Kansas, Oklahoma, New Mexico and Texas. Additionally, southern portions of the Great Plains are drained by river basins that empty directly into the Gulf of Mexico, including the Brazos, Colorado, San Antonio Bay, Nueces and Rio Grande basins. Throughout this region, portions of large prairie rivers characterized by shallow gradients, sandy bottoms, relatively high turbidity and lying within the contiguous United States were chosen based on inhabitance by pelagic-spawning cyprinids and availability of historical ichthyofaunal and streamflow data.

4

Objective 1: Methods

We used four criteria to define stream fragments available for inclusion in this study. First, fragments had to be easily definable based on presence of confirmed or suspected barriers to fish passage. The longitudinal stream length between barriers was used to define fragment length and was quantified using the stream layer associated with the National Hydrologic Dataset (NHD) from the United States Geological Survey (USGS). Fragment lengths were measured in kilometers and followed the sinuous path of streams, incorporating main-channel habitats, but not oxbows or unconnected secondary channels. In cases where the downstream barrier consisted of a reservoir, the upper reaches of impounded waters were used to define the lower boundary of the fragment (Bonner and Wilde 2000, Dudley and Platania 2007). In cases where upper reaches of large rivers were fragmented by a downstream barrier, but not an upstream barrier (e.g., North Fork of the Red River, Winston et al. 1991), fragment lengths were measured from the confluence of major tributaries defining the giving river, or the upper most historical collection of pelagic-spawning cyprinids (based upon accounts in Lee et al. 1980, as in Dudley and Platania 2007). Some stream fragments were defined according to stream desiccations, in which stream beds remained dry for greater than 100 days of the year for the period 1969-2009. Whereas this type of barrier is likely semi-permeable (i.e., passable during high flow events) we included stream desiccations as barriers to fish passage because of the substantial period of the year in which movement was precluded (Luttrell et al. 1999).

Second, at least one USGS streamflow gauge had to be present within each fragment to allow for comparisons among contemporary and historical flow regime components. When possible, USGS gauges near the downstream portion of fragments were selected so that flow regime components were characteristic of the hydrology occurring with the given fragment. Additionally, USGS gauges were selected for inclusion based upon availability of streamflow data. Gauges with historical data predating the onset of notable human disturbance to streamflow (i.e., pre-1970; Milly et al. 2005, Gido et al. 2010) were prioritized for inclusion and we targeted a period of at least 30 years (i.e., 1938-1968). When historical streamflow data were limited among available gauges (i.e., <30, but >10 continuous years), we noted the historical period of gauges included in analysis. When historical streamflow data were sparse among available gauges (i.e., < 10 continuous years), we excluded the associated fragment from historical flow analysis. Lastly, to facilitate inter-basin comparisons in flow regime, only one USGS gauge was included for each fragment regardless of total fragment length.

Third, fragments had to be currently or historically inhabited by at least one of eight species of confirmed or suspected pelagic-spawning cyprinid. We selected eight species based on similarities in inhabited streams, distributions within these streams, and susceptibility to capture among similar sampling methodologies. Species selected for inclusion consisted of four confirmed members of the pelagic-spawning guild: the plains minnow *Hybognathus placitus* (Platania and Altenbach 1998), the Arkansas River shiner *Notropis girardi* (Moore 1944), the sturgeon chub *Macrhybopsis gelida* (Hoagstrom et al. 2006) and the peppered chub *Macrhybopsis tetranema* (Bottrell et al. 1964). Additionally, we included four species of suspected pelagic-spawning guild members based on literature accounts and similarities in phylogeny and morphology: the flathead chub *Platygobio gracilis* (Bonner and Wilde 2000), the shoal chub *Macrhybopsis hyostoma* (Cross and Collins 1995), the silver chub *Macrhybopsis storeriana* (Simon 1999) and the prairie chub *Macrhybopsis australis* (Eisenhour 2004). In general, these species inhabit large-order Great Plains prairie streams where their distributions

are limited to mainstem habitats including shallow braided streams, sandy shoals and backwaters.

Forth, historical data pertaining to the conservation status of pelagic-spawning cyprinids had to be available for each fragment. Because of differences in sampling methodology, data were used to define coarse levels of conservation status: stable, declining, and extirpated. Stable described populations with no reduction in abundance (e.g., density, relative abundance, rank abundance) or distribution (e.g., area inhabited, presence/absence among sampling sites) through time, despite continued monitoring for up to 20 years. Declining (or depleted) described populations with reductions in either abundance or distribution over a period of at least 20 years of continued sampling. Extirpated (or undetectable) described populations not detected within a given fragment in at least 20 years despite continued monitoring, or were reported as extirpated during literature accounts. When conservation status of a species could not be readily determined within a given fragment, a broad cross-basin analysis of the status of Great Plains fishes was used to confirm or define conservations status (i.e., Hoagstrom et al. 2010a). We tested for significant differences in fragment length among population statuses within species using a single factor Analysis of Variance (ANOVA, $\alpha = 0.05$) and pair-wise contrasts in the form of Fischer's Least Significant Differences (LSD, $\alpha = 0.05$). Additionally, we report grand mean (i.e., mean among all populations of all species) fragment lengths associated with stable, declining and extirpated populations among species.

Objective 1: Results

Within the Great Plains region of North America, encompassing the entire GPLCC area, 60 stream fragments met the requirements for inclusion in our study. The longitudinal length of these fragments ranged from 38 km to 711 km (Table 1). Fragments ranged in distribution from Montana and North Dakota south to New Mexico and Texas, and from Colorado and Wyoming east to portions of Louisiana and Missouri (Figure 2). Barriers to fish dispersion included 36 dams associated with water diversions, hydroelectric generation and reservoir storage, 21 upper bounds defined by tributary confluences or upstream extent of pelagic-spawning cyprinid distributions, 39 lower bounds defined by impounded water, and six localized regions where water withdrawals resulted in stream desiccations. USGS streamflow gauges occurred in each fragment and historical data were generally (57 out of 60) available during the historical time period (1938-1968) and always available for the contemporary time period (1969-2009).

Conservation status of confirmed or suspected pelagic-spawning cyprinids consisted of 57% extirpated, 21% declining, and 22% stable populations (n = 157 observations among species) within the 60 fragments included in our analyses (Table 2). Among species, the plains minnow occurred in the highest number of fragments (n = 48) and the narrowly distributed prairie chub occurred in the lowest number (n = 4). Excluding extensively dewatered fragments (i.e., fragments 33, 40, 41; Luttrell et al. 1999), lengths differed significantly among populations of plains minnow with different conservation statuses ($F_{1,43} = 41.78$, $P < 0.01$), as did lengths associated with Arkansas River shiner ($F_{1,14} = 37.59$, $P < 0.01$), sturgeon chub ($F_{1,10} = 25.19$, $P < 0.01$), flathead chub ($F_{1,24} = 29.78$, $P < 0.01$), shoal chub ($F_{1,12} = 75.76$, $P < 0.01$) and silver chub ($F_{1,12} = 75.76$, $P < 0.01$) populations (Figure 3). Comparisons among species revealed that regardless of confirmed or suspected status as pelagic-spawning, extirpated populations were on average (i.e., grand mean among populations according to conservation status ± SD) associated with 140 (± 55) km long fragments, whereas declining populations averaged 205 (± 65) km and stable populations averaged 425 (± 185) km. Comparisons among species grand means (i.e.,

mean for each species according to conservation status ± SD) showed differences ($F_{1,22} = 48.39$, $P < 0.01$) among extirpated (mean = 136 ± 21), declining (226 ± 69) and stable populations (458 ± 137), so that fragment length varied significantly with conservation status.

Objective 2: Methods

Given stream fragment lengths were related to conservation status within and among pelagic-spawning cyprinid species, we tested for potential thresholds in stream fragment length associated with declines and extirpations (i.e., localized extinctions). For the initial analysis, we used recursive portioning in the form of Classification Tree Analysis (CTA; De'ath and Fabricius 2000) to assign conservation status based on stream fragment length. We asked if thresholds existed for species persistence (i.e., extant populations) and local extinction, which might lend insight into the minimum possible fragment length needed to maintain pelagic-spawning cyprinid populations. For extinction threshold analysis, declining and stable populations of species were combined to represent fragments capable of supporting persistence of pelagic-spawning species, although we acknowledge declining populations may in fact be related to fragment length (Dudley and Platania 2007, this study). Finally, we tested for a relationship between cumulative extirpation within pelagic-spawning cyprinid communities and stream fragment length by regressing proportion of community extirpation as a function of stream fragment length (following Dudley and Platania 2007). We tested for significance of the slope using polynomial logistic regression and quantified the correlation coefficient using a Nagelkerke R^2 value.

Objective 2: Results

Fragment length thresholds associated with changes in conservation status and localized extirpations varied by species. Classification Tree Analysis revealed the majority (i.e., >50%) of extirpated populations occurred within relatively short stream fragment lengths among all confirmed and suspected pelagic-spawning cyprinids (Table 3). Calculation of intermediate fragment lengths was not possible for *N. girardi*, *M. gelida*, *M. tetranema* or *M. australis* because these species included only one declining or only one stable population. In these cases, the minimum threshold associated with population persistence was a more informative measure of the effect of fragmentation on population status. However, for both *M. tetranema* and *M. australis* only one declining and one stable population of each species occurred. For both species, the median fragment length between declining and extirpated population statuses was used to estimate the minimum threshold necessary for population persistence. These estimates combined with CTA results for remaining species produced minimum fragment lengths ranging 103 to 297 km, below which species were extirpated altogether (Figure 4). When pelagic-spawning cyprinid communities (i.e., all species occurring within a fragment) were considered, and the proportion of species extirpated from communities was regressed against stream fragment length, differential thresholds in persistence contributed to a logistic relationship (Figure 5). This pattern resulted in 100% extirpation of community members within fragments <115 km, variable percentages in extirpation among fragments ranging 115-275 km and no reported extirpations among fragments >275 km in length. In this manner, stream fragmentation explained 71% of community member extirpations within the 57 stream fragments included in analysis (excluding extensively dewatered fragments confounded by immeasurable levels of fragmentation because of temporal variability in stream desiccations; i.e., fragments 33, 40, 41;

Luttrell et al. 1999). In summary, extirpations among eight species of confirmed or suspected pelagic-spawning cyprinids were correlated with reductions in stream fragment length.

Objective 3: Methods

We conducted a cross-fragment analysis of flow regimes to determine primary components associated with the decline and extirpation of pelagic-spawning cyprinid populations. Because these results might be confounded by effects of fragmentation, specifically in shorter fragments (i.e., <115 km) where we observed 100% extirpation of all species, we removed fragments <115 km in length from flow regime analysis. We justified the exclusion of these fragments based on previously published findings that pelagic-spawning cyprinids cannot persist in stream fragments less than approximately 100 km in length (Dudley and Platania 2007). Additionally, we excluded fragments for which previous reports of extirpations associated with barriers to dispersions were published. This case applied to four fragments upstream of reservoirs where flow regimes were unregulated, and extirpations likely occurred because of interruption of source-sink dynamics (fragments 24, Eberle et al. 1997; 31, Rahel and Thel 2004a; 47, Luttrell et al. 1999; 54, Winston et al. 1991). Among remaining fragments (n = 45), lengths ranged 120-711 km and percent of extirpations ranged 0% to 100%, which provided the potential to assess impacts of flow regime with limited contribution of fragmentation. Contemporary (1969-2009) streamflow data were downloaded from USGS gauges and analyzed using Indicators of Hydrologic Alteration (IHA) software (Richter et al. 1996) to examine 13 flow parameters for each of the 45 fragments (Table 4). Among these fragments, we factored out the influence of stream fragment length by treating the parameter as a covariable and conducting a partial ordination in the form of Principal Components Analysis (using CANOCO software; ter Braak 1989) to determine streamflow parameters that explained the most variation in flow regimes. We then considered historical changes in streamflow parameters with large loadings (a measure of variation explained within multivariate space) and the relationship to percent extirpation among pelagic-spawning cyprinid communities.

Objective 3: Results

The first two Principal Components (PCs) explained 43% of variation in streamflow parameters among stream fragments (Table 4). The first PC explained 28% of variation and represented a flow magnitude gradient, with large loadings from four flow parameters: base flow index, mean annual flow, annual coefficient of variation and number of zero days. The second PC explained 15% of variation and represented primarily a timing and frequency gradient, with large loadings from four flow parameters: low pulse duration, date of max, consistency/predictability and fall rate. Collectively, parameters identified as explaining a large amount of variation in streamflow regime were generally related to magnitude of stream discharge (see Poff et al. 1997). Historical changes in stream discharge, measured as percent of 1938-1968 flows occurring during 1969-2009, ranged 13% to 208% among fragments included in flow analysis. These changes in stream discharge were not related ($F_{1,40}$ = 2.04, P = 0.2) to fragment length, suggesting stream fragmentation had little influence on discharge changes in fragments >115 km in length. However, percent extirpation among pelagic-spawning cyprinid communities was positively correlated (r = 0.36, P = 0.02) with reductions in stream discharge. Furthermore, 83% of stream fragments >100 km in length and containing at least one stable population of pelagic-spawning cyprinid (n = 18) indicated either no notable change or a positive trajectory in base flow index (i.e., a measure of the lower envelope of mean discharge) during

1969-2009 (Table 5). In summary, percent extirpations and reductions in distribution and abundance of Great Plains pelagic-spawning cyprinids were correlated with declining streamflows.

Objective 4: Methods

Given the association between flow magnitude and persistence of pelagic-spawning cyprinid communities, we quantified future changes in streamflow (i.e., discharge) projected under climate change scenarios. Specific values for percent change in evapotranspiration and precipitation differ according to climate change models, and depend upon model assumptions concerning future rates of anthropogenic carbon dioxide emissions, solar variability and volcanic activity (Milly et al. 2005). To address these differences, we used the average projected percent change in streamflow according to 12 climate models developed in accordance with Intergovernmental Panel on Climate Change (IPCC 2007) '20C3M' simulations. We obtained these data from Milly et al. (2005) in the form of a 1-degree global grid illustrating percent change in discharge for the period 2041-2060 relative to the period 1900-1970. The historical period 1900-1970 generally corresponded with our historical period of flows (i.e., 1938-1969) and represents a period prior to drastic modifications by groundwater withdrawls (Gido et al. 2010). Using a Geographic Information System (GIS), we estimated the percent change in discharge projected for the region surrounding each stream fragment included in our study based on raster values obtained from the model developed by Milly et al. (2005; also see methods of Xenopolous and Lodge 2006). This approached yielded an estimate of future change in discharge brought about by changes in climate independent of human alteration of flow for each fragment. We then adjusted observed discharges during the period 1969-2009 according to percent changes in discharge expected under climate change to yield a conservative estimate of discharge during the period 2041-2060. This is a conservative estimate of future discharge because is it assumes human withdrawal of water will remain static relative to withdrawals during the period 1969-2009, although we acknowledge withdrawals may increase before the period 2041-2060. These data were then used to prioritize specific stream fragments where discharge availability might become increasingly stressed in the future, and concern for persistence of pelagic-spawning cyprinids might become elevated.

We prioritized specific Great Plains stream fragments that might require increased conservation management in the future by considering the combined limitations imposed by stream fragmentation, anthropogenic water withdrawals and future declines in discharge associated with climate change. This was done by conducting a relative-rank analysis in which potential for conservation success was projected according to specific values of stream fragmentation and reductions in discharge associated anthropogenic water withdrawal and climate change. For fragment length limitations, we used the grand mean length of stream associated with extirpated (140 km), declining (205 km) and stable (425 km) populations of all pelagic-spawning cyprinids included in our study. Projected success was scaled as unlikely for fragments <140 km in length, low for fragments 140-205 km in length, medium for fragments 205-425 km in length, and high for fragments >425 km in length. Similarly, values for percent reduction in future discharge were used to project success of conservation in terms of reduced discharge associated with anthropogenic water withdrawals and climate change. In this case, we multiplied predicted percent change in discharges for the period 2041-2060 (to address changes brought about by climate change; Milly et al. 2005) with mean annual discharges during the period 1969-2009 (to address changes brought about by anthropogenic water withdrawals) and

estimated percent of historical discharge that might occur in the future. Projected success was scaled as unlikely for fragments with <25% of historical discharge, low for fragments with 25-50% of historical discharge, medium for fragments with 50-75% of historical discharge, and high for fragments with >75% of historical discharge.

Using the relative-rankings described above and considering the combined limitations imposed by fragmentation and reduced discharge, we provided four levels of prioritization of fragments useful in addressing conservation of Great Plains pelagic-spawning cyprinids. Fragments with medium to high projected conservation success rates in terms of fragmentation and flow alteration were designated as high priorities for maintaining current conditions (i.e., high priority). Fragments with low projected conservation success rates in terms of fragmentation, but medium to high projected rates for flow alteration were designated as priorities for maintaining current flow conditions (i.e., flow priority). Fragments with medium to high projected conservation success in terms of fragmentation, but low projected success in terms of flow alteration were designated as priorities for maintaining connectivity (i.e., connectivity priority). Lastly, fragments with unlikely to low projected conservation status in terms of either fragmentation or flow alteration were designated as low priorities in general (i.e., low priority).

Objective 4: Results

Among 42 stream fragments of appropriate longitudinal length with adequate historical data, global climate change models predicted an average increase in mean annual discharge for 13 fragments and an average decrease in mean annual discharge for 29 fragments (Table 5). Percent change in 2041-2060 mean discharge relative to 1900-1970 ranged positively from 1.65% to 5.09% and negatively from –1.66% to –11.9%, and exhibited a notable latitudinal gradient. Fragments in the northern Great Plains region, including the states of Montana, South Dakota and portions of Wyoming, indicated increases in mean discharge, whereas fragments in the central and southern Great Plains regions, including the states of Nebraska, Kansas, New Mexico, Oklahoma and portions of Texas, indicated a general decrease in mean discharge (Figure 6). Furthermore, among 21 fragments that indicated a negative trajectory in base flow index during 1969-2009, 100% will likely experience additional reductions in discharge due to climate change before 2060. Alternatively, of the 21 fragments that indicated an increasing trajectory or no notable change in base flow index during 1969-2009, 62% will likely experience increases in mean discharge associated with climate change before 2060. Consequently, climate change impacts will likely have a strong regional signal, where fragments in dry and drying regions, specifically the central and southern Great Plains, might experience increased imperilment of pelagic-spawning cyprinids because of reductions in streamflow.

Prioritization of fragments for which contemporary stream fragmentation and future changes in climate might dictate increased conservation-oriented management action suggested success of conservation initiatives may vary according to individual fragment characteristics. Although we limited our analysis to fragments >100 km in length, five fragments were <140 km in length and consequently might experience unlikely pelagic-spawning cyprinid conservation success because of fragment length limitations. Remaining fragments were partitioned among low (n = 15), medium (n = 15) and high (n = 7) potentials for successful conservation, but might be limited by magnitude of streamflow because of reductions in discharge (Table 5). Reduced streamflow arising from anthropogenic water withdrawal and climate change suggested future conservation success might be unlikely among five fragments under current withdrawal rates.

Remaining fragments were partitioned among low (n = 9), medium (n = 7) and high (n = 21) projected conservation success rates. Priority listings aligned well with percent extirpations within pelagic-spawning cyprinid communities, specifically for fragments listed as high priorities, which were characterized by <33% extirpation among community members in 100% of occurrences (n = 14). Similarly, 92% of fragments listed as low priorities (n = 13) were characterized by >60% extirpation among community members. Distribution of fragments designated as high priorities for conservation included the northern and southern Great Plains, but central regions were generally devoid of high priority fragments (Figure 7). Instead, many fragments from which pelagic-spawning cyprinids have already become extirpated (i.e., fragments listed as low priorities for conservation), were distributed throughout the central Great Plains.

Significance

Pelagic-spawning cyprinid communities inhabiting fragmented streams throughout the Great Plains represent a disappearing guild of fishes, as evidenced by high imperilment rates and conservation listings at state (e.g., Haslouer et al. 2005, Hubbs et al. 2008), regional (e.g., Rahel and Thel 2004a, 2004b) and national (e.g., Jelks et al. 2008) levels. Reported reductions in abundance and distribution include extirpation from 45% of historical range in the Missouri River Basin (sturgeon chub, Rahel and Thel 2004a), 55% of historical range in the Arkansas River basin (shoal chub, Luttrell et al. 1999), 80% of historical range for the Arkansas River shiner (Wilde 2002) and 90% of historical range for peppered chub (Luttrell et al. 1999). Our findings supported extirpation from a majority (i.e., >50%) of fragments included in this study for the flathead chub (61%), silver chub (64%) and sturgeon chub (75%), and values that closely match previously reported extirpation rates for the Arkansas River shiner (79%) and peppered chub (88%). Similar extirpations have occurred among six species of pelagic-spawning cyprinids in the Rio Grande and Pecos River basins of New Mexico and Texas, where two guild members are now extinct and remaining members are restricted to unfragmented river reaches (Platania and Altenbach 1998, Dudley and Platania 2007). Two species of pelagic-spawning cyprinids endemic to the Brazos River of Texas are now restricted to approximately a third of their historical range because of stream fragmentation and associated effects of reservoirs (Durham and Wilde 2009a). These reported patterns of decline transcend a large spatial scale (i.e., the entire Great Plains), include multiple levels of phylogeny (i.e., 4 genera, 16 species, 2 subspecies; Platania and Altenbach 1998, Durham and Wilde 2009a, this study), span 13 North American ecoregions, and collectively include 8% of the imperiled freshwater cyprinids in North America (Jelks et al. 2008). Consequently, pelagic-spawning cyprinids represent a substantial challenge for conservation of biodiversity in North America.

A growing body of literature suggests imperilment of pelagic-spawning cyprinid species is a direct consequence of stream fragmentation. Winston et al. (1991) reported the extirpation of plains minnow and prairie chub in fragmented upper reaches of the North Fork Red River of Oklahoma. Since, authors have reported the extirpation of pelagic-spawning cyprinids from upstream reaches of fragmented streams in Texas (Wilde and Ostrand 1999), New Mexico (Pittenger and Schiffmiller 1997), Oklahoma (Luttrell et al. 1999), Kansas (Eberle et al. 2002), Missouri (Eisenhour 2004) and Colorado (Rahel and Thel 2004b). This pattern is likely driven by impoundments acting as barriers to dispersal and precluding upstream recolonization by adult individuals following localized extinctions (Luttrell et al. 1999). Additional support for this conclusion exists from the Canadian River of New Mexico and Texas and the Pecos River of

New Mexico, where adult Arkansas River shiner and plains minnow were recorded stockpiling at upstream barriers (Bonner 2000, Hoagstrom et al. 2010b). In a similar fashion, instream barriers and impoundments act to preclude downstream dispersion of drifting eggs and pre-larvae during development. For example, within the Rio Grande and Pecos River basins of New Mexico and Texas, pelagic-spawning cyprinids are now extirpated from stream fragments with <100 km between instream barriers. These extirpations were likely associated with reduced distances available for development from drifting pre-larval to non-drifting larval stages, resulting in high mortality within downstream impoundments (Moore 1944, Platania and Altenbach 1998, Dudley and Platania 2007). Within the Great Plains, we found estimated minimum thresholds in fragment length varied among eight species, but were consistently >100 km in length. Suspected pelagic-spawning shoal chub exhibited the shortest threshold in longitudinal length (103 km), which was consistent with Platania and Altenbach's (1998) conclusion that the speckled chub *Macrhybopsis aestivalis* (once synonymous with shoal chub, Eisenhour 2004) require relatively shorter longitudinal stream lengths for completion of life history. Similarly, our estimated minimum thresholds for Arkansas River shiner and peppered chub (217 and 205 km, respectively) were consistent with Bonner and Wilde's (2000) conclusion that the Canadian River between Ute and Meredith reservoirs (220 km) represents the near minimum length required for completion of life history. We found percent of extirpated populations among eight species of suspected or confirmed pelagic-spawning cyprinids was positively correlated with estimated minimum thresholds in fragment length, strongly suggesting stream fragmentation has played a role in observed declines in abundance and distribution. Our results contradict the findings of Widmer et al. (2010) and Medley et al. (2007) who suggest that given the appropriate habitat complexity, reproduction of pelagic spawning fishes is possible in stream fragments <100 km. However, the above studies were based only on retention of artificially manufactured eggs and did not consider the many factors that long stream fragments can play in the success of these species. In summary, imperilment associated with stream fragmentation provides a parsimonious mechanism that links widely dispersed literature accounts of pelagic-spawning cyprinid declines and explains over 70% of variation in extirpation among eight highly imperiled Great Plains fishes.

Although fragmentation alone explained most of the variation associated with Great Plains pelagic-spawning cyprinid declines, reduction in discharge was correlated with extirpations among stream fragments >100 km in length. Reproductive success of pelagic-spawning cyprinids is dependent on stream discharge to initiate spawning (Durham and Wilde 2006, 2009a) and to retain eggs in suspension long enough for hatching (Moore 1944, Bottrell et al. 1964). Discharge can also influence recruitment of larval fishes. For example, age-0 *M. tetranema* and smalleye shiner *Notropis buccula* constitute the highest contribution to population growth rate, and survival of this life stage is dependent upon stream discharge to the point that even 5% reductions in contemporary discharges might cause up to 85% reduction in population size (Wilde and Durham 2008, Durham and Wilde 2009b). Throughout the Great Plains, we found extirpation of pelagic-spawning cyprinids occurred to the highest extent in the central and southern Great Plains regions, where notable reductions in discharge have occurred since at least the 1970s (Cross et al. 1985, Pigg 1987, 1991, Gido et al. 2010). Furthermore, the central and southern Great Plains regions included stream fragments created by complete stream desiccation for a majority of the year—in this case reductions in discharge likely contributed to declines and extirpations by inducing both fragmentation and negative effects on reproductive success. Among extensively dewatered stream fragments >140 km in length, long enough to support at

least declining populations of pelagic-spawning cyprinids, the majority of community members (i.e., >66%) were extirpated in 73% of occurrences when stream discharges were reduced by at least half (n=11). Consequently, the possibility exists for reductions in discharge, both related to anthropogenic withdrawal and climate change, to contribute to declines and extirpations among Great Plains pelagic-spawning cyprinids (see Taylor 2010). This finding is consistent with the findings of a recent large-scale literature review that found alteration to magnitude of discharge was detrimental to many fluvial organisms, notably fishes (Poff and Zimmerman 2010).

Although our findings were useful in explaining a large amount of variation in extirpation of imperiled pelagic-spawning Great Plains fishes, additional factors exist that should be considered for future conservation. Eighteen upstream barriers included in this study were hydroelectric or deep-storage reservoirs capable of manipulating flow regime in downstream reaches. These impoundments carry both consequences and potential management options for pelagic-spawning communities occurring immediately downstream. Reservoirs that release water from the hypolimnion contribute to cooler tail-water temperatures, and effects might extend downstream on the order of kilometers (Edwards 1978). Development rates of drifting eggs and pre-larvae are prolonged during cooler water temperatures, contributing to further downstream transport before larval stages are reached (Dudley and Platania 2007). Similarly, sustained high flows associated with reservoir releases contribute to increased downstream transport through homogenization of habitat (e.g., deep, incised channels) and increased rate of flow (Dudley and Platania 2007). Our analysis did not include measurements of water temperature or channel morphology, two factors that might be manipulated more easily than removal of large impoundments or diversion dams to facilitate pre-larval development within stream fragments (e.g., Widmer et al. 2010). However, our findings across a diversity of streams with varying channel morphologies suggest that fragment lengths of at least 100 kilometers are likely necessary regardless of channel morphology characteristics, indicating that increased connectivity is the ideal method of mitigation. A management option associated with upstream reservoirs is that flows might be deliberately managed to promote recruitment of native fishes (e.g., Propst and Gido 2004). In terms of pelagic-spawning cyprinids, this might include providing high flow pulses that act as spawning cues and provide additional instream habitat for larval development (Taylor and Miller 1990). High flow pulses are known to synchronize otherwise asynchronous spawning and increase spawning intensity among pelagic-spawning cyprinids and might provide an effective method for targeting increased recruitment (Durham and Wilde 2008b). As such, the potential exists for indispensable reservoirs to contribute to conservation of the very species they now imperil, especially if water releases are conducted in concert with storm events or sheetflow steaming from tributary contributions. These conservation options are likely only feasible in fragments with sufficient length, and among remaining fragments decommissioning hazardous or unnecessary impoundments during re-licensing will allow for increased connectivity (e.g., Marks et al. 2009).

Future mitigation approaches that enhance conservation delivery for Great Plains pelagic-spawning cyprinids will likely require the restoration of natural connectivity in the system. Current fragmentation management practices involved the use of fishways that allow passage in an upstream direction for a wide range of fishes (e.g., Schmetterling et al. 2002). However, a paucity of empirical data exists pertaining to the passage of small-bodied cyprinids through fishways, though existing evidence suggests passage is possible (Laine et al. 1998, Prchalova et al. 2006). The greater challenge will ultimately involve the downstream passage of drifting eggs and pre-larvae, especially through large reservoirs (Agostinho et al. 2007). The uncertainty

associated with allowing downstream passage through reservoirs strengthens the necessity for preserving unfragmented stream reaches that support intact fish communities (e.g. Hoagstrom et al. 2010a). We identified 14 stream fragments of appropriate length with relatively little contemporary and projected declines in discharge that support >67% of historical pelagic-spawning cyprinid communities. Among these, the lower Platte River of Nebraska and the Ninnescah River and associated portion of the Arkansas River of Kansas supported four species with declining or stable status. These fragments are of particular importance because they represent strong-holds of species within respective state boundaries, and in the case of the Ninnescah River, support 50% of the remaining populations of *M. tetranema* (Luttrell et al. 1999). Proposed reservoir construction has the potential to increase fragmentation within remaining large stream fragments and, depending upon location of construction, might cause further declines and extirpations (e.g., Durham and Wilde 2009a). Accordingly, conservation of imperiled stream-dwelling fishes will ultimately require trade-offs between ecological needs of streams and rivers and the perceived human needs associated with freshwater resources (Richter et al. 2003, Lytle and Poff 2004, Taylor et al. 2008).

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Table 1. Identification number, location (i.e., state), description and longitudinal length (km) of Great Plains stream fragments inhabited by pelagic-spawning cyprinid communities.

Fragment	Location	Fragment Description	Length (km)
1	MT/ND	Yellowstone River downstream of Intake Dam (length between Fort Peck Dam and upper reaches of Lake to Sakakawea)	327
2	MT	Yellowstone River between Cartersville Dam and Intake Dam	266
3	SD	Mainstem Grand River of South Dakota	256
4	SD	Mainstem Monroe River of South Dakota	387
5	SD	Cheyenne River between Angostura Dam and upper reaches of Lake Oahe	395
6	SD	Mainstem Bad River of South Dakota	184
7	SD	Mainstem White River of South Dakota	705
8	NE	Niobrara River between Box Butte Dam and Spencer Dam	445
9	NE	Niobrara River between Spencer Dam and upper reaches of Lewis and Clark Lake	65
10	WY	North Platte River between Alcova Dam and upper reaches of Glando Reservoir	228
11	WY	North Platte River between Glendo Dam and upper reaches of Guernsey Reservoir	46
12	WY	North Platte River between Guernsey Dam to WY/NE divsersion dam	96
13	NE	North Platte River betweenWY/NE diversion dam and upper reaches of McConaughy Reservoir	198
14	NE	North Platte River between Kingsley Dam and Diversion dam at North Platte, NE	96
15	NE	Platte River North Platte to wier dam near Elm Creek, NE	133
16	NE	Platte River between weir dam near Elm Creek, NE and Colombus, NE	217
17	KS/NE	Republican River between dam at Bonny, CO and upper reaches of Swanson Reservoir	136
18	NE	Republican River between Trenton Dam and upper reaches of Harlan County Reservoir	181
19	NE	Republican River between Harlan County Dam and upper reaches of Milford Reservoir	332
20	KS	Kansas River between Milford Dam and Bowersock Dam	177
21	KS	Big Blue River between Marysville Dam and upper reaches of Tuttle Creek Reservoir	66
22	KS	Delaware River between Mission Lake Dam and upper reaches of Perry Lake	61
23	MO	Osage River upstream of upper reaches of Truman Reservoir	85
24	NE	North Fork Solomon River upstream of upper reaches of Kirwin Reservior	109
25	NE	North Fork Solomon River between Kirwin Dam and upper reaches of Waconda Reservoir	93
26	KS	South Fork Solomon River between Hoxie, KS and upper reaches of Webster Reservoir	90
27	KS	South Fork Solomon River between Webster Dam and upper reaches of Waconda Reservoir	134
28	KS	Saline River upstream of upper reaches of Wilson Reservoir	189
29	KS	Smokey Hill River between Wallace County Kansas and upper reaches of Cedar Bluff Reservoir	173
30	KS	Smokey Hill River between Cedar Bluff Dam and upper reaches of Kanopolis Reservoir	222
31	CO	Arkansas River between Salida, CO and dam at Florence, CO	119
32	CO/KS	Arkansas River between John Martin Dam and Lakin, KS	179

Table 1 continued.

Fragment	Location	Fragment Description	Length (km)
33*	KS	Arkansas River between Lakin, KS and Great Bend, KS	290
34	KS	Arkansas River between Great Bend, KS and wier dam at Wichita, KS	178
35	KS/OK	Arkansas River between wier dam at Wichita, KS and upper reaches of Kaw Reservoir	153
36	KS/OK	Ninnescah River and portion of Arkansas River upstream of upper reaches of Kaw Lake	251
37	OK	Arkansas River between Kaw Dam and upper reaches of Keystone Lake	120
38	KS/OK	Mainstem Medicine Lodge River upstrem of upper reaches of Great Salt Plains Lake	165
39	KS/OK	Mainstem Salt Fork Arkansas River upstream of upper reaches of Great Salt Plains Lake	163
40*	KS/OK	Cimarron River between Castaneda, OK and just East of Liberal, KS	277
41*	KS/OK	Cimarron River between just East of Liberal, KS and Keystone Lake	434
42	OK	North Canadian River between Pony Creek confluence and Optima Dam	38
43*	OK	North Canadian River between Optima Dam and Fort Supply (Wolf Creek Confluence)	191
44	OK	North Canadian River between Fort Supply and upper reaches of Canton Lake	139
45	OK	North Canadian River between Canton Dam and Overholser Dam, Oklahoma City	161
46	OK	North Canadian River between Overholser Dam, Oklaoma City and upper reaches of Urika Reservoir	339
47	OK	Deep Fork River upstream of upper reaches of Lake Eufaula	183
48	NM	South Canadian River upstream of upper reaches of Conchas Lake	180
49	NM	Ute Creek between Gladstone, NM to upper reaches of Ute Reservoir	189
50	NM/TX	South Canadian River between Ute Dam and upper reaches of Lake Meredith	220
51*	TX/OK	South Canadian River between Sanford Dam and Roger Mills County, OK	214
52	OK	South Canadian River between Roger Mills County, OK and Urika Reservoir	462
53	OK	Washita River upstream of upper reaches of Foss Reservoir	93
54	OK	North Fork of the Red River upstream of upper reaches of Altus Reservoir	108
55	TX/OK	Upper Red River between Prairie Dog Town Fork and upper reaches of Lake Texoma	455
56	OK/LA	Red River between Denison Dam and Dam at Shrevport, LA	689
57	TX	North Fork Wichita River between Truscott, TX and upper reaches of Lake Kemp	149
58	TX	Brazos River between McMillan Dam (on Double Mountain Fork) and upper reaches of Possum Kingdom Reservoir	616
59	TX	Brazos River between Morris Sheppard Dam to upper reaches of Lake Waco	171
60	TX	Brazos River downstream of Whitney Dam to Gulf of Mexico	711

*Asterisks denote fragments associated with stream desiccations

Table 2. Community composition, species status (E = extirpated, D = declining, S = stable) and data sources for pelagic-spawning cyprinid communities in 60 Great Plains stream fragments (see Table 1 for description and location of fragments). Labels follow first three letters of genus and species, *N* denotes rare occurrence of a species within a fragment.

Frag	Hyb pla	Pla gra	Mac gel	Mac sto	Mac hyo	Mac tet	Not gir	Mac aus	Citations
1	S	D	D	-	-	-	-	-	18, 19, 34
2	S	S	E	-	-	-	-	-	18, 19, 34
3	S	S	E	-	-	-	-	-	29, 30, 34
4	S	S	-	-	-	-	-	-	30, 34
5	S	S	S	-	-	-	-	-	30, 31, 34
6	S	S	-	-	-	-	-	-	29, 30, 34
7	S	S	S	-	-	-	-	-	30, 34
8	-	S	-	-	-	-	-	-	26, 29
9	E	E	E	E	-	-	-	-	11, 26, 34
10	S	D	-	-	-	-	-	-	12, 33
11	E	E	E	-	-	-	-	-	11, 26, 34
12	E	E	E	-	-	-	-	-	11, 26, 34
13	S	-	E	-	-	-	-	-	11, 12, 34
14	E	E	-	-	-	-	-	-	12, 34
15	D	E	E	-	D	-	-	-	11, 14, 26, 34
16	S	D	E	D	D	-	-	-	11, 14, 26, 34
17	-	E	-	-	-	-	-	-	4, 26
18	-	E	-	E	D	-	-	-	4, 11, 34
19	S	D	-	D	S	-	-	-	4, 26, 34
20	E	E	E	E	E	-	-	-	4, 11, 34
21	-	-	-	-	E	-	-	-	23
22	-	E	-	E	E	-	-	-	4
23	-	-	-	E	E	-	-	-	16, 20
24	E	-	-	-	-	-	-	-	4, 22, 34
25	E	-	-	-	-	-	-	-	4, 22, 34
26	E	-	-	-	-	-	-	-	4, 22, 34
27	E	-	-	-	-	-	-	-	4, 22, 34
28	E	-	-	E	-	-	-	-	4, 15, 34
29	D	-	-	-	-	-	-	-	15, 34
30	D	-	-	-	-	-	-	-	15, 34
31	-	E	-	-	-	-	-	-	26
32	-	E	-	-	-	E	-	-	5, 20, 34
33	E	E	-	-	-	E	E	-	5, 20, 27, 28, 34
34	E	E	-	E	-	E	E	-	5, 20, 27, 28, 34
35	D	-	-	E	E	-	E	-	5, 20, 27, 28, 34
36	D	-	-	D	-	S	D	-	4, 28, 34
37	D	-	-	E	S	E	E	-	20, 27, 25, 34
38	D	-	-	-	-	E	E	-	5, 34
39	E	-	-	-	-	E	E	-	5, 34
40	E	E	-	-	-	E	E	-	5, 34
41	D	-	-	-	-	E	D	-	5, 20, 34
42	E	-	-	-	-	E	E	-	7, 9, 20, 34
43	D	-	-	-	-	E	E	-	7, 9, 20, 34
44	D	-	-	-	-	E	E	-	7, 9, 20, 34
45	D	-	-	-	-	E	E	-	7, 9, 20, 34

Table 2 continued.

Frag	*Hyb pla*	*Pla gra*	*Mac gel*	*Mac sto*	*Mac hyo*	*Mac tet*	*Not gir*	*Mac aus*	Citations
46	-	-	-	-	-	-	*N*	-	9, 34
47	-	-	-	-	E	-	E	-	9, 20, 34
48	E	E	-	-	-	-	E	-	8, 34
49	D	E	-	-	-	E	E	-	17, 34
50	D	E	-	-	-	D	D	-	21, 33, 34
51	D	*N*	-	-	-	E	E	-	1, 21, 34
52	S	-	-	-	S	*N*	S	-	1, 9, 20, 25, 34
53	-	-	-	-	-	-	-	E	24, 34
54	E	-	-	-	-	-	-	E	10, 34
55	S	-	-	-	-	-	-	S	13, 34
56	S	-	-	S	S	-	-	-	25, 34
57	D	-	-	-	-	-	-	D	2, 34
58	S	-	-	-	S	-	-	-	2, 34
59	E	-	-	-	D	-	-	-	3
60	-	-	-	S	S	-	-	-	32

1: Gene Wilde, Texas Tech University, unpublished data; **2**: Fran Gelwick, Texas A&M University, unpublished data; **3**: Jack Davis, Brazos River Authority, unpublished data; **4**: GAP Database KSU; **5**: Cross *et al.* (1985); **6**: Cross and Moss (1987); **7**: Pigg (1987); **8**: Sublett *et al.* (1990); **9**: Pigg (1991); **10**: Winston *et al.* (1991); **11**: Hesse *et al.* (1993); **12**: Lynch and Roh (1996); **13**: Taylor *et al.* (1996); **14**: Chadwick *et al.* (1997); **15**: Eberle *et al.* (1997); **16**: Pflieger (1997); **17**: Pittenger and Schiffmiller (1997); **18**: Patton *et al.* (1998); **19**: Helfrich *et al.* (1999); **20**: Luttrell *et al.* (1999); **21**: Bonner and Wilde (2000); **22**: Eberle *et al.* (2002); **23**: Gido *et al.* (2002); **24**: Eisenhour (2004); **25**: Miller and Robison (2004); **26**: Rahel and Thel (2004a); **27**: Rahel and Thel (2004b); **28**: Haslouer *et al.* (2005); **29**: Hoagstrom *et al.* (2006); **30**: Hoagstrom *et al.* (2007a); **31**: Hoagstrom *et al.* (2007b); **32**: Runyan (2007); **33:** Wilde and Durham (2008); **34**: Hoagstrom *et al.* (2010a).

Table 3. Percent occurrence of population statuses within relative fragment lengths identified using Classification Tree Analysis (short, intermediate, long), minimum threshold in fragment length for population persistence, and Cohen's Kappa and P-values associated with models for changes in status (Status) and population persistence (Threshold) for eight species of Great Plains pelagic-spawning cyprinids. Bolded values denote the majority (i.e., >50%) of population statuses associated with relative fragment lengths. Classification Tree Analysis could not be conducted for *M. tetranema* or *M. australis* because of limited occurrences of declining and stable populations, see text for method of assigning threshold values.

Species/Status	Relative Fragment Length (km)			Minimum Threshold	Model	Cohen's Kappa	P-value
	Short	Intermediate	Long				
H. placitus	<115	115-254	>254	115 km	Status	0.77	<0.01
Extirpated	**56%**	44%	-		Threshold	0.81	<0.01
Declining	-	**100%**	-				
Stable	-	27%	**73%**				
N. girardi	<217	-	>217	217 km	Status	0.8	<0.01
Extirpated	**100%**	-	-		Threshold	0.77	0.01
Declining	-	-	**100%**				
Stable	-	-	**100%**				
M. gelida	<297	-	>297	297 km	Status	0.79	0.01
Extirpated	**100%**	-	-		Threshold	0.79	0.01
Declining	-	-	**100%**				
Stable	-	-	**100%**				
M. tetranema	<205	-	>205	205 km	Status	N/A	N/A
Extirpated	**100%**	-	-		Threshold	N/A	N/A
Declining		-	**100%**				
Stable		-	**100%**				
P. gracilis	<224	224-360	>360	183 km	Status	0.79	<0.01
Extirpated	**100%**	-	-		Threshold	0.85	<0.01
Declining	25%	**75%**	-				
Stable	14%	29%	**57%**				
M. hyostoma	<103	103-143	>143	103 km	Status	0.8	<0.01
Extirpated	**50%**	50%	-		Threshold	0.75	<0.01
Declining	-	**100%**	-				
Stable	-	-	**100%**				
M. storeriana	<203	203-511	>511	203 km	Status	0.8	<0.01
Extirpated	**100%**	-	-		Threshold	1	<0.01
Declining	-	**100%**	-				
Stable	-	-	**100%**				
M. australis	<128	-	>128	128 km	Status	N/A	N/A
Extirpated	**100%**	-	-		Threshold	N/A	N/A
Declining	-	-	**100%**				
Stable	-	-	**100%**				

Table 4. Flow component, Indicators of Hydrologic Alteration (IHA) parameter code, data transformation and loadings along principal component (PC) I and II for flow regimes analyzed among 45 Great Plains prairie stream fragments. Fragments with length <100 km (i.e., fragments 9, 11, 12, 14, 21, 22, 23, 25, 26, 42, 53) or for which extirpations associated with barriers to dispersion are documented (i.e., 24, 31, 47, 54) were removed from flow regime analysis.

Flow Component[1]	IHA Parameter[2]	Transformation	PC I (28%)	PC II (15%)
Duration	High pulse duration	$Log_{10}(X+1)$	-0.87	-0.31
Duration	Low pulse duration	$Log_{10}(X+1)$	-0.89	-1.10
Frequency	Number of zero days	$Log_{10}(X+1)$	1.42	1.20
Frequency	High low frequency	$Log_{10}(X+1)$	0.75	-0.43
Frequency	Extreme low flow frequency	$Log_{10}(X+1)$	0.87	1.06
Magnitude	Mean annual flow	$Log_{10}(X+1)$	-1.39	0.63
Magnitude	Annual coefficient of variation	$Log_{10}(X+1)$	1.68	-0.42
Magnitude	Base flow index	Arcsine(sqrt(X))	-1.44	1.10
Rate of change	Number of reversals	$Log_{10}(X+1)$	-1.19	-0.40
Rate of change	Fall rate	$Log_{10}(X*-1+1)$	-0.29	1.32
Timing	Consistency/Predictability	Arcsine(sqrt(X))	-0.11	2.25
Timing	Date of maximum	$Log_{10}(X+1)$	0.55	0.66
Timing	Date of minimum	$Log_{10}(X+1)$	0.68	-0.87
Area	Stream fragment length	$Log_{10}(X+1)$	0.00	0.00

[1]Flow components follow Poff et al. 1997, except area
[2]Parameter names follow IHA listings (see Richter et al. 1996), except fragment length

Table 5. Fragment number, associated USGS gauge and range of historical data, historical, contemporary and projected future discharges (Mean Q; m^3 sec^{-1}) and percent change in discharge related to climate change (Δ Mean Q) for 42 Great Plains stream fragments >100km in length (fragments 15, 32 and 52 were removed because of data availability).

Fragment Number	USGS Gauge	Historical Range	Mean Q (m^{3}sec^{-1})			Δ Mean Q (%)[4]
			1938-1968[1]	1969-2009[2]	2041-2060[3]	
1	6177000	1938-1968	260.30	275.1 (↑)	284.18	3.49
2	6309000	1938-1968	324.60	320.1 (-)	336.62	5.09
3	6357800	1958-1968	4.16	7.81 (↑)	7.92	2.7
4	6360500	1954-1968	3.95	8.23 (↑)	8.34	2.7
5	6438500	1951-1968	16.58	23.79 (↑)	24.36	3.46
6	6441500	1938-1968	4.91	4.66 (-)	4.79	2.7
7	6452000	1938-1968	16.68	16.51 (-)	16.79	1.65
8	6461500	1946-1968	23.41	20.77 (-)	21.16	1.65
13	6652000	1958-1968	36.64	37.28 (-)	36.55	-1.98
14	6674500	1938-1968	14.36	19.22 (↑)	18.98	-1.66
16	6770500	1938-1968	25.38	41.43 (↑)	40.60	-3.26
17	6827500	1938-1968	1.37	0.43 (↓)	0.34	-6.69
18	6843500	1946-1968	8.66	3.72 (↓)	3.58	-1.66
19	6856000	1946-1968	23.40	13.82 (↓)	12.74	-4.62
20	6887500	1938-1968	154.70	155.6 (-)	158.15	1.65
27	6874000	1946-1968	4.34	2.2 (↓)	2.00	-4.62
28	6867000	1946-1968	3.94	2.05 (↓)	1.79	-6.69
29	6860000	1940-1968	1.20	0.16 (↓)	0.08	-6.69
30	6865500	1941-1968	9.57	7.02 (↓)	6.58	-4.62
33	7139000	1938-1968	5.68	2.97 (↓)	2.29	-11.9
34	7141300	1941-1968	11.51	3.76 (↓)	3.23	-4.62
35	7146500	1938-1968	58.08	63.01 (↑)	64.64	2.8
36	7145500	1938-1968	11.59	12.81 (-)	11.75	-9.17
37	7152500	1938-1968	114.10	151.3 (↑)	154.49	2.8
38	7149000	1938-1968	3.02	3.77 (↑)	3.49	-9.17
39	7148400	1938-1951	3.66	2.97 (↓)	2.63	-9.17
40	7154500	1951-1968	0.64	0.23 (↓)	0.15	-11.9
41	7158000	1938-1968	10.59	6.19 (↓)	5.22	-9.17
43	7234000	1938-1968	2.79	0.5 (↓)	0.17	-11.9
44	7238000	1947-1968	5.82	3.59 (↓)	3.06	-9.17
45	7239000	1938-1968	5.04	3.02 (↓)	2.91	-2.11
46	7242000	1938-1968	19.16	26.82 (↑)	27.14	1.65
48	7221500	1938-1968	4.95	2.83 (↓)	2.49	-6.95
49	7226500	1942-1968	0.69	0.3 (↓)	0.26	-5.23
50	7227500	1938-1968	11.27	4.24 (↓)	3.65	-5.23
51	7228000	1938-1968	11.65	1.94 (↓)	1.39	-4.76
55	7315500	1938-1968	70.67	71.45 (-)	69.96	-2.11
56	7337000	1938-1968	337.20	401.6 (↑)	414.14	3.72
57	7311900	1960-1968	4.46	3.07 (↓)	2.92	-3.43
58	8082500	1938-1969	10.78	7.54 (↓)	7.25	-2.71
59	8093100	1939-1968	47.42	59.08 (↑)	57.79	-2.71
60	8111500	1939-1968	191.60	200.2 (↑)	191.81	-4.38

[1]Values calculated for 1938-1968 when data were available
[2]Trajectories assigned according to slope of base flow index calculated using Indicators of Hydrologic Alteration.
[3]Calculated as product of mean Q 1969-2009 and Δ mean Q (from Milly *et al.* 2005)
[4]Percent difference in mean Q between periods 1900-1970 and 2041-2060 (Milly *et al.* 2005)
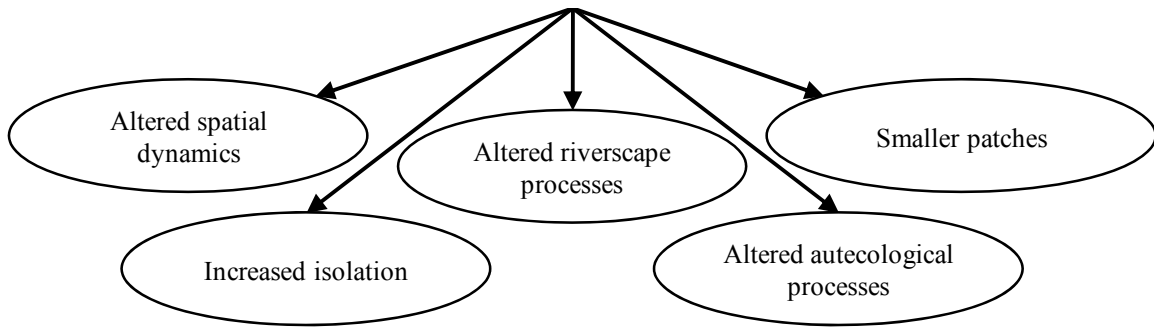
Table 6. Fragment number, percent community extirpation, projected conservation success according to contemporary stream fragmentation and future climate change, and priority listing for 42 Great Plains stream fragments >100 km in longitudinal length. See text for further description of priority listings.

Fragment Number	Percent Extirpated (%)	Projected Conservation Success		Priority Listing
		Fragmentation[1]	Climate Change[2]	
1	0	Medium	High	High
2	33	Medium	High	High
3	33	Medium	High	High
4	0	Medium	High	High
5	0	Medium	High	High
6	0	Low	High	Flow
7	0	High	High	High
8	0	High	High	High
13	50	Low	High	Flow
14	100	Unlikely	High	Low
16	20	Medium	High	High
17	100	Unlikely	Unlikely	Low
18	67	Medium	Low	Connectivity
19	0	Medium	Medium	Connectivity
20	100	Low	High	Flow
27	100	Unlikely	Low	Low
28	100	Low	Low	Low
29	0	Low	Unlikely	Low
30	0	Medium	Medium	Connectivity
33	100	Medium	Low	Connectivity
34	100	Low	Low	Low
35	75	Low	High	Flow
36	0	Medium	High	High
37	60	Unlikely	High	Low
38	67	Low	High	Flow
39	100	Low	Medium	Flow
40	100	Medium	Unlikely	Low
41	33	High	Low	Connectivity
43	67	Low	Unlikely	Low
44	67	Unlikely	Medium	Low
45	67	Low	Medium	Flow
46	0	Medium	High	High
48	100	Low	Low	Low
49	75	Low	Low	Low
50	25	Medium	Low	Connectivity
51	67	Medium	Unlikely	Low
55	0	High	High	High
56	0	High	High	High
57	0	Low	Medium	Flow
58	0	High	Medium	High
59	50	Low	High	Flow
60	0	High	High	High

[1]Projected conservation success ranked according to fragment lengths (Unlikely = <140 km, Low = 140-205 km, Medium = 205-425 km, High = >425km)
[2]Projected conservation success ranked according to percent 1900-1970 flows occurring during 2041-2060 (Unlikely = <25%, Low = 25-50%, Medium = 50-75%, High = >75%)

Impacts of riverscape fragmentation on stream fishes

Altered spatial dynamics

Altered riverscape processes

Smaller patches

Increased isolation

Altered autecological processes

Corresponding impacts of climate change on aquatic systems*

Altered streamflow regimes / increased water development

Altered streamflow & thermal regimes / reduced ice cover

Altered streamflow regimes / increased water development

Altered streamflow regimes / increased water development

Altered streamflow & thermal regimes / increased salinity
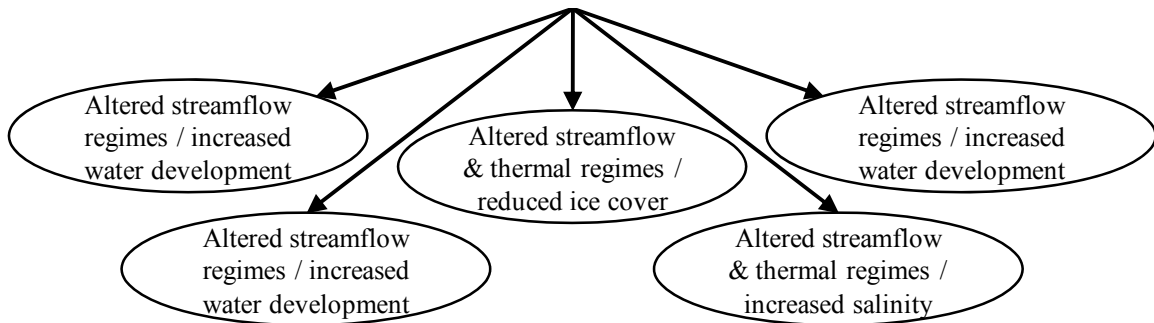
Figure 1. Aspects of stream-dwelling fish ecology that are impacted by riverscape fragmentation and associated aspects of climate change that might exacerbate impacts. Impacts of climate change on aquatic systems are from Rahel and Olden (2008).
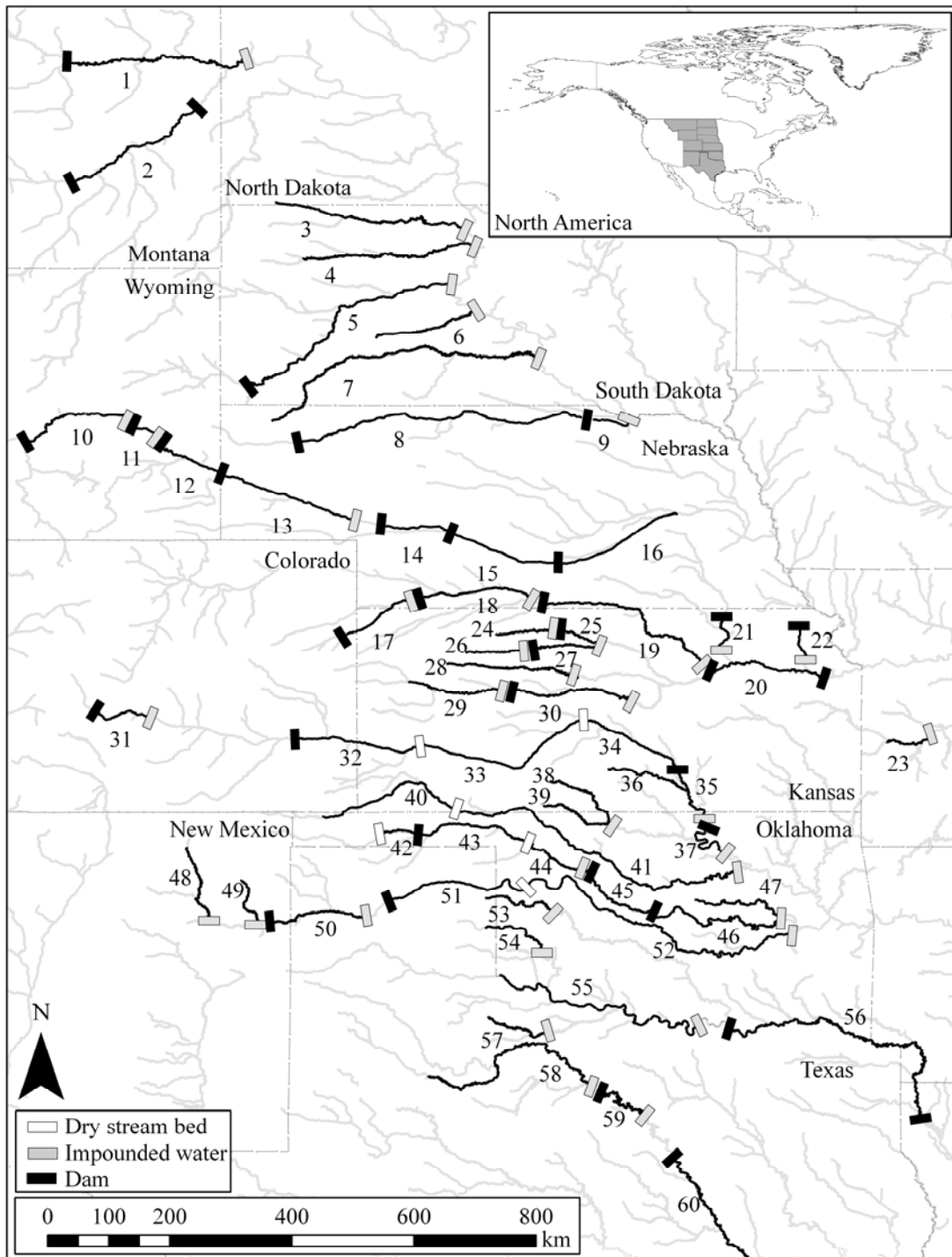
Figure 2. Distribution of North American Great Plains stream fragments included in analyses. Fragment numbers correspond with descriptions in Table 1, rectangles represent fragmenting factors: dry stream beds (white), impounded waters (grey) and dams associated with water diversions and impoundments (black).
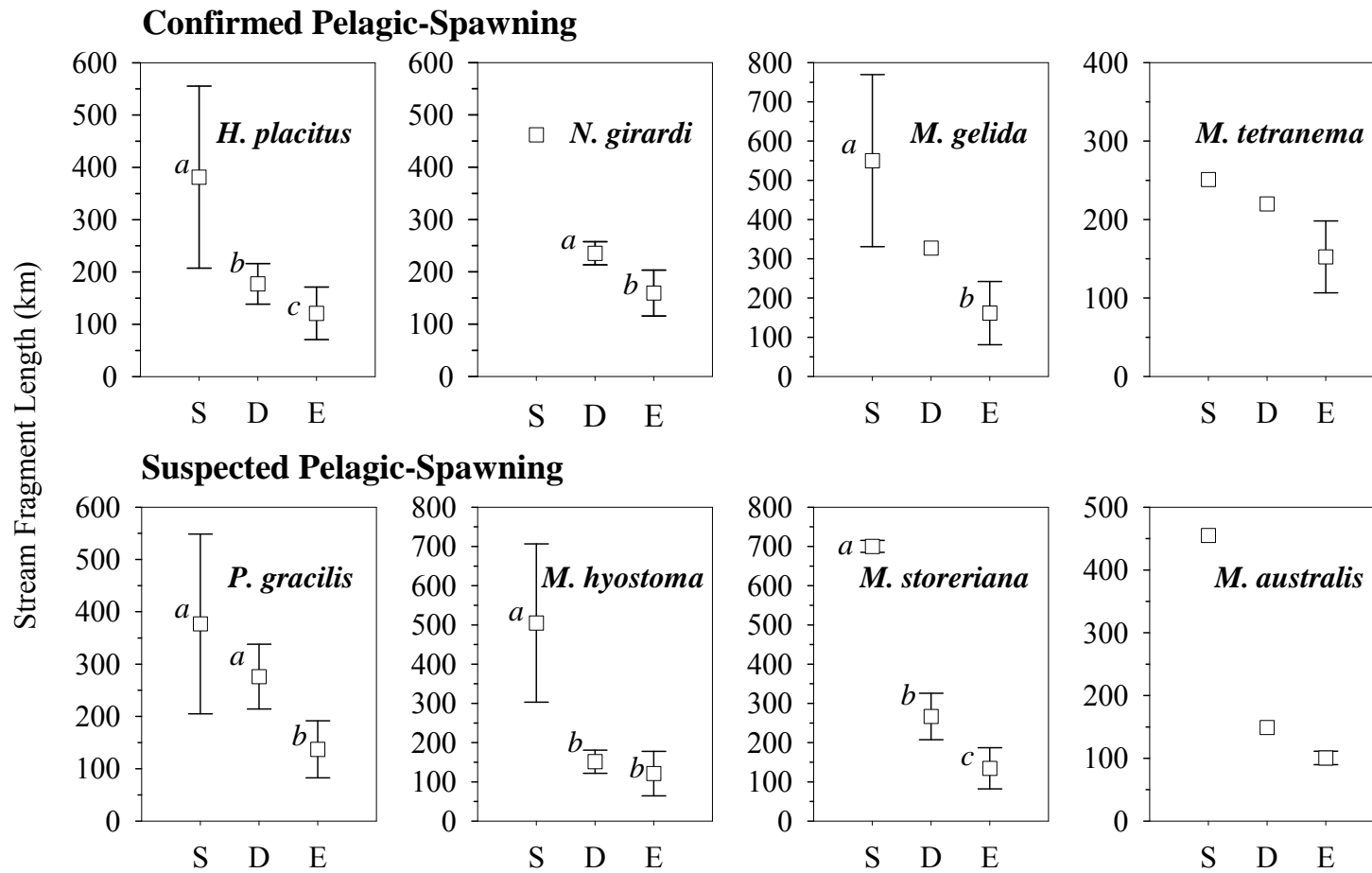
Figure 3. Mean (±SD) stream fragment lengths for confirmed (upper row) and suspected (lower row) Great Plains pelagic-spawning cyprinid populations according to conservation status: stable (S), declining (D) and extirpated (E). Lowercase letters represent statistical differences within a species ($\alpha = 0.05$; see text for statistical procedures).
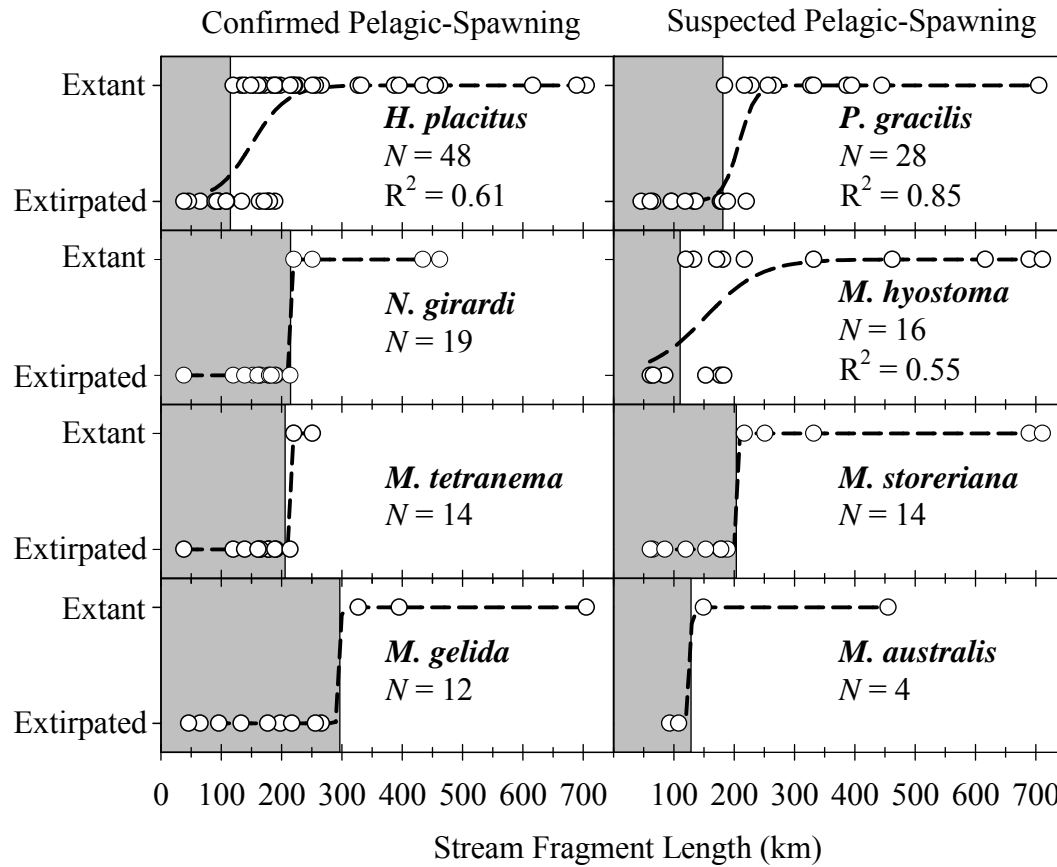
Figure 4. Distributions of extant and extirpated populations of confirmed (left column) and suspected (right column) pelagic-spawning Great Plains cyprinids according to stream fragment lengths. Species with overlapping distributions of extant and extirpated populations indicated significant differences in distributions ($\chi^2 > 7.2$, $P < 0.01$ for all) and logistic regression Nagelkerke R^2 values are reported. Shaded areas reflect thresholds for extirpations associated with fragmentation.
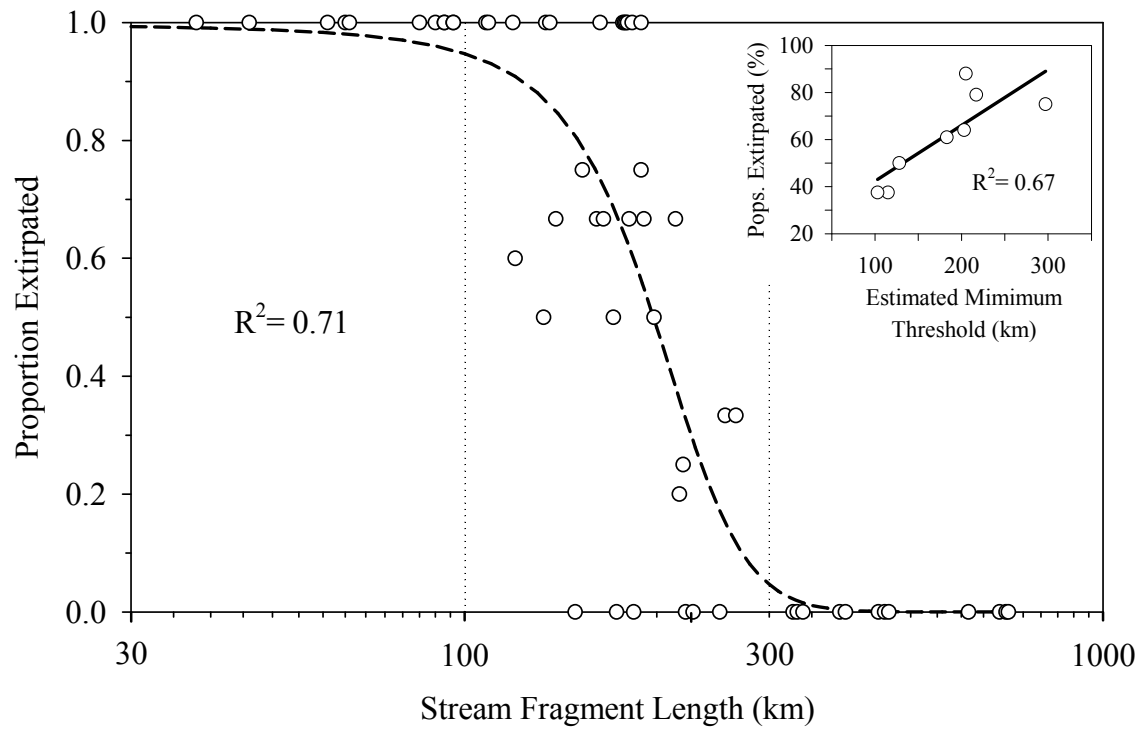
Figure 5. Proportion of species extirpated from Great Plains pelagic-spawning cyprinid communities as a function of stream fragment length (x-axis log-scaled). Logistic regression Nagelkerke R^2 value is reported. Insert illustrates percentage of extirpated populations for each species as a function of the estimated minimum threshold (km) necessary for persistence (see Table 3).
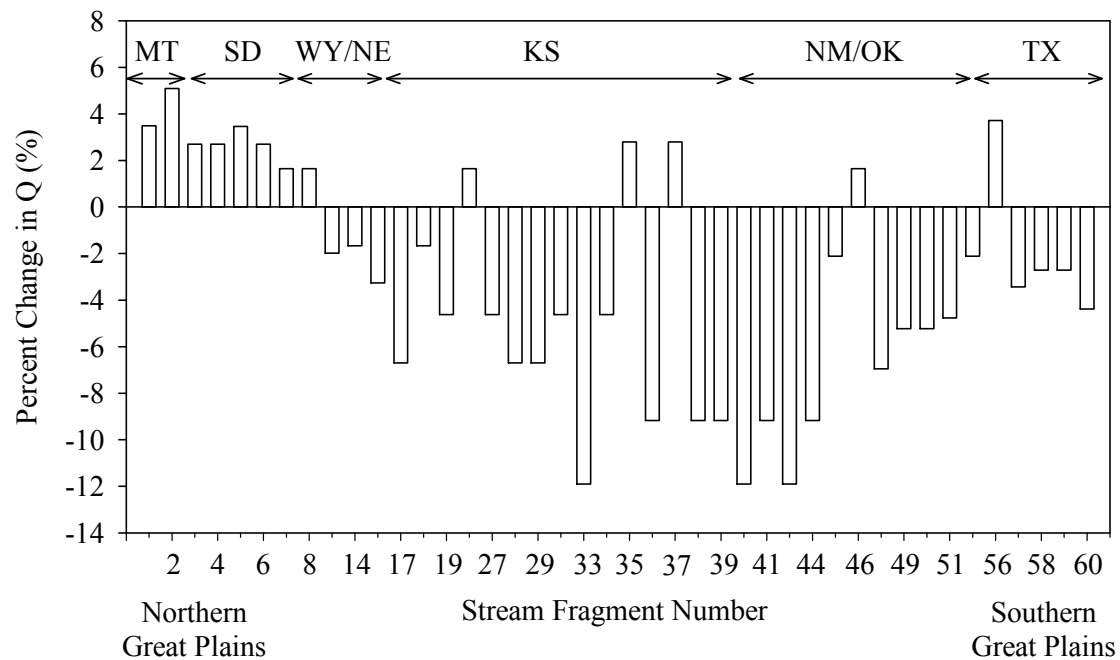
Figure 6. Percent change in mean annual discharge (Q; m^{3}sec^{-1}) invoked by climate change among 42 Great Plains stream fragments for which historical streamflow data were available (data based on Milly et al. 2005). Fragments are arranged by latitude along the x-axis and correspond with descriptions in Table 4.
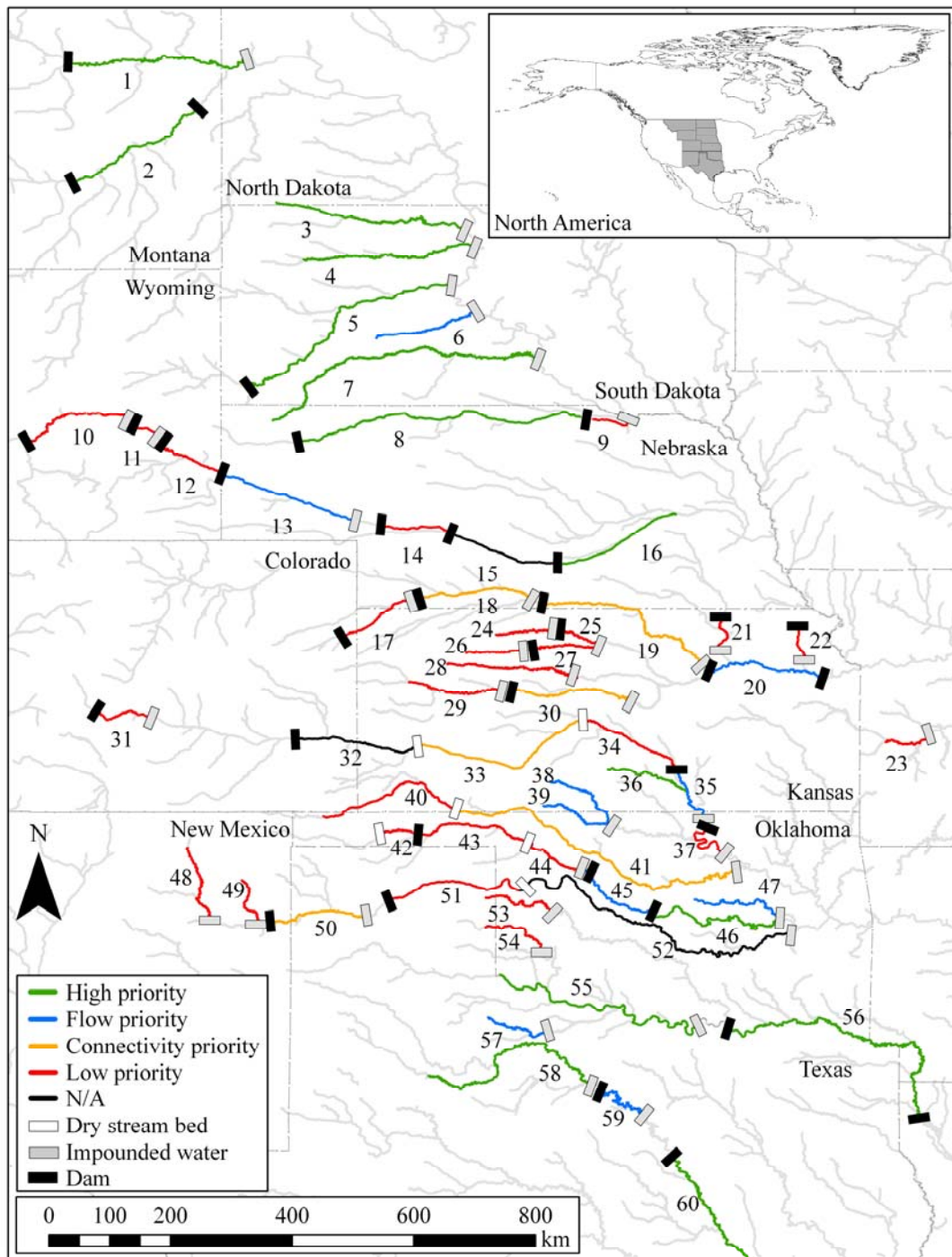
Figure 7. Distribution of Great Plains stream fragments listed as priorities for preservation of longitudinal fragment length and discharge magnitude (high priority, green), preservation of discharge magnitude (flow priority, blue), preservation of longitudinal fragment length (connectivity priority, orange) and fragments for which conservation of pelagic-spawning cyprinids might be limited or unlikely (low priority, red). Data availability for fragments 15, 32 and 52 precluded assignment of priorities; see text for description of priority assignments.