American Fisheries Society Symposium 91:57–83, 2019

Who's Asking? Interjurisdictional Conservation Assessment and Planning for Great Plains Fishes

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Abstract—Aquatic biodiversity is threatened by anthropogenic activities operating across jurisdictional and conservation area boundaries. Strategic conservation planning for broad, multispecies and multijurisdictional landscapes benefits from data-driven approaches emphasizing persistence of priority species while accounting for human uses and stakeholder priorities. This study presents such an assessment for conservation of priority fishes of the U.S. Great Plains. Distribution models for 28 priority fishes were incorporated into a prioritization framework using the open-source software Zonation. A series of assessments were produced, including (1) identification of distinct conservation areas based on connectivity and compositional similarity of priority streams, (2) perspectives for fish habitat condition prioritized towards undisturbed habitat (indicating protection potential) and disturbed habitat (indicating restoration potential), (3) ranking species conservation values at local (state) and global scales, and (4) development of "bang-for-buck" perspectives emphasizing richness of species at state, basin, and study region scales. Assessment highlights include prioritizations primarily among unfragmented main-stem reaches,

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considerable state-boundary-based edge effects for rankings when using state-based conservation values, and identification of eight distinct regions containing natural communities of priority taxa. Further, we integrate an assessment product into a tiered framework for conservation implementation that facilitates coordination among stakeholders across jurisdictions and increases efficiency of conservation efforts. This set of analyses thus provides varying perspectives to

Introduction

Aquatic biodiversity is threatened globally by human appropriation of freshwater resources at scales and rates that ecosystems cannot sustain (Dudgeon et al. 2006). Consequently, freshwater resource conservation is required to maintain both human and freshwater organism populations (Vorosmarty et al. 2010). These conservation actions must occur at scales relevant for species persistence and survival, an operationally challenging caveat given the current and projected limited availability of resources allocated to conservation activities (Strayer and Dudgeon 2010). In fact, resources are generally too limited to protect all freshwater ecosystem services in the face of increasing pressures from population growth and changing climates (Dodds et al. 2013), and there is growing recognition that no project-, species-, or geographic-based conservation focus fully accounts for the innate complexities associated with tradeoffs between ecosystem health and social and economic development (Martinuzzi et al. 2014).

Shifting demands on global landscapes related to both human population growth and resource allocation are driving new approaches to aquatic conservation prioritization frameworks. An earlier “fortress” approach to species protection (e.g., nature preserves, marine reserves) has transitioned into broader, more integrated planning approaches that seek efficiencies in managing both species protection and human uses (Williams et al. 2011; Palomo et al. 2014). This shift is driven primarily by three major themes. First, the need for monetary efficiencies is magnified because of the stark contrast between the amount of conservation work needed and the limited funding allotted to solving environmental problems (Groves et al. 2002; Balmford and Whitten 2003). Second, research indicates that proactive and systematic approaches to conservation are more effective than reactionary or ad hoc approaches, and a general forward-looking approach to conservation is necessary (Margules and Pressey 2000). Finally, conservation frameworks must be built with an understanding of the biological processes that underpin biodiversity at broad scales (Pressey et al. 2007). Thus, conservation constituents across the globe seek to develop economically strategic conservation plans anchored by proactive management and cast in the light of human ecosystem needs and interactions. Internationally, more than 190 countries are committed to implementing conservation planning to reduce biological loss (Balmford et al. 2005), and the Intergovernmental Platform on Biodiversity and Ecosystem Services was recently established to facilitate conservation of biodiversity and sustainable development (Díaz et al. 2015). In the United States, every state and territory has a State Wildlife Action Plan (SWAP) to guide protection of its unique and at-risk natural resources. These plans are intended to provide a funding roadmap for research, management, restoration, and recovery projects for priority species and important habitats. However, as with most globally available conservation plans, adaptive management and implementation strategies remain largely elusive within SWAPs and the critical initial step related to mapping areas of importance is largely ignored (Fontaine 2011). Furthermore, biogeographic boundaries rarely align with geopolitical boundaries and conservation initiatives developed within states might not identify areas of importance that exist across state boundaries (e.g., Pracheil et al. 2012).

Mapping areas of importance is operationally challenging given the sliding scale of importance across diverse sets of stakeholders. Consequently, the initial mapping stage (conservation assessment) acts as a bottleneck for the remaining conceptual stages (planning, management, and review of progress) necessary for proactive conservation of broad, multispecies landscapes (Knight et al. 2006). In an effort to overcome such bottlenecks, systematic conservation assessments consisting of technical, computer-based evaluation of priority elements, gap analysis, and area design are typically generated to inform and anchor conservation planning and implementation frameworks. Research suggests that successful assessments address three primary objectives: species representation, species resiliency, and the ability to implement sustainable management while accounting for human development (Margules and Pressey 2000; Knight et al. 2006; Williams et al. 2011). The vast majority of conservation assessment research has focused on the first objective of developing plans and algorithms to assess, characterize, and prioritize patterns of biodiversity (Pressey et al. 2007). Assessments that attempt to account for species-specific life history needs and that present the results in a perspective that managers can implement are rare (Knight et al. 2008). Further, efficient broadscale conservation planning and delivery for imperiled species are often confounded by differential jurisdictional management practices and priorities (Soberon and Sarukhan 2009). In a sense, the delineation of areas of importance depend entirely on the viewpoint of a stakeholder(s) or who is asking the question "which areas are important?" (Halpern and Warner 2003). Assessments must address these differences and provide managers perspectives that allow efficiencies in allocating conservation funding and actions within and across jurisdictional units (Hunter and Hutchinson 1994; Vane-Wright 1996; Moilanen and Arponen 2011).

Multiple perspectives can guide delineation of areas of importance. Priority areas might be identified through analysis of spatial patterns in habitat connectivity (e.g., Moilanen et al. 2008), species richness or taxonomic groups (e.g., Stewart et al. 2018), in intact habitats (e.g., Thornbrugh et al. 2018), or along political boundaries (e.g., Moilanen et al. 2013). For freshwater ecosystems in particular, native fish conservation areas (NFCAs; Williams et al. 2011) represent an alternative and innovative conservation perspective that complements traditional, reactive approaches to management of aquatic resources. The NFCAs approach is defined as designating areas that adequately support maintenance of processes that create habitat complexity, protection of all life stages, long-term persistence of priority species, and a framework for sustainable management over time (Williams et al. 2011). For example, Dauwalter et al. (2011) completed an NFCA conservation assessment that identified networks of watersheds large enough to manage for both aquatic species population persistence and compatible human uses in selected watersheds of western North America. Such approaches might be applied to other regions where protection of native fishes and human development are colliding (Limburg et al. 2011). In such regions, preservation of ecosystem goods and services that support both native biodiversity and human populations will become increasingly necessary as water security is altered by human extraction and climate change (Vorosmarty et al. 2010). However, methodologies for the establishment of NFCAs require further development, and to our knowledge, there are no quantitative comparisons for priority areas selected using NFCAs versus conservation perspectives involving surrogate taxa, habitat templates, or political units. These competing prioritizations might now be quantitatively compared with the NFCA perspective using sophisticated statistical frameworks (e.g., Zonation; Lehtomaki and Moilanen 2013) to assess consistencies and inconsistences in spatial priorities (Grantham et al. 2010). Of particular importance is the application of multiple conservation planning perspectives in regions where large numbers of imperiled species inhabit changing landscapes that span multiple jurisdictional boundaries (Kark et al. 2015).

The Great Plains of North America exemplifies a region where NFCAs are needed to ensure the long-term persistence of fishes and water. Agricultural water withdrawals from the High Plains aquifer over the past 60 years have outpaced recharge resulting in depletion of massive portions of the aquifer (Steward et al. 2013; Perkin et al. 2017). These changes in water availability are confounded by mismanagement of surface flows through retention and diversion, at times leading to 90% reductions in flow magnitude downstream of impoundments (Costigan and Daniels 2012). As a result, the majority of fishes endemic to the Great Plains are in need of conservation (Hoagstrom et al. 2011) and are recognized in various categories of imperilment at state, regional, and national levels (Haslouer et al. 2005; Jelks et al. 2008). Causes for fish species declines include habitat fragmentation, dewatering, flow regime alteration, water pollution, and introduction of nonnative species (Gido et al. 2010; Hoagstrom et al. 2011). These factors act in concert to transform fish communities throughout the region and increase the urgency to implement coordinated, multijurisdictional conservation interventions to ensure long-term persistence of native fishes (Perkin et al. 2015b).

Here, we present a workflow, sequence of assessments, and planning tools aimed at providing natural resource managers with a spatial perspective for allocation of conservation actions to help assure persistence of native freshwater fishes of the Great Plains. We utilized species distribution models of 28 priority fishes and a spatial prioritization framework to compare spatial conservation value assignments derived from multiple conservation perspectives. The set of resulting assessments are designed to capture ecological characteristics of priority fishes while accounting for individual species conservation status rankings, connectivity requirements, effects of fragmentation and metapopulation size, habitat condition, and differences between regional and broader jurisdictional spatial management priorities. Additionally, we identified distinct high-priority areas based on distance and compositional similarity of communities, forming a starting point for the delineation of NFCAs for the Great Plains. Finally, we validated our NFCA designations by comparing them to two recent conservation area identification studies (Hoagstrom et al. 2011; Perkin et al. 2015a) and integrated part of the NFCA product into an example framework for implementation of partnership- and stakeholder-led conservation of aquatic resources through regionally based fish habitat partnerships. The three-tier spatial framework (sensu Abell et al. 2007) we used identified broadscale watershed-based management regions, mid-level-scale priority areas consisting of NFCAs, and fine-scale identification of critical management areas for a suite of priority species in the southern Great Plains. This example-tiered framework provides stakeholders with a spatial roadmap and communication tool for coupling science needs and management recommendations to priority areas at relevant scales (Fausch et al. 2002).

Methods

Study area

The Great Plains of central North America extend from the Rocky Mountains eastward to the 95th meridian and from Alberta, Canada south to the Rio Grande. Major river basins include Missouri–White, Platte, Republican–Kansas, Arkansas–Cimarron, Canadian, Red, and Brazos–Colorado (Figure 1) that drain from west to east across Montana, Wyoming, North Dakota, South Dakota, Nebraska, Colorado, Kansas, Oklahoma, New Mexico, and Texas. The interstate nature of basin architectures in the region results in multiple jurisdictional boundaries along water courses. The Great Plains Landscape Conservation Cooperative (GPLCC; www.greatplainslcc.org) was created to coordinate transjurisdictional resource management. The extent chosen for this study was the intersection of the GPLCC's extent with U.S. Geological Survey (USGS) Hydrologic Unit Code 4 (HUC4) boundaries modified to exclude areas far downstream of the GPLCC (Figure 1).

Species data

Species occurrence records were obtained from museum specimen-vouchered occurrences and published data. Museum speci-

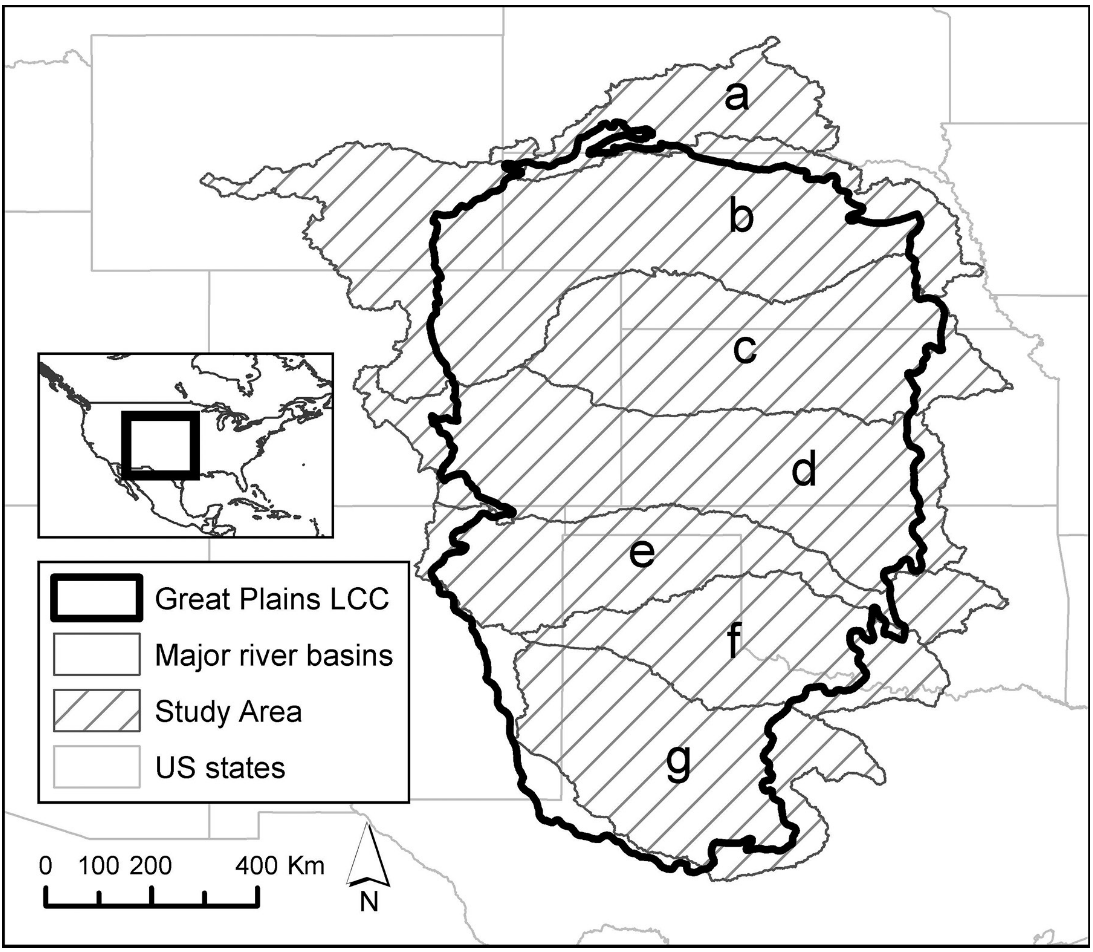

Figure 1. Map of Great Plains Landscape Conservation Cooperative (LCC) extent and study area extent used for analyses. Major river basins indicated and included in the study area include (a) Missouri–White, (b) Platte, (c) Republican–Kansas, (d) Arkansas-Cimarron, (e) Canadian, (f) Red, and (g) Brazos–Colorado.

men-vouchered data sources included two studies (Hendrickson et al. 2010; Cohen et al. 2013) that reviewed and inventoried existing aquatic resource occurrence data from the Global Biodiversity Information Facility (www.gbif.org). Additional museum specimen-voucher records for the entire state of Oklahoma based on collections made by Parham (2009) were obtained from the Sam Noble Oklahoma Museum of Natural History, and data were extracted from the global occurrence database Fishnet2 (http://fishnet2.net). Published data sources included records from the Kansas Aquatic Gap Program, as recently described by Gido et al. (2010), and recent (1993–2013) fish collections made throughout the Great Plains, as described by Perkin et al. (2015b).

These data sets together represent an attempt to use a broad source of primary fish occurrence data, both contemporary and historical. These types of broad, occurrence-based data sets provide a conservative foundation for identifying sets of conservation areas for current and future conservation efforts.

Species distribution models

We used species distribution models (SDMs) to convert point occurrence data into range-wide continuous probability coverages (Guisan et al. 2013). Fish species were chosen on the basis of their state and federal conservation rankings and the authors' expert opinion (Table 1). Environmental variables used in SDM construction (Table 2) were selected in part based on expert evaluation of models created from subsets of variables for a set of species with well-known distributions (see Labay et al. 2011). Selection of functionally relevant environmental predictors is a critical step in species distribution modeling that is ecologically meaningful. The expert opinion-based method to select the set of environmental covariates added an additional model parameterization check in addition to standard statistical validation methods. Climatic, hydrologic, and topographic variables were used to account for broadscale physiological constraints as determinants of distribution (Graham and Hijmans 2006), and two hydrology-based geographic variables controlled for historical zoogeography by categorically constraining predictions of species occurrences to watersheds from which they are documented. Hydrologic variables from the National Hydrology Dataset Plus (NHDPlus Version 1, www.horizon-systems.com/nhdplus/) were converted to 30-arc-second grids using the NHDPlus catchment unit. The spatial extent used for SDM construction was species-specific (Anderson and Raza 2010) and determined by the intersection of input species occurrence data with major watersheds (USGS HUC4). The species-specific extents were divided into a grid of 30-arc-second cells (1 km^2 at equator). The raw environmental grid data were used for modeling, as opposed to a stream-segment based framework, to retain maximum spatial and data resolution.

Species distribution models were constructed using the maximum entropy algorithm encoded in the Maxent software package (version 3.3.4; Phillips et al. 2006). This framework is known to be robust for species distribution modeling with presence-only records (Elith et al. 2006, 2011) and was recently shown to be nearly mathematically equivalent to a Poisson regression model (Renner and Warton 2013). We implemented Maxent following default parameterization recommendations (Phillips and Dudík 2008; Elith et al. 2011) using auto features and 10,000 background points sampled randomly with models cross-validated with 10 replicates to estimate errors around fitted functions. Individual species model performance was evaluated using a receiver operating characteristic (ROC) analysis. The ROC analysis characterizes model performance at all possible thresholds using the area under the curve (AUC). An optimal model with perfect discrimination would have an AUC

Table 1. For each species included in the assessment, number of occurrence records used in model construction and species-specific model and assessment statistics. AUC = area under curve, BQP = boundary quality penalty, and NFCA = Native fish conservation area, determined by similarity of top-ranked river segments based on distance and compositional similarity.

Common name	Genus species	Test AUC	Test AUC SD	No. records in model	Guild group[a]	BQP distance (unit = cells = ~1 km²)	BQP curve[b]	NFCA[c]
Shortnose Gar	*Lepisosteus platostomus*	0.9533	0.0180	336	D	1	1	2–6
Southern Redbelly Dace	*Chrosomus erythrogaster*	0.9309	0.0118	813	A	10	2	3,4,5
Western Silvery Minnow	*Hybognathus argyritis*	0.9637	0.0183	113	B	100	4	2
Brassy Minnow	*H. hankinsoni*	0.9165	0.0231	290	D	1	1	2,3
Plains Minnow	*H. placitus*	0.9529	0.0097	971	B	100	4	3,4,5
Cardinal Shiner	*Luxilus cardinalis*	0.9857	0.0072	266	A	10	2	3,5
Common Shiner	*L. cornutus*	0.9196	0.0127	1,048	A	10	2	2,3
Prairie Chub	*Macrhybopsis australis*	0.9941	0.0038	65	B	100	4	4,6
Sturgeon Chub	*M. gelida*	0.9822	0.0103	56	B	100	4	2,3
Shoal Chub	*M. hyostoma*	0.9626	0.0141	265	B	100	4	2,3
Peppered Chub	*M. tetranema*	0.9753	0.0896	19	B	100	4	3,6
Hornyhead Chub	*Nocomis biguttatus*	0.9593	0.0072	1,044	A	10	2	2,3,5
Emerald Shiner	*N. atherinoides*	0.9503	0.0052	2,910	C	50	3	2,3
River Shiner	*N. blennius*	0.9655	0.0105	591	B	100	4	2,3
Smalleye Shiner	*N. buccula*	0.9765	0.0162	25	B	100	4	7
Arkansas River Shiner	*N. girardi*	0.9712	0.0182	117	B	100	4	3,4
Sharpnose Shiner	*N. oxyrhynchus*	0.9735	0.0178	44	B	100	4	6,7
Chub Shiner	*N. potteri*	0.9766	0.0117	146	B	100	4	6,7
Topeka Shiner	*N. topeka*	0.9844	0.0084	255	A	10	2	3
Suckermouth Minnow	*Phenacobius mirabilis*	0.9598	0.0056	2,067	A	10	2	2,3,6
Slim Minnow	*Pimephales tenellus*	0.9882	0.0040	313	E	1	1	3,4,5
Flathead Chub	*Platygobio gracilis*	0.9682	0.0138	204	C	50	3	2
Blue Sucker	*Cycleptus elongatus*	0.9678	0.0169	209	C	50	3	3
River Redhorse	*Moxostoma carinatum*	0.9639	0.0151	177	C	50	3	2
Neosho Madtom	*Noturus placidus*	0.9934	0.0055	84	E	1	1	3
Northern Plains Killifish	*Fundulus kansae*	0.9607	0.0099	697	A	10	2	3,4
Plains Topminnow	*F. sciadicus*	0.9333	0.0273	106	A	10	2	2,3
Arkansas Darter	*Etheostoma cragini*	0.9891	0.0049	432	A	10	2	3

[a] Guild groupings based on reproductive behaviors as follows: A = lithophilic gravel spawner with adhesive eggs; B = pelagophilic or open substratum or pelagic-broadcast spawning with semibuoyant nonadhesive eggs; C = lithopelagophilic with demersal nonadhesive eggs; D = phytophilic spawner, adhesive eggs onto plants; and E = speleophilic nester, crevices with adhesive eggs.
[b] See Figure 3.
[c] See Figure 4b.

Table 2. Environmental variables used in species distribution models.

Layer category	Description	Source
Topological	Aspect	30-arc second DEM
Topological	Slope	30-arc second DEM
Topological	Compound topological index ln[acc.flow/tan(slope)]	30-arc second DEM
Topological	Altitude	30-arc second DEM
Climate	Annual mean temperature	Wordclim variable 1
Climate	Mean diurnal range [mean of monthly (max temp – min temp)]	Wordclim variable 2
Climate	Isothermality (variable 2/variable 7)(*100)	Wordclim variable 3
Climate	Temperature seasonality (SD*100)	Wordclim variable 4
Climate	Max temperature of warmest month	Wordclim variable 5
Climate	Min temperature of coldest month	Wordclim variable 6
Climate	Temperature annual range (P5-P6)	Wordclim variable 7
Climate	Annual precipitation	Wordclim variable 12
Climate	Precipitation of wettest month	Wordclim variable 13
Climate	Precipitation of driest month	Wordclim variable 14
Climate	Precipitation seasonality (coefficient of variation)	Wordclim variable 15
Climate	Precipitation of wettest quarter	Wordclim variable 16
Climate	Precipitation of driest quarter	Wordclim variable 17
Climate	Precipitation of warmest quarter	Wordclim variable 18
Climate	Precipitation of coldest quarter	Wordclim variable 19
Geographic	Major river basins	U.S. Geologic Survey
Geographic	8-digit hydrologic unit code	U.S. Geologic Survey
Hydrologic	Cumulative drainage	National Hydrology Dataset Plus
Hydrologic	Mean annual flow	National Hydrology Dataset Plus
Hydrologic	Mean annual velocity	National Hydrology Dataset Plus

of 1.0 while a model that predicted species occurrences at random would have an AUC of 0.5 (Hanley and McNeil 1982). We recognize that validation using the AUC statistic has been criticized as misleading as it tends to overestimate model quality, may show spurious high performance with small sample sizes, and may reward overparameterization (Lobo et al. 2008). However, we use AUC (albeit conservatively via a high threshold of acceptability) for its commonality and lack of viable alternative measures and because our intended use of these models was in the context of coarse-scale identification of prime suitable habitat. Species models were considered reliable and included in prioritization analyses if they had an average test AUC over 10 replicates >0.9. Information from rangewide SDMs was restricted to the study area extent for assessment analyses, and all spatial data analyses were conducted at a 30-arc-second (1 km^2 at the equator) grid resolution.

Conservation assessment software

We used the planning software Zonation (Moilanen et al. 2005), which includes many feature options designed to allow sophisticated prioritizations to be performed while accounting for a wide range of parameters. The primary function of the software is to produce a landscape ranking based on conservation value defined by spatially explicit levels of species, habitat, or ecosystem occurrence. It does this in our case by initially considering the entire landscape being analyzed and iteratively removing spatial grid cells that result in the smallest loss of conservation value as defined by SDM estimation of relative probability of occurrence. Zonation allows alternative cell removal rules that emphasize different types of conservation values. The two removal rules used in this study include one that emphasizes species rarity (core-area zonation [CAZ]; Moilanen et al. 2005) and another that emphasizes species richness (additive-benefit zonation [ABZ]; Moilanen 2007). Most analyses performed for this study use the CAZ approach, which aims to identify core priority landscapes for each species regardless of overall species richness. In other words, the CAZ approach can potentially consider two streams equally important, even if one contains substantially more priority species. The intent with the CAZ approach is to identify the set of core areas most relevant for conservation of all priority species. The ABZ approach is used in our final set of assessment analyses that consider differing spatial results from prioritization on the basis of species richness at multiple scales.

Assessment analyses

We produced a series of five analyses to provide differing conservation area perspectives utilizing various Zonation features to “direct” the cell removal process (Figure 2). These five analyses focused on prioritizing areas with (1) intact connectivity where imperiled species persist (primary prioritization), (2) shared occurrences of imperiled species where resources might be focused to promote conservation of native fish assemblages and the habitats they require (NFCA prioritization), (3) least-disturbed habitats (fish habitat quality prioritization), (4) shared political boundaries where jurisdictional lines are frequently drawn (global versus administrative boundary unit prioritization), and (5) high species richness where conservation efforts might be applied to maximize taxonomic benefits (species richness prioritization).

Primary prioritization (CAZ + S + C).— Our primary prioritization utilized the core area removal process (CAZ) and incorporated species-specific weighting (S) and connectivity constraints (C). Species global or subnational (i.e., U.S. state) conservation status ranks set by NatureServe (Faber-Langendoen et al. 2009; Table 3) were used as species-specific weights in the Zonation prioritization algorithm (Moilanen et al. 2005).

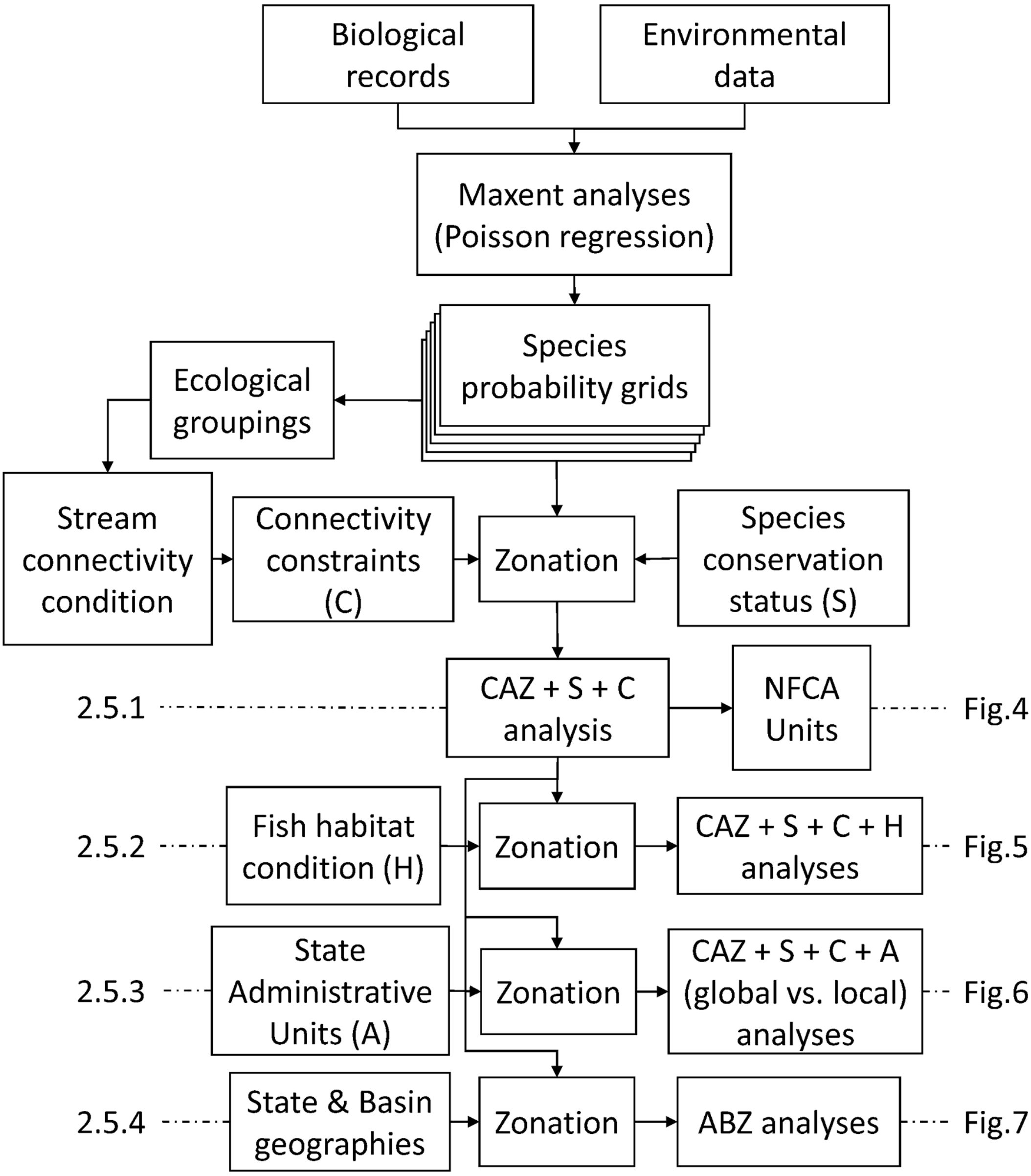

Figure 2. Schematic of the analysis sequence with indication of the sections in methods where each analysis is discussed (left) and the figures that display resultant spatial products (right). CAZ (core-area zonation) and ABZ (additive-benefit zonation) indicate the two removal rules used in this study that emphasizes either species rarity (CAZ, Moilanen et al. 2005) or species richness (ABZ, Moilanen 2007). NFCA = native fish conservation areas.

Species with elevated NatureServe imperilment rankings (e.g., G1 critically imperiled versus G5 demonstrably secure) thus had elevated weights in Zonation, resulting in higher priority for protection of their occurrences (Table 4).

We incorporated habitat connectivity into our analyses via two methods. The first,

Table 3. For each species included in the assessment, NatureServe rank for each state in the study area and globally (Faber-Langendoen et al. 2009). See Table 4 for status code definitions.

Species	SD	WY	NE	CO	KS	OK	NM	TX	Global
Shortnose Gar	S4		S5		S4	S4		S3	G5
Southern Redbelly Dace	S1			S1	S2	S5	S1		G5
Western Silvery Minnow	S5	S2	S5		S2				G4
Brassy Minnow	S5	S5	S4	S3	S1				G5
Plains Minnow	S5	S3	S4	SH	S2	S5	S3	S4	G4
Cardinal Shiner					S3	S4			G4
Common Shiner	S5	S3	S2	S2	S4	S4			G5
Prairie Chub						S4	SNR		G3
Sturgeon Chub	S2	S1	S1		S1				G3
Shoal Chub			S4		S4	S5		SNR	G5
Peppered Chub				SX	S1	S4	S1	S1	G1
Hornyhead Chub	S3	S1	SH	SX	S1				G5
Emerald Shiner	S5		S4		S5	S5		S4	G5
River Shiner	S2		S4		S3	S3		S3	G5
Smalleye Shiner								S2	G2
Arkansas River Shiner					S1	S1	S1	S2	G2
Sharpnose Shiner								S3	G3
Chub Shiner						S4		S4	G4
Topeka Shiner	S2		S1		S2				G3
Suckermouth Minnow	SH	S2	S4	S2	S5	S4	S2	S4	G5
Slim Minnow					S4	S3			G4
Flathead Chub	S5	S5	S5	S3	S1	S1	S4	S2	G5
Slim Minnow	S3		S1		S3	S3	S1	S3	G3
River Redhorse					S1	S1			G4
Neosho Madtom					S2	S1			G2
Northern Plains Killifish		S5	S4	S5	S3	SNR	SNR	SNR	G5
Plains Topminnow	S3	S3	S3	S4	S1	S1			G4
Arkansas Darter				S2	S2	S2			G3

boundary length penalty (BLP; Moilanen and Wintle 2007) is a generalized induction of priority area aggregation based on the structure characteristics (perimeter) of the priority network that produces more compact and less-fragmented solutions. The hierarchy of cell removal is influenced not only by species occurrence probability, but also by the increase/decrease of boundary length that results from cell removal. In our case, this method promotes clustering of priority areas into more continuous segments as opposed to highly fragmented priority reaches along a stream. The second method, boundary quality penalty (BQP; Moilanen and Wintle 2007), is species-specific and accounts for many ecological characteristics of both rivers and species. It is a quantitative way to induce aggregation that accounts for species responses to fragmentation, edge effects, metapopulation size, and connectivity. Here, we used BQP to specifically address species-specific response to fragmentation and thus indirectly account for landscape connectivity and habitat loss. The BQP method was implemented with two expert-defined components, a distance radius determined by when a species is affected by increasing fragmentation or smaller fragment size and a response curve

Table 4. Weights assigned according to NatureServe rank (Faber-Langendoen et al. 2009). Global and state weights were assigned to represent relative priority from one level to the next.

Weight	Status code	Status
0	SX	Presumed extirpated
0	SH	Possibly extirpated
6	S1	Critically imperiled
5	S2	Imperiled
4	S3	Vulnerable
3	S4	Apparently secure
2	S5	Secure
1	SNR	Species not recorded (but present)
0	OR	Out of range
5	G1	Critically imperiled
4	G2	Imperiled
3	G3	Vulnerable
2	G4	Apparently secure
1	G5	Secure

determined by how a species responds to fragmentation and habitat loss. The BQP radius defines the distance (in cell units of the species distribution model grid) across which habitat loss or fragmentation affects a species. For example, a pelagic, broadcast-spawning fish with drifting eggs (e.g., Sharpnose Shiner N) needs long stretches of river (Wilde and Urbanczyk 2013) and thus would have a large radius of fragmentation effect relative to a speleophilic nester that uses crevices and has adhesive eggs (e.g., Slim Minnow; Perkin et al. 2015b). Expert opinion among coauthors (B.J.L., J.S.P.) anchored in fish reproductive guild ecology (Simon 1999; Frimpong and Angermeier 2009) was used to determine radii for each species (Table 1). In absence of life history characteristics that would inform radius size selection for a species, a 10-km radius was used (Hitt and Angermeier 2008, 2011). The second component, BQP response curve, defines the relative magnitude of the effect of fragmentation on each species (Figure 3). For example, for obligate riverine species, we assumed a stronger response to fragmentation and used an extreme penalty curve (see line 4 in Figure 3). We again used coauthor judgment (B.J.L., J.S.P.) anchored in fish reproductive guild ecology to assign response curves (Table 1; Figure 3).

We formally incorporated connectivity constraints into Zonation analyses. Hierarchical prioritization in Zonation can take place on species occurrence while accounting for landscape condition (here, connectivity) and how species differentially respond to that condition. We used the total main-stem distance between upstream and/or downstream dams as a condition layer. This measure was generated by overlaying point locations of large dams on the NHDPlus Version 1 stream network and calculating total main-stem availability (river kilometers) based on distances to main-stem dams (if present) above and below each stream reach (Cooper et al. 2017). Upstream main-stem pathways represent the longest upstream route, whereas downstream main-stem pathways were defined as the shortest route to a stream network outlet (e.g., ocean). Species in guild grouping B (Table 1) were assigned to this connectivity condition factor during select analyses.

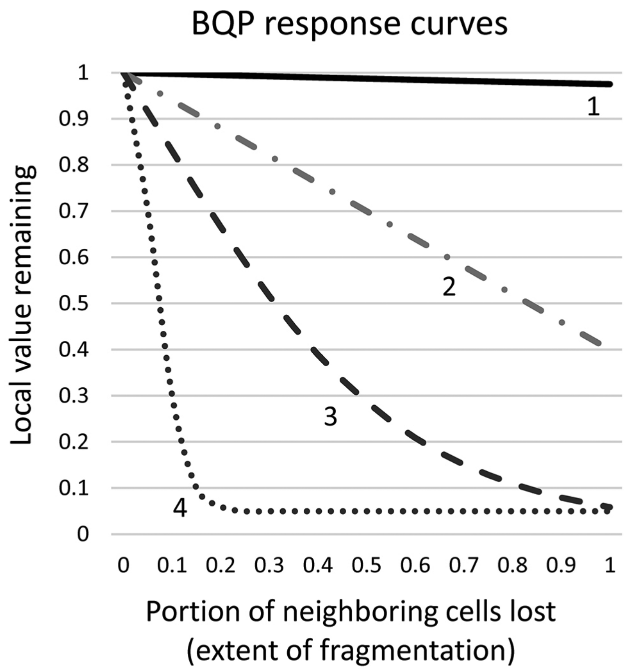

Figure 3. Four species-specific fragmentation penalty curves representing response to fragmentation. BDQ = boundary quality penalty. See Table 1 for species-curve designations.

Native fish conservation areas prioritization.—Zonation allows for the identification of distinct species-based geographic units (here called NFCAs) based on the distance and compositional similarity between priority areas. This is done using four user-specified parameters: (1) percentage of the landscape to consider for inclusion in the management units, (2) minimum inclusion top fraction for each unit (what top fraction must be present in each separate unit), (3) a maximum distance between units, and (4) a maximum difference in compositional similarity between units. To identify NFCAs for Great Plains streams, we used the "CAZ + S + C" primary prioritization as the starting point and chose to consider (1) the top 10% of the landscape ranking, (2) a minimum inclusion of the top 2% (meaning each unit had to contain a portion of the top 2% of the ranking), (3) 25 grid cells (approximately 20 km) for the maximum distance allowed between spatially discrete patches that are included in the same management unit, and (4) a maximum difference in species composition so that two units were determined different if 2 species out of 10 (20%) had a 1-log difference in their probability density (SDM value) between the two areas.

Fish habitat quality prioritization (CAZ + S + C + H).—We also used the index of cumulative disturbance of river fish habitats of the National Fish Habitat Assessment (H), described in Esselman et al. (2011), to rank cells across the landscape as having fish habitat that ranged from undisturbed to highly degraded in relation to occurrence of priority taxa. Incorporating these two condition extremes into separate prioritizations allows for divergent perspectives of stakeholders—those that value and focus on proactive protection of intact habitat versus those that value and focus on restoration of disturbed habitat. The prioritization analyses labeled "CAZ + S + C + H" (see Figure 2) represent two analyses, one in which the fish habitat disturbance condition was used to prioritize selection of low levels of disturbance and another to prioritize areas of high levels of disturbance in need of restoration.

Global versus administrative unit prioritization (CAZ + S + C + A).—We utilized a relative weighting feature in Zonation, also used by Moilanen and Arponen (2011), to provide both an administrative unit prioritization perspective (A) for each state and a broader full study region prioritization. The first is produced by weighting species differentially across states according to each state's NatureServe "S" status (Faber-Langendoen et al. 2009) and the second by weighting species according to their global "G" status.

Species richness prioritization.—We used the additive-benefit cell removal method (Moilanen 2007) to identify richness hot spots within basin, state, and study area geographies. The additive-benefit function calculates a solution where value is additive across biodiversity features (species in this case), and where feature-specific representation is converted to value via concave power functions (Moilanen 2007). Here, the exponent of the power function was set to $z = 0.25$ for all species. The ABZ can be seen as a "bang-for-buck" type of analysis that puts value in areas with more species and downgrades areas that have fewer species. The ABZ approach is not typically used when the analysis features are representing themselves (species); however, we perform these analyses to illustrate the difference in assessment results, and to provide a "bang-for-buck" perspective, particularly for stakeholders and practitioners working at the state level.

Results

Primary prioritization

The primary prioritization (Figure 4a) showed that the majority of the highly ranked, tier 1 (top 2% of landscape) areas were located on unfragmented sections of main-stem streams and, in the southern part of the study area, were bounded downstream or upstream by major impoundments. Examples include the Brazos River upstream of Possum Kingdom Lake, the Red River upstream of Lake Texoma, and the Canadian River downstream of Lake Meredith. This can be attributed to the high numbers of pelagic-broadcast spawning taxa (e.g., Sharpnose Shiner and Arkansas River Shiner) in the analysis and the explicit parameterization concerning their life history requirements related to drifting eggs. However, highly ranked areas also included smaller tributaries, where the core areas of other rare species occurred. These include Brassy Minnow and Plains Topminnow, with core habitat in the tributaries of the Loup River in central Nebraska, and Arkansas Darter and Northern Plains Killifish, with core habitat in western tributaries of the Arkansas River in south-central Kansas. Inclusion of factors to force connectivity and spatial aggregation of resulting rankings in the primary prioritization resulted in a large number of unique areas in the top 10% (1,390). From a mobile organism's perspective, many of these separate areas are connected and should perhaps be managed as combined units.

Native fish conservation area prioritization

The NFCA prioritization grouped priority areas from the primary prioritization according to natural biological communities and proximity. Within the top 10% of the landscape, eight distinct geographic units (i.e., NFCAs) were identified. Each contained a portion of the top 2% of the core areas, and each had sufficient geographic separation and compositional similarity (Figure 4b). Together, these comprised 9.2% of the study area and generally corresponded with major river basins, including (north to south) the White (South Dakota), Platte (Nebraska),

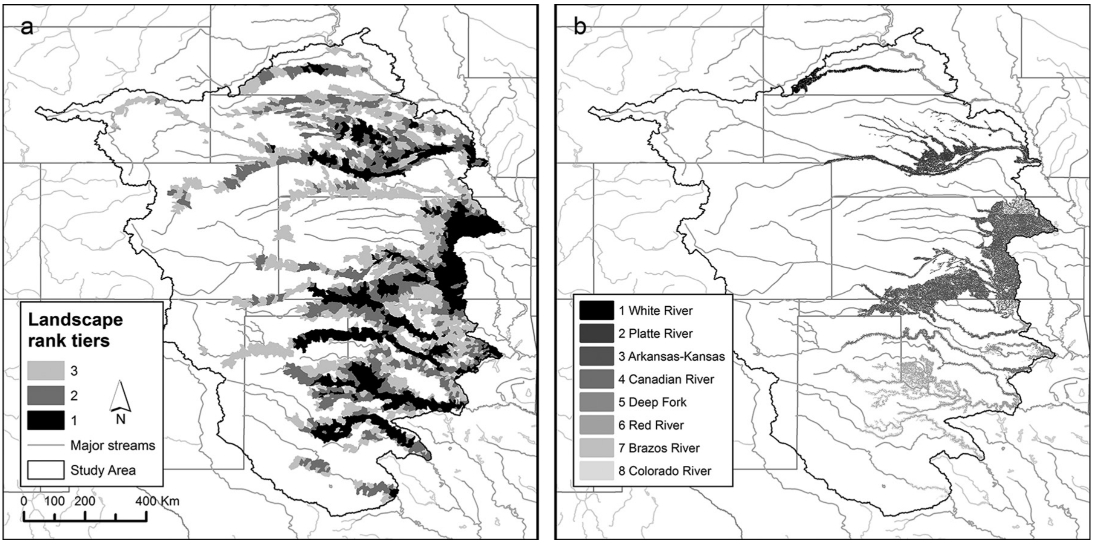

Figure 4. **(a)** Primary prioritization with core-area function zonation analysis, species-specific rankings, connectivity constraints, ecological groupings, and stream connectivity condition factor (CAZ + S + C) for the 28 priority fishes of the Great Plains. Landscape rank tiers represent top fractions of the landscape ranking results and are symbolized by maximum ranking per 12-digit USGS hydrologic unit code: tier 1 = 2% (1–0.98 cell values), tier 2 = 5% (0.98–0.95), and tier 3 = 10% (0.95–0.9). **(b)** Broadscale native fish conservation areas (NFCAs) identified according to distance and compositional similarity among highly ranked regions of the CAZ + S + C analysis.

Arkansas and Kansas (Kansas), Canadian (Texas and Oklahoma), Deep Fork (Oklahoma), Red (Texas and Oklahoma), Brazos (Texas), and Colorado (Texas) River basins.

Fish habitat quality prioritization

Prioritizing "undisturbed" (Figure 5a) and "disturbed" (Figure 5b) habitats each produced top-tier rankings that cover approximately 2% of the study area. These two results overlapped broadly, especially in the Brazos and Red river basins as a result of high-priority core habitats having relatively moderate values of disturbance. Nevertheless, there were notable differences between the two prioritization results. Some of the largest differences occurred in tributaries in the Platte and Arkansas River basins. Within the Platte, relatively undisturbed habitats were identified as core areas in the tributaries of the Loup River (Figure 5a) in Nebraska, whereas disturbed fish habitats were identified along the main-stem Platte River (Figure 5b). In the Arkansas River basin, headwater tributaries of the Salt Fork Arkansas River contained large amounts of core area of relatively undisturbed habitat, whereas the Chikaskia and portions of the Ninnesah contained core areas of disturbed habitat. These results provide guidance for different types of management of habitat for priority fishes (e.g., protection versus rehabilitation).

Global versus administrative unit prioritization

Differences between prioritizations with species weighted according to their global and subnational conservation rankings showed considerable differences. Global weights resulted in less fragmentation of core areas (n = 653) compared to variable

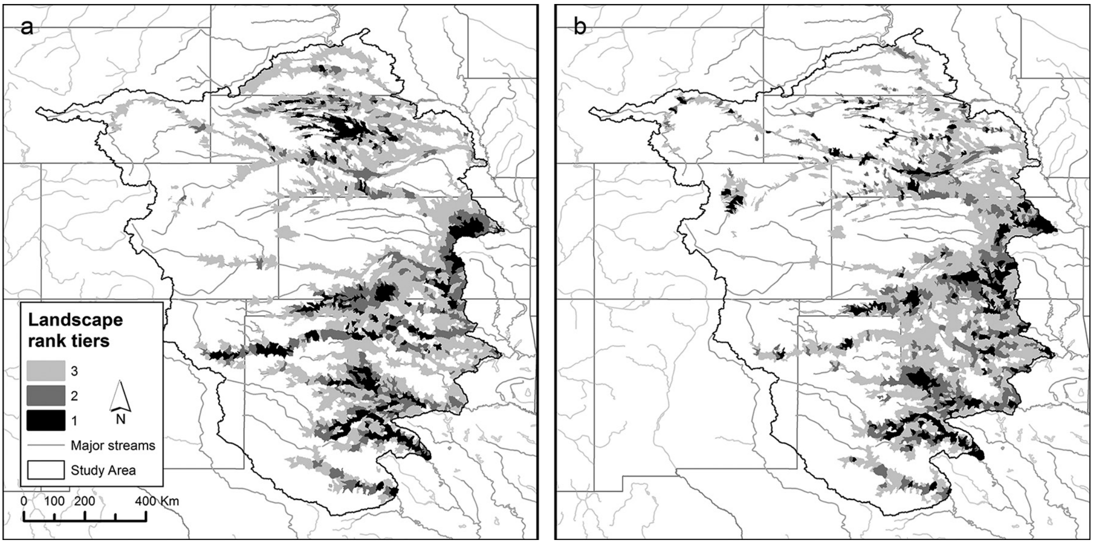

Figure 5. Riverscape prioritization derived from CAZ + S + C setup by adding a fish habitat disturbance condition factor (H) to prioritize areas where the 28 priority fishes of the Great Plains coincide with **(a)** relatively undisturbed habitat and **(b)** relatively disturbed habitat according to the cumulative fish habitat disturbance metric of the National Fish Habitat Action Plan (Esselman et al. 2011). Landscape rank tiers represent top fractions of the landscape ranking results and are symbolized by maximum ranking per 12-digit USGS hydrologic unit code: tier 1 = 2% (1–0.98 cell values), tier 2 = 5% (0.98–0.95), and tier 3 = 10% (0.95–0.9).

state rankings (n = 2,249). Globally weighted rankings generally identified core areas in eastern (Figure 6a) main-stem streams, whereas state-weighted ranking identified more uniform distribution of core areas (Figure 6b). New Mexico, Colorado, and Wyoming completely lacked top-tier (2% of landscape) core areas in the globally weighted prioritization, yet contained 29% of the top-tier core area when prioritization was based on variable state-level conservation weighting for species.

Species richness prioritization

The multiscale prioritization differed dramatically when using the additive benefit cell removal rule (see 2.5.4; providing a "bang-for-buck" perspective) at different spatial extents. Figure 7 shows riverscape prioritizations using the additive benefit cell removal method rule for major river basins (Figure 7a), U.S. states (Figure 7b), and the study area (Figure 7c). The top-tier (2% of landscape) core areas identified vary greatly according to geographic extent of analysis, but in general, core areas are in eastern streams due to higher richness in downstream reaches throughout the study area.

Discussion

Our study provides a comparison of five conservation perspectives useful for identifying areas of conservation interest across the U.S. Great Plains. The primary perspective summarized the distribution of highly connected habitats that are still habited by species in need of conservation, but it identified a large number of small, local areas that were actually nested within broader areas sharing unique assemblages of fishes. Our second perspective based on the NFCA concept (Williams et al. 2011) aggregated these local areas within a smaller number of geographically larger units. The NFCA units tended

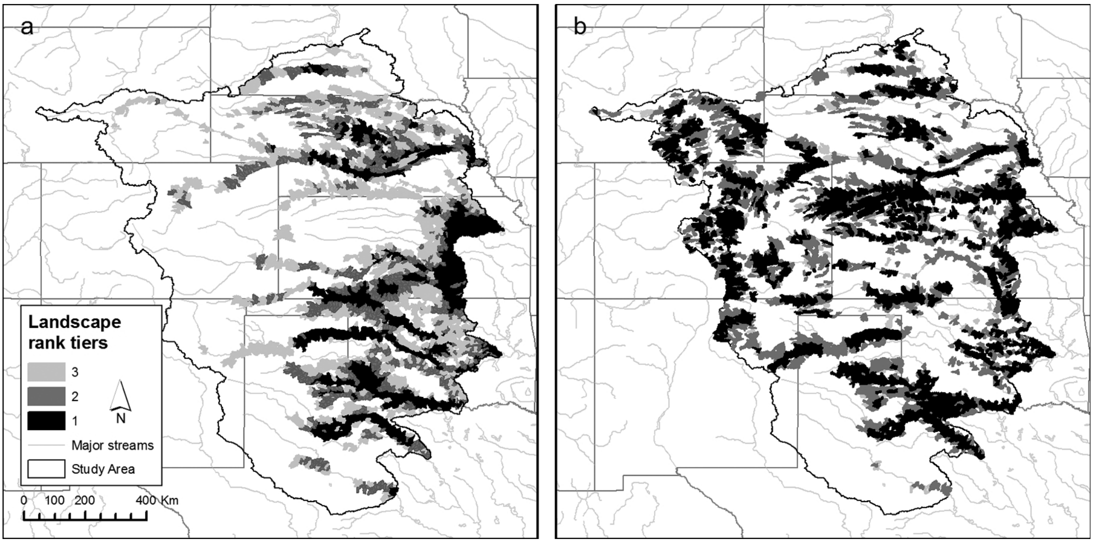

Figure 6. Riverscape prioritization derived from CAZ + S + C setup with **(a)** global species conservation rank weights (Table 3) and **(b)** species weights differentially set in each state based on state-level NatureServe conservation rankings (S rank; Table 3). Landscape rank tiers represent top fractions of the landscape ranking results and are symbolized by maximum ranking per 12-digit USGS hydrologic unit code: tier 1 = 2% (1–0.98 cell values), tier 2 = 5% (0.98–0.95), and tier 3 = 10% (0.95–0.9).

to include the same areas of "undisturbed" habitats identified through the perspective based solely on habitat condition as well as areas identified in the global (Great Plains), but not local (individual states) perspectives. Furthermore, the NFCAs contained the majority of the diversity hotspots identified through the perspective based only on species richness, measured at either the river basin or study area extents (but not state extent). Our results highlight that priorities within geopolitical boundaries (states) were not always spatially consistent with priority areas identified with a broader perspective, meaning broadscale coordination among states with a shared focus on NFCAs is a more efficient conservation methodology compared to state SWAPs operating independently (Neeson et al. 2015). Together, these findings suggest the question "where are areas of interest for fish conservation in the Great Plains?" depends largely on who's asking when viewed from the perspective of individual states, but this subjectivity is reduced considerably when broader geographic perspectives that match the scale of fish biogeography are considered.

Assessment validation with other studies

We assessed the validity of our NFCA framework by comparing it with two previous conservation perspectives for Great Plains fishes (Figure 8). Hoagstrom et al. (2011) suggested that 18 high-quality native fish refuge streams persisted in the Great Plains, each inhabited by at least three Great Plains endemics. Our NFCA model captured 9 of that study's 13 proposed refuge streams that were in our study area (Figure 8). The major area of disagreement between conservation outlooks hinged on inclusion by Hoagstrom et al. (2011) of Plains Sand Shiner *Notropis stramineus missuriensis*, a widely distributed subspecies not included in our NFCA framework because it has a stable conservation status. Endemic fish refuges proposed by

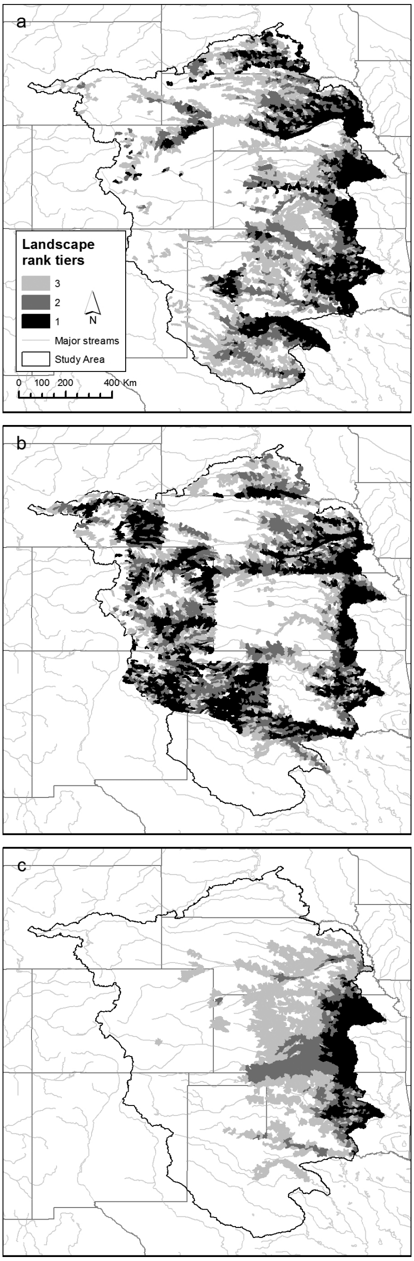

Figure 7. Riverscape prioritization using the additive benefit cell removal prioritization rule for **(a)** major river basin, **(b)** U.S. state, and **(c)** study area geographic extents. Landscape rank tiers represent top fractions of the landscape ranking results and are symbolized by maximum ranking per 12-digit USGS hydrologic unit code: tier 1 = 2% (1–0.98 cell values), tier 2 = 5% (0.98–0.95), and tier 3 = 10% (0.95–0.9).

Hoagstrom et al. (2011) but not identified by our model as having high conservation value tended to be in the western Great Plains where Plains Sand Shiner persists despite declines or extirpations of other native fishes (e.g., Perkin et al. 2015a). Similarly, 13 of 16 stream fragments identified as high conservation priorities by Perkin et al. (2015a) were captured in our NFCA framework, likely because parameterization of our assessment utilized some of the same species-specific life history requirements included by Perkin et al. (2015a). The three not identified in our NFCA framework all occurred along the western edge of the Great Plains where water availability has declined greatly because of depletion of the High Plains aquifer (Steward et al. 2013; Perkin et al. 2017). Although some native species habitat persists in these western extremes of the study area (e.g., the North and South Platte rivers of Wyoming; Steward et al. 2013) and the conservation of these habitats is critical for the maintenance of regional native fish diversity, the ecological future for fish habitat in some portions of the western Great Plains is dire (Falke et al. 2011). Thus, the spatially explicit nature of inclusion and exclusion of previously proposed priority areas for Great Plains fishes is related to the riverscape ranking basis of our modeling framework. That is, excluded stream segments are of high value, albeit at a lower relative ranking compared to the identified NFCA streams. Areas highlighted as having high conservation value in our model are simply the NFCAs in which top-tier priority species' core habitat exists, and we stress that conservation opportunities in the portion of the Great Plains lying outside our proposed NFCAs should not be ignored. Additionally, the NFCA framework identified here is intended to be an initial starting point for local experts to amend as necessary to address stakeholder priorities at finer scales, and consideration of critical exclusions must be made by users of any particular assessment perspective provided here.

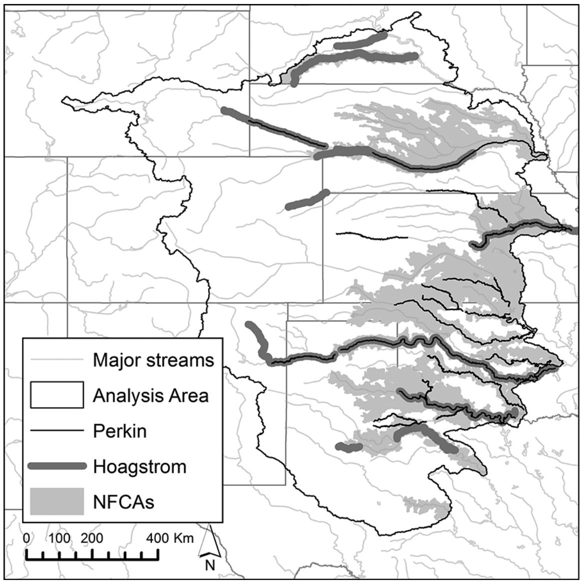

Figure 8. Map showing overlap in between the native fish conservation area (NFCA) framework presented in this study and two previous conservation priority perspectives for Great Plains fishes (Hoagstrom et al. 2011; Perkin et al. 2015a).

Delivery as planning tools for conservation action

To bridge the assessment-implementation gap in conservation planning, conservation assessment products such as those produced here must be complemented with an implementation strategy (Fausch et al. 2002). This can be achieved by mainstreaming of the planning products, coupling their recommendations into communication and collaboration tools used to coordinate among various stakeholders. In practice, this can mean interpreting and redesigning these tools to facilitate a framework for decisions by the diverse personnel and organizations whose work influences natural resource management (e.g., Pierce 2003). As an example, the federally required SWAP of every U.S. state and territory include guidance for interpretive land and stream habitat management (Fontaine 2011). Placing these guidelines into a strategic planning framework, as provided by high-level assessment and planning products such as produced here, facilitates efficiencies for land-use planners and conservation organizations wanting to coordinate and justify decisions and recommendations regarding best management practices.

This mainstreaming of assessment and planning products is not a one-time effort, but requires continuous input and feedback from stakeholders involved. Further, the activity of mainstreaming and purposing these assessment products must be uniquely performed for, and by, individual implementation organizations and researchers, as the information and perspectives presented do not offer the same value across different groups (e.g., federal versus state biologists). That is, the utility of the conservation perspectives provided in our comparative study depends on who is asking for guidance, and our approaches are not intended for fine-scale prioritization requiring local expertise. Resources should be put towards tailoring assessment products into use-case planning tools depending on specific needs of implementation practitioners. At the very minimum, these tools, especially the perspectives of species management units, provide regional managers with organizational frameworks that help them communicate more efficiently with different conservation-implementation and funding groups. However, optimally, individual conservation groups, researchers, and state agencies should be encouraged to come together and actively leverage their unique opportunities to mainstream assessment products, as presented here, to highlight linkages between natural resources and socioeconomic benefits.

Integration into a framework for implementation of the National Fish Habitat Partnership

The National Fish Habitat Action Plan (NFHAP; AFWA 2012; www.fishhabitat.org) provides strategies for cooperative, interjurisdictional, and landscape-scale conservation of fishes and other aquatic resources in the United States and serves as the strategic plan for the National Fish Habitat Partnership and associated network of regionally focused fish habitat partnerships. Within the southern extent of the U.S. Great Plains, the Southeast Aquatic Resources Partnership (SARP; www.southeastaquatics.net) is actively engaged in regional implementation of the NFHAP. The Southeast Aquatic Resources Partnership is a collaborative, multiagency conservation partnership geographically aligned with the 14 member states of the Southeastern Association of Fish and Wildlife Agencies, including the U.S. Great Plains states of Oklahoma and Texas.

Formed in 2001, the mission of SARP is to protect, conserve, and restore aquatic resources, including habitats throughout the region, for the continuing benefit, use, and enjoyment of the American people (SARP 2014). Since the partnership's inception, SARP has served as a regional catalyst and network builder for fish habitat conservation, spearheading regional assessments of flow alteration, riparian condition, and fish passage barriers and supporting on-the-ground delivery of aquatic habitat restoration projects (SARP 2014). The partnership was formally recognized as a fish habitat partnership in 2007, and in 2008, SARP and partners published the Southeast Aquatic Habitat Plan (SARP 2008), which established regional conservation objectives and targets (i.e., 5-, 10-, and 15-year outcomes) used by the partnership to monitor progress and to continually adapt and refine regional fish habitat conservation strategies (SARP 2014).

Critically important to the success of SARP and other fish habitat partnerships is the ability to facilitate communication and cooperative planning among local, state, and federal natural resources management agencies, nongovernmental organizations, and other partners. Cooperative planning allows for identification of shared geographic (e.g., ecoregions, watersheds) and thematic (e.g., dam removal, flow restoration, riparian restoration) priorities and supports strategic investments and leveraging of available technical and financial resources, often allowing for significant expansion of the scope and scale of local conservation projects (e.g., extent of watershed restored, inclusion of project-based monitoring or applied research necessary to evaluate and improve restoration designs). To examine the utility of the conservation assessment products assembled through this study, a pilot conservation planning process was conducted in collaboration with SARP and the GPLCC. The conservation planning process occurred in NFCAs prioritized within the Brazos, Canadian, Colorado, and Red River basins (Figure 4b; Birdsong et al. 2018). Local, state, and federal natural resources management agencies, universities, and nongovernmental organizations were invited to participate in a series of webinars, workshops, and field days. These events were used to profile the conceptual underpinnings of the NFCAs approach (Williams et al. 2011), to review and obtain feedback on the geographic extent of the Great Plains river basins prioritized as NFCAs (Figure 4b), and to identify strategies and project-level conservation actions to restore and preserve focal species and their habitats within the NFCAs. A total of 77 conservation professionals participated in the conservation planning process, collectively identifying 96 project-level conservation actions across the four NFCAs (Birdsong et al. 2018). Actions were prioritized and formulated into multiyear (5–10 years), watershed-based conservation action plans intended to guide and facilitate collaborative conservation within NFCAs of the Brazos, Canadian, Colorado, and Red River basins (Birdsong et al. 2018). Subsequent efforts to implement the conservation action plans assembled for those four NFCAs are described by Birdsong et al. (2019, this volume).

This watershed-based conservation planning within NFCAs of the U.S. Great Plains provided a successful case study that has since been transferred and replicated within other regions of the United States. The Desert Fish Habitat Partnership and Western Native Trout Initiative, two fish habitat partnerships focused on implementation of NFHAP within the southwestern United States, recently cooperated to implement this same conservation assessment and prioritization approach within the Rio Grande basin (Labay et al. 2018), which encompasses the U.S. states of Colorado, New Mexico, and Texas. For the portion of the Rio Grande basin located within the Chihuahuan Desert ecoregion of Texas, six NFCAs were prioritized. The same conservation planning process completed for NFCAs of the U.S. Great Plains was employed by Garrett et al. (2019, this volume) to assemble conservation action plans for those six NFCAs.

Prioritization of NFCAs has served as the impetus for increased and focused investments in native fish conservation (e.g., research, monitoring, habitat restoration, and protection) within select river basins of the U.S. Great Plains and the southwestern United States (Birdsong et al. 2018, 2019; Garrett et al. 2019). These outcomes have demonstrated the utility of assessment products described herein in enhancing communication and fostering collaboration among nongovernmental organizations, universities, and state and federal agencies and in facilitating the leveraging of staff, expertise, project funding, and other resources toward delivery of proactive conservation projects.

Conclusions

Conservation assessment is the technical task of identifying, categorizing, and ranking areas of importance in terms of their relevance to accomplishing conservation goals. This study performed a series of assessments for 28 priority fishes of the U.S. Great Plains that account for various ecological and landscape condition factors that influence fish distributions. The assessment process involved creating a framework for design and implementation of linked components (e.g., species life history traits and stream condition) at a scale appropriate for identification of broad priority areas for streams in the study area.

These assessments constitute a critical component of the broader process of conservation planning. Initial decisions regarding data and assessment design determine how the resultant assessment tools can and should be applied and integrated into the broader planning process, and throughout the setup process, we created an assessment that utilized a NFCA approach to identify important areas for the target fishes while accounting for spatial variability in stakeholder priorities (Williams et al. 2011). Interjurisdictional conservation decisions and solutions require consideration of the priorities held by those who are asking for conservation management direction. Because local protection areas may be insufficient to address regional threats to diversity (e.g., Spurgeon et al. 2014), communication and cooperation among stakeholders are key to successful aquatic resource conservation. Implementation of a broadscale multispecies assessment such as this complements traditional reactive management and restoration by providing tools to facilitate communication, cooperation, and coordination among stakeholders and partners, increasing efficiency of future monitoring and management efforts.

Acknowledgments

This study was supported by the Great Plains Landscape Conservation Cooperative (grant F13AP00523), The University of Texas at Austin, and the Texas Parks and Wildlife Department State Wildlife Grant Program (grant TX T-106–1; CFDA# 15.634). Thanks to the thousands of collectors who contributed to the museum and literature databases of fish occurrence. We also thank Steven Krentz, Adam Cohen, James Broska, and the GPLCC steering committee and science team for discussions and concept reviews, and Jon Paul Pierre for help with geodatabase and metadata provision and management.

References

Abell, R., J. D. Allan, and B. Lehner. 2007. Unlocking the potential of protected areas for freshwaters. Biological Conservation 134:48–63.

AFWA (Association of Fish and Wildlife Agencies). 2012. National fish habitat action plan, 2nd Edition. AFWA, Washington, D.C.

Anderson, R. P., and A. Raza. 2010. The effect of the extent of the study region on GIS models of species geographic distributions and estimates of niche evolution: preliminary tests with montane rodents (genus *Nephelomys*) in Venezuela. Journal of Biogeography 37:1378–1393.

Balmford, A., L. Bennun, B. T. Brink, D. Cooper, I. M. Cote, P. Crane, A. Dobson, N. Dudley, I. Dutton, R. E. Green, R. D. Gregory, J. Harrison, E. T. Kennedy, C. Kremen, N. Leader-Williams, T. E. Lovejpy, G. Mace, R. May, P. Mayaux, P. Morling, J. Phillips, K. Redford, T. H. Ricketts, J. P. Rodriguez, M. Sanjayan, P. J. Schei, A. S. Van Jaarsveld, and B. A. Walther. 2005. Ecology: the convention on biological diversity's 2010 target. Science 307:212–213.

Balmford, A., and T. Whitten. 2003. Who should pay for tropical conservation, and how could the costs be met? Oryx 37:238–250.

Birdsong, T. W., D. C. Dauwalter, G. P. Garrett, B. J. Labay, M. Bean, J. Broska, J. Graham, S. Magnelia, K. B. Mayes, M. McGarrity, K. M. Johnson, S. Robertson, T. Thompson, S. Vail-Muse, and J. B. Whittier. 2018. Native fish conservation areas of the southwestern USA: facilitating landscape-scale conservation of aquatic habitats and freshwater fishes. Southeast Aquatic Resources Partnership, Panama City, Florida.

Birdsong, T. W., G. P. Garrett, B. J. Labay, M. G. Bean, P. T. Bean, J. Botros, M. J. Casarez, A. E. Cohen, T. G. Heger, A. Kalmbach, D. A. Hendrickson, S. J. Magnelia, K. B. Mayes, M. E. McGarrity, R. McGillicuddy, M. M. Parker, and S. Robertson. 2019. Texas native fish conservation areas network. Pages 183–229 *in* D. C. Dauwalter, T. W. Birdsong, and G. P. Garrett, editors. Multispecies and watershed approaches to freshwater fish conservation. American Fisheries Society, Symposium 91, Bethesda, Maryland.

Cohen, A.E., B.J. Labay, D.A. Hendrickson, M. Casarez, S. Sakar. 2013. Final Report: Data provision and projected impact of climate change on fish biodiversity within the Desert LCC. U.S. Department of the Interior, Bureau of Reclamation, Desert Landscape Conservation Cooperative. Available: https://repositories.lib.utexas.edu/handle/2152/22475.

Cooper, A. R., Infante, D. M., Daniel, W. M., Wehrly, K. E., Wang, L., and Brenden, T. O. 2017. Assessment of dam effects on streams and fish assemblages of the conterminous USA. Science of the Total Environment 586:879–889.

Costigan, K. H., and M. D. Daniels. 2012. Damming the prairie: human alteration of Great Plains river regimes. Journal of Hydrology 444:90–99.

Dauwalter, D. C., J. S. Sanderson, J. E. Williams, and J. R. Sedell. 2011. Identification and implementation of native fish conservation areas in the upper Colorado River basin. Fisheries 36:278–288.

Díaz, S., S. Demissew, J. Carabias, C. Joly, M. Lonsdale, N. Ash, A. Larigauderie, J. R. Adhikari, S. Arico, A. Báldi, A. Bartuska, I. A. Baste, A. Bilgin, E. Brondizio, K.M.A. Chan, V. E. Figueroa, A. Duraiappah, M. Fischer, R. Hill, T. Koetz, P. Leadley, P. Lyver, G. M. Mace, B. Martin-Lopez, M. Okumura, D. Pacheco, U. Pascual, E. S. Pérez, B. Reyers, E. Roth, O. Saito, R. J. Scholes, N. Sharma, H. Tallis, R. Thaman, R. Watson, T. Yahara, Z. A. Hamid, C. Akosim, Y. Al-Hafedh, R. Allahverdiyev, E. Amankwah, T. S. Asah, Z. Asfaw, G. Bartus, A. L. Brooks, J. Caillaux, G. Dalle, D. Darnaedi, A. Driver, G. Erpul, P. Escobar-Eyzaguirre, P. Failler, A. M. M. Fouda, B. Fu, H. Gundimeda, S. Hashimoto, F. Homer, S. Lavorel, G. Lichtenstein, W. A. Mala, W. Mandivenyi, P. Matczak, C. Mbizvo, M. Mehrdadi, J. P. Metzger, J. B. Mikissa, H. Moller, H. A. Mooney, P. Mumby, H. Nagendra, C. Nesshover, A. A. Oteng-Yeboah, G. Pataki, M. Roué, J. Rubis, M. Schultz, P. Smith, R. Sumaila, K. Takeuchi, S. Thomas, M. Verma, Y. Yeo-Chang, and D. Zlatanova. 2015. The IPBES conceptual framework—connecting nature and people. Current Opinion in Environmental Sustainability [online serial] 14:1–16.

Dodds, W. K., J. S. Perkin, and J. E. Gerken. 2013. Human impact on freshwater ecosystem services: a global perspective. Environmental Science and Technology 47:9061–9068.

Dudgeon, D., A. H. Arthington, M. O. Gessner, Z. Kawabata, D. J. Knowler, C. Leveque, R. J. Naiman, A. H. Prieur-Richard, D. Soto, M. L. Stiassny, and C. A. Sullivan. 2006. Freshwater biodiversity: importance, threats, status and conservation challenges. Biological Reviews 81:163–182.

Elith, J., C. H. Graham, R. P. Anderson, M. Dudík, S. Ferrier, A. Guisan, R. J. Hijmans, F. Huettmann, J. R. Leathwick, A. Lehmann, J. Li, L. G. Lohmann, B. A. Loiselle, G. Manion, C. Moritz, M. Nakamura, Y. Nakazawa, J. McC. Overton, A. T. Peterson, S. J. Phillips, K. S. Richardson, R. Scachetti-Pereira, R. E. Schapire, J. Soberón, S. Williams, M. S. Wisz, and N. E. Zimmermann. 2006. Novel methods improve prediction of species' distributions from occurrence data. Ecography 29:129–151.

Elith, J., S. J. Phillips, T. Hastie, M. Dudík, Y. E. Chee, and C. Yates. 2011. A statistical explanation of MaxEnt for ecologists. Diversity and Distributions 17:43–57.

Esselman, P. C., D. M. Infante, L. Wang, D. Wu, A. R. Cooper, and W. W. Taylor. 2011. An index of cumulative disturbance to river fish habitats of the conterminous United States from landscape anthropogenic activities. Ecological Restoration 29:133 –151.

Faber-Langendoen, D., L. Master, J. Nichols, L. Master, K. Snow, A. Tomaino, R. Bittman, G. Hammerson, B. Heidel, L. Ramsay, A. Teucher, and B. Young. 2009. NatureServe conservation status assessments: methodology for assigning ranks. NatureServe, Arlington, Virginia.

Falke, J. A., K. D. Fausch, R. Magelky, A. Aldred, D. S. Surnford, L. K. Riley, and R. Oad. 2011. The role of groundwater pumping and drought in shaping ecological futures for stream fishes in a dryland river basin of the western Great Plains, USA. Ecohydrology 4:682–697.

Fausch, K. D., C. E. Torgersen, C. V. Baxter, and H. W. Li. 2002. Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes. BioScience 52:483–498.

Fontaine, J. J. 2011. Improving our legacy: incorporation of adaptive management into state wildlife action plans. Journal of Environmental Management 92:1403–1408.

Frimpong, E. A. and P. L. Angermeier. 2009. Fish traits: a database of ecological and life-history traits of freshwater fishes of the United States. Fisheries 34:487–495.

Garrett, G. P., T. W. Birdsong, M. G. Bean, and B. J. Labay. 2019. Chihuahuan native fish conservation areas: a multispecies and watershed approach to preservation of freshwater fish diversity. Pages 231–252 *in* D. C. Dauwalter, T. W. Birdsong, and G. P. Garrett, editors. Multispecies and watershed approaches to freshwater fish conservation. American Fisheries Society, Symposium 91, Bethesda, Maryland.

Gido, K. B., W. K. Dodds, and M. E. Eberle. 2010. Retrospective analysis of fish community change during a half-century of landuse and streamflow changes. Journal of the North American Benthological Society 29:970–987.

Graham, C. H., and R. J. Hijmans. 2006. A comparison of methods for mapping species ranges and species richness. Global Ecology and Biogeography 15:578–587.

Grantham, H. S., R. L. Pressey, J. A. Wells, and A. J. Beattie. 2010. Effectiveness of biodiversity surrogates for conservation planning: different measures of effectiveness generate a kaleidoscope of variation. PLOS (Public Library of Science) ONE [online serial] 5(7):e11430.

Groves, C. R., D. B. Jensen, L. L. Valutis, K. H. Redford, M. L. Shaffer, J. M. Scott, J. V. Baumgartner, J. V., Higgins, M. W. Beck, and M. G. Anderson. 2002. Planning for biodiversity conservation: putting conservation science into practice. Bioscience 52:499–512.

Guisan, A., R. Tingley, J. B. Baumgartner, I. Naujokaitis-Lewis, and P. R. Sutcliffe. 2013. Predicting species distributions for conservation decisions. Ecology Letters 16:1424–1435.

Halpern, B. S., and R. R. Warner. 2003. Matching marine reserve design to reserve objectives. Proceedings of the Royal Society of London B 270:1871–1878.

Hanley, J. A., and B. J. McNeil. 1982. The meaning and use of the area under a receiver operating characteristic (ROC) curve. Radiology 143:29–36.

Haslouer, S. G., M. E. Eberle, D. R. Edds, K. B. Gido, C. S. Mammoliti, J. R. Triplett, J. T. Collins, D. A. Distler, D. G. Huggins, and W. J. Stark. 2005. Current status of native fish species in Kansas. Transactions of the Kansas Academy of Science 108:32–46.

Hendrickson, D. A., S. Sarkar, and A. Molineux. 2010. Final Report: provision and inventory of diverse aquatic ecosystem-related resources for the Great Plains Landscape Conservation Cooperative (GPLCC). U.S. Fish and Wildlife Service Great Plains Landscape Conservation Cooperative. Available: https://lccnetwork.org/resource/final-report-provision-and-inventory-diverse-aquatic-ecosystem-related-resources-gplcc. (April 2019).

Hitt, N. P., and P. L. Angermeier. 2008. Evidence for fish dispersal from spatial analysis of stream network topology. Journal of the North American Benthological Society 27:304–320.

Hitt, N. P., and P. L. Angermeier. 2011. Fish community and bioassessment responses to stream network position. Journal of the North American Benthological Society 30:296–309.

Hoagstrom, C. W., J. E. Brooks, and S. R. Davenport. 2011. A large-scale conservation perspective considering endemic fishes of the North American plains. Biological Conservation 144:21–34.

Hunter, M. L., Jr., and A. Hutchinson. 1994. The virtues and shortcomings of parochialism: conserving species that are locally rare, but globally common. Conservation Biology 8:1163–1165.

Jelks, H. L., S. J. Walsh, N. M. Burkhead, S. Contreras-Balderas, E. Díaz-Pardo, D. A. Hendrickson, J. Lyons, N. E. Mandrak, F. McCormick, J. S. Nelson, S. P. Platania, B. A. Porter, C. B. Renaud, J. J. Schmitter-Soto, E. B. Taylor, and M. L. Warren. 2008. Conservation status of imperiled North American freshwater and diadromous fishes. Fisheries 33:372–407.

Kark, S., A. Tulloch, A. Gordon, T. Mazor, N. Bunnefeld, and N. Levin. 2015. Cross-boundary collaboration: key to the conservation puzzle. Current Opinion in Environmental Sustainability 12:12–24.

Knight, A. T., R. M. Cowling, and B. M. Campbell. 2006. An operational model for implementing conservation action. Conservation Biology 20:408–419.

Knight, A. T., R. M. Cowling, M. Rouget, A. Balmford, A. T. Lombard, and B. M. Campbell. 2008. Knowing but not doing: selecting priority conservation areas and the research–implementation gap. Conservation Biology 22:610–617.

Labay, B., A. E. Cohen, B. Sissel, D. A. Hendrickson, F. D. Martin, and S. Sarkar. 2011. Assessing historical fish community composition using surveys, historical collection data, and species distribution models. PLOS (Public Library of Science) ONE [online serial] 6(9):e25145.

Labay, B., B. Prince, J. Ogren, and K. Tasker. 2018. Rio Grande fishes conservation assessment and mapping. Siglo Group, Austin, Texas.

Lehtomaki, J., and A. Moilanen. 2013. Methods and workflow for spatial conservation planning using Zonation. Environmental Modelling and Software 47:128–137.

Limburg, K. E., R. M. Hughes, D. C. Jackson, and B. Czech. 2011. Human population increase, economic growth, and fish conservation: collision course or savvy stewardship? Fisheries 36:27–35.

Lobo, J. M., A. Jiménez-Valverde, and R. Real. 2008. AUC: a misleading measure of the performance of predictive distribution models. Global Ecology and Biogeography 17:145–151.

Margules, C. R. and R. L. Pressey. 2000. Systematic conservation planning. Nature (London) 405:243–253.

Martinuzzi, S., S. R. Januchowski-Hartley, B. M. Pracheil, P. B. McIntyre, A. J. Plantinga, D. J. Lewis, and V. C. Radeloff. 2014. Threats and opportunities for freshwater conservation under future land use change scenarios in the United States. Global Change Biology 20:113–124.

Moilanen, A. 2007. Landscape zonation, benefit functions and target-based planning: unifying reserve selection strategies. Biological Conservation 134:571–579.

Moilanen, A., B. J. Anderson, A. Arponen, F. M. Pouzols, and C. D. Thomas. 2013, Edge artefacts and lost performance in national versus continental conservation priority areas. Diversity and Distributions 19:171–183.

Moilanen, A., and A. Arponen. 2011. Administrative regions in conservation: balancing local priorities with regional to global preferences in spatial planning. Biological Conservation 144:1719–1725.

Moilanen, A., A. M. A. Franco, R. I. Early, R. Fox, B. Wintle, and C. D. Thomas. 2005. Prioritizing multiple-use landscapes for conservation: methods for large multispecies planning problems. Proceedings of the Royal Society B 272:1885–1891.

Moilanen, A., and B. A. Wintle. 2007. The boundary-quality penalty: a quantitative method for approximating species responses to fragmentation in reserve selection. Conservation Biology 21:355–364.

Moilanen, A., J. Leathwick, and J. Elith. 2008, A method for spatial freshwater conservation prioritization. Freshwater Biology, 53: 577–592.

Neeson, T. M., M. C. Ferris, M. W. Diebel, P. J. Doran, J. R. O'Hanley, and P. B. McIntyre. 2015. Enhancing ecosystem restoration efficiency through spatial and temporal coordination. Proceedings of the National Academy of Sciences of the United States of America 112:6236–6241.

Palomo, I., C. Montes, B. Martín-López, J. A. González, M. García-Llorente Paloma Alcorlo, and M. R. García Mora. 2014. Incorporating the social–ecological approach in protected areas in the Anthropocene. BioScience 64:181–191.

Parham, R. W. 2009. Structure of assemblages and recent distribution of riverine fishes in Oklahoma. The Southwestern Naturalist 54:382–399.

Perkin, J. S., K. B. Gido, A. R. Cooper, T. F. Turner, M. J. Osborne, E. R. Johnson, and K. B. Mayes. 2015a. Fragmentation and dewatering transform Great Plains stream fish communities. Ecological Monographs 85:73–92.

Perkin, J. S., K. B. Gido, K. H. Costigan, M. D. Daniels, and E. R. Johnson. 2015b. Fragmentation and drying ratchet down Great Plains stream fish diversity. Aquatic Conservation: Marine and Freshwater Ecosystems 25:639–655.

Perkin, J. S., K. B. Gido, J. A. Falke, K. D. Fausch, H. Crockett, E. R. Johnson, and J. Sanderson. 2017. Groundwater declines are linked to changes in Great Plains stream fish assemblages. Proceedings of the National Academy of Sciences of the United States of America 114:7373–7378.

Phillips, S. J., R. P. Anderson, and R. E. Schapire. 2006. Maximum entropy modeling of species geographic distributions. Ecological Modelling 190:231–259.

Phillips, S. J., and M. Dudík. 2008. Modeling of species distributions with Maxent: new extensions and a comprehensive evaluation. Ecography 31:161–175.

Pierce, S. M. 2003. The STEP handbook: integrating the natural environment into land use decisions at the municipal level: towards sustainable development. University of Port Elizabeth, Terrestrial Ecology Research Unit, Report 47, Port Elizabeth, South Africa.

Pracheil, B. M., M. A. Pegg, L. A. Powell, and G. E. Mestl. 2012. Swimways: protecting Paddlefish through movement-centered management. Fisheries 37:449–457.

Pressey, R. L., M. Cabeza, M. E. Watts, R. M. Cowling, and K. A. Wilson. 2007. Conservation planning in a changing world. Trends in Ecology and Evolution 22:583–592.

Renner, I. W., and D. I. Warton. 2013. Equivalence of MAXENT and Poisson point process models for species distribution modeling in ecology. Biometrics 69:274–281.

SARP (Southeast Aquatic Resources Partnership). 2008. Southeast aquatic habitat plan. SARP, Social Circle, Georgia.

SARP (Southeast Aquatic Resources Partnership). 2014. Conserving fish habitat from rivers to the sea: the story of the Southeast Aquatic Resources Partnership. SARP, Social Circle, Georgia.

Simon, T. P. 1999. Assessment of Balon's reproductive guilds with application to Midwestern North American freshwater fishes. Pages 97–121 *in* T. P. Simon, editor. Assessing the sustainability and biological integrity of water resources using fish communities. CRC Press, Boca Raton, Florida.

Soberon, J. M., and J. K. Sarukhan. 2009. A new mechanism for science-policy transfer and biodiversity governance? Environmental Conservation 36:265–267.

Spurgeon, J. J., R. H. Stasiak, G. R. Cunningham, K. L. Pope, and M. A. Pegg. 2014. Status of native stream fishes within selected protected areas of the Niobrara River in western Nebraska. Great Plains Research 24:71–78.

Steward, D. R., P. J. Bruss, X. Yang, S. A. Staggenborg, S. M. Welch, and M. D. Apley. 2013. Tapping unsustainable groundwater stores for agricultural production in the High Plains Aquifer of Kansas, projections to 2110. Proceedings of the National Academy of Sciences of the United States of America 110:E3477–E3486.

Stewart, D. R., Z. E. Underwood, F. J. Rahel, and A. W. Walters. 2018. The effectiveness of surrogate taxa to conserve freshwater biodiversity. Conservation Biology 32:183–194.

Strayer, D. L., and D. Dudgeon. 2010. Freshwater biodiversity conservation: recent progress and future challenges. Journal of the North American Benthological Society 29:344–358.

Thornbrugh, D. J., S. G. Leibowitz, R. A. Hill, M. H. Weber, Z. C. Johnson, A. R. Olsen, and D. V. Peck. 2018. Mapping watershed integrity for the conterminous United States. Ecological indicators 85:1133–1148.

U.S. Fish and Wildlife Service. 2005. Final designation of critical habitat for the Arkansas River basin population of the Arkansas River Shiner (*Notropis girardi*). Federal Register 70:107(13 October 2005):59808–59846.

U.S. Fish and Wildlife Service. 2014. Endangered and threatened wildlife and plants; designation of critical habitat for Sharpnose Shiner and Smalleye Shiner. Federal Register 79:149(4 August 2014):45242–45271.

Vane-Wright, R. I. 1996. Systematics and the conservation of biological diversity. Annals of the Missouri Botanical Garden 83:47–57.

Vorosmarty, C. J., P. B. McIntyre, M. O. Gessner, D. Dudgeon, A. Prusevich, P. Green, S. Glidden, S. E. Bunn, C. A. Sullivan, C. Reidy Liermann, and P. M. Davies. 2010. Global threats to human water security and river biodiversity. Nature (London) 467:555–561.

Wilde, G. R., and A. C. Urbanczyk. 2013. Relationship between river fragment length and persistence of two imperiled Great Plains cyprinids. Journal of Freshwater Ecology 28:445–451.

Williams, J. E., R. N. Williams, R. F. Thurow, L. Elwell, D. P. Philipp, F. A. Harris, J. L. Kershner, P. J. Martinez, D. Miller, G. H. Reeves, C. A. Frissell, and J. R. Sedell. 2011. Native fish conservation areas: a vision for large-scale conservation of native fish communities. Fisheries 36:267–277.