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Conservation planning in an uncertain climate: Identifying projects that remain valuable and feasible across future scenarios

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Abstract

1. Conservation actors face the challenge of allocating limited resources despite uncertainty about future climate conditions. In many cases, the potential value and feasibility of proposed projects vary across climate scenarios. A key goal is to identify areas where conservation outcomes can balance both environmental and human needs.

2. We developed a conservation prioritization framework that jointly considers the value and feasibility of candidate projects across future climate scenarios. We then applied this framework to the challenge of meeting environmental flow targets across the Red River basin of the south-central United States.

3. To estimate the conservation feasibility of meeting environmental flow goals in a river reach in each climate scenario, we used a basin-wide hydrologic planning tool to quantify the reduction in societal water usage needed to meet environmental flow targets. To estimate the biodiversity value of each river reach in each climate scenario, we used climate-driven species distribution models and species' conservation status.

4. We found that river reaches in the east-central portion of the basin may be good candidates for conservation investments, because they had high biodiversity value and high sociopolitical feasibility in all future climate scenarios. In contrast, sites in the arid western reaches of the basin had high biodiversity value, but low feasibility of achieving environmental flow goals.

5. Our framework should have broad applicability given that the value and feasibility of conservation projects vary across climate scenarios in ecosystems around the world. It may serve as a coarse filter to identify sites for more detailed analyses and could be integrated with complementarity-based approaches to conservation planning to balance species' representation across projects.

KEYWORDS
climate change, climate uncertainty, conservation feasibility, conservation planning, conservation prioritization, environmental flows

1 | INTRODUCTION

Climate change and human activities are impacting ecosystems globally, directly contributing to biodiversity loss and threatening human well-being (Díaz et al., 2019; Green et al., 2019; Reside et al., 2018; Scheffers & Pecl, 2019; Tickner et al., 2020). The conservation community has responded to this crisis by developing spatial conservation prioritization (hereafter 'prioritization') frameworks to guide conservation investments (Kukkala & Moilanen, 2013; Sinclair et al., 2018). Broadly defined, these frameworks aim to identify strategies for allocating resources among sites to maximize return-on-investment in the protection of biodiversity or ecosystem services. It is increasingly recognized that prioritizations must account for uncertain future climatic conditions (Carvalho et al., 2011; Heller & Zavaleta, 2009; Jones et al., 2016) and consider societal uses and values of biodiversity and natural resources (Guerrero et al., 2018; Karimi et al., 2017; Knight et al., 2010; Whitehead et al., 2014). However, conservation planning frameworks often fail to consider factors like competing societal values for shared resources, and organizational and resource governance processes, which may result in a failure to step from research to implementation phases of planning (McIntosh et al., 2018; Moon et al., 2014; Toomey et al., 2017). Conservation planning thus faces a pressing challenge to effectively inform decision-making under climatic uncertainty while being conscious of the societal context in which conservation actions are being recommended.

When developing a conservation prioritization, accounting for the feasibility of a conservation project is necessary to overcome the 'implementation crisis', wherein sophisticated conservation plans are developed but not implemented (Knight et al., 2008; Toomey et al., 2017). Accounting for conservation feasibility in the assessment phase of planning could help complement the *where* to act (i.e. the identification of priority sites) with the *how* to act (i.e. the procurement of adequate funding, support, permissions and resources to facilitate implementation; Adams et al., 2019). Conservation feasibility may be broadly defined as the likelihood of successful implementation of a conservation action at a target area or site (Guerrero & Wilson, 2017; Karimi et al., 2017; Popejoy et al., 2018). Thus, locations with high conservation feasibility are likely to be those locations where the sociopolitical costs of conservation actions are low. Many sociopolitical factors (i.e. social attitudes and public values that influence resource management, economics, policymaking and governance structures) contribute to conservation feasibility, including government organizations, resource users, policies, funding and institutional mandates (Guerrero & Wilson, 2017). Recent studies focus on assessing conservation feasibility by examining stakeholders' or landowners' willingness to participate in a conservation program (Kwayu et al., 2014; Ma et al., 2012; Wunder et al., 2018) or sell land or water rights (Adams et al., 2014; Guerrero et al., 2010; Tulloch et al., 2014; Wang et al., 2017), whether social and ecological values align (Bagstad et al., 2016; Bryan et al., 2011; Whitehead et al., 2014), social–political and resource governance structures and contexts (Jupiter et al., 2017), conservation conflicts and sociopolitical

resistance to conservation actions (Rastogi et al., 2014; Von Essen & Hansen, 2015) and trade-offs between shared resources (Zamani Sabzi, et al., 2019).

Effective conservation prioritization under climate change requires strategies that operate under climatic uncertainty (Bates et al., 2019; He & Silliman, 2019; Jones et al., 2016; Kukkala & Moilanen, 2013; Meir et al., 2004). In this context, uncertainty about future climate conditions stems from variability of projected model or scenario outcomes (Reside et al., 2018). While some researchers argue that climate projections ought to be ignored in conservation planning because they are hampered by uncertainties (i.e. forecast-free approaches, Groves et al., 2012), others have attempted to identify strategies that are robust to climate uncertainty by assessing how conservation priorities vary across a wide range of future climate scenarios (Araújo & Luoto, 2007; Araújo & New, 2007; Conroy et al., 2011; Lawler & Michalak, 2018). In practice, researchers might use predictive species distribution models to identify locations that have consistently high biodiversity value across a wide range of future climate scenarios (Carvalho et al., 2011; Kujala et al., 2013). These sites are likely to be good choices for conservation investments given that biodiversity values are high in all future scenarios.

Simultaneously accounting for climate uncertainty, biodiversity value and feasibility is challenging because biodiversity value is often decoupled from the sociopolitical costs of conservation actions (Bonebrake et al., 2018; Scheffers & Pecl, 2019) and both factors vary independently across future scenarios. Some examples of sociopolitical factors that vary independently of biodiversity value include polarized views of the importance of conservation (Coffey & Joseph, 2013), the exclusion of some stakeholders from planning and negotiations (Foote et al., 2007), funding and economic costs (Balmford et al., 2003) and sociopolitical borders that exacerbate shared resource conflicts (Dallimer & Strange, 2015). Because biodiversity value and conservation feasibility vary independently across locations and across future climate scenarios, researchers must account for these factors separately to identify areas at the intersection of high biodiversity value and low sociopolitical cost.

Despite many practical and theoretical advances in conservation planning, studies have disproportionately focused on terrestrial and marine ecosystems (Linke et al., 2019). Conservation initiatives are difficult to implement in freshwater ecosystems because of complex factors that influence planning, such as uncertainty in water resources under future climate change scenarios (McPherson, 2016; Moilanen et al., 2008; Wilby et al., 2010), species range shifts under climate change (Araújo et al., 2004; Carvalho et al., 2011), shared resource conflicts (Pahl-Wostl et al., 2013; Zamani Sabzi, Rezapour, et al., 2019) and complex water resource governance systems (Daher et al., 2019; Portney et al., 2017). One approach to improving biodiversity outcomes in freshwater ecosystems is through accelerating the implementation of environmental flows (Tickner et al., 2020). Environmental flows are a system to maintain or restore ecologically relevant aspects of hydrologic regimes that have been altered by human infrastructure or practices; this may include changes in the quantity, timing and variability of river flows (Arthington et al., 2018). Maintaining environmental flows

can significantly improve biodiversity outcomes by restoring functional ecological processes that freshwater species rely on for their life histories (Acreman et al., 2014; Mims & Olden, 2012; Nilsson & Renöfält, 2008; Olden et al., 2014; Vaughn, 2018; Vaughn et al., 2015). Water-limited river basins are good model systems for studying the challenges of conservation planning because of the projected decrease in water availability under future climate change (Christensen et al., 2004; Grafton et al., 2011; Ma et al., 2008), projected increase in future societal water demand (Flörke et al., 2018) and varying levels of sociopolitical resistance to environmental flow regulations and policies (Mekonnen & Hoekstra, 2016; Pittock & Finlayson, 2011). As a result of the inherently coupled relationship between humans and freshwater ecosystems, there are significant trade-offs between human needs and conservation outcomes (Crespo et al., 2019; Guo et al., 2019; Sivapalan et al., 2014; Zamani Sabzi, Rezapour, et al., 2019; Ziv et al., 2012).

The Red River basin in the south-central United States exemplifies the difficulties of accounting for complex factors and uncertainty in freshwater conservation planning. Here, climate change is expected to decrease overall precipitation and river flows, but there is considerable uncertainty about where and when water shortages may occur (Bertrand & McPherson, 2019; Xue et al., 2016). As a result, societal (i.e. municipal, industrial and agricultural) and ecosystem (i.e. environmental flows) water needs cannot be fully met unless reductions in societal uses are implemented (Zamani Sabzi, Rezapour, et al., 2019). Additionally, fish and freshwater mussel species are projected to decline and undergo range shifts because of climate- and human-induced changes in water availability that inhibit habitat connectivity and produce extreme thermal regimes (Gill et al., 2020; Perkin et al., 2015, 2017). Thus, conservation actors in this region must account for future climatic uncertainty and consider conservation feasibility in planning for freshwater ecosystems but lack the appropriate guidance on where to invest conservation resources.

In this study, we developed a prioritization framework for identifying target sites for implementing conservation actions that remain valuable and feasible across a range of future climate scenarios. Our prioritization framework represents a simple, flexible and low-data method that can be used in the assessment phase of conservation planning to rapidly winnow the number of sites under consideration. We then applied this framework to the challenge of prioritizing water conservation actions across a drought-prone river basin, the Red River, in the south-central United States. In this application, we sought to identify river reaches that consistently had high biodiversity value and conservation feasibility across a range of possible future climate scenarios. We used data from recent high-resolution hydrologic and climatic modelling in the Red River to parameterize a mathematical optimization model for allocating water conservation incentives across the basin. From these models, we estimated the biodiversity value of river reaches by examining fish species distributions and their endangerment criteria below 38 major reservoirs, as well as the conservation feasibility of river reaches by estimating the sociopolitical cost of fully meeting environmental flow targets. This case study demonstrates the utility of our framework for identifying conservation projects that remain valuable and feasible across a range of climate scenarios.

2 | METHODS

2.1 | Prioritization framework

We developed a prioritization framework that integrates measures of biodiversity value and conservation feasibility, and how they vary across future climate scenarios. (Figure 1). Our framework extends other two-dimensional planning models that consider biodiversity value and feasibility (Guerrero & Wilson, 2017; Popejoy et al., 2018) by exploring how both factors vary across future climate scenarios. Thus, our framework identifies sites that consistently have high biodiversity value and high conservation feasibility across a range of future climate conditions.

In our framework, the biodiversity value (vertical axis) of a site can be quantified using any common ecological measure (e.g. species richness, abundance, occupancy, habitat use and suitability, species' risk of endangerment and ecosystem services) or an index of biodiversity derived from these measures (Capmourteres & Anand, 2016). Thus, the axis is broadly defined to accommodate the many different ways that humans value, use and prioritize biodiversity (Manfredo et al., 2016). The conservation feasibility (horizontal axis) of a site can be estimated by considering the sociopolitical factors that contribute to the likelihood of successful implementation of a conservation action (Popejoy et al., 2018).

Partitioning the biodiversity-feasibility space into user-defined quadrants reveals four tiers of conservation priority (Figure 1). Sites with
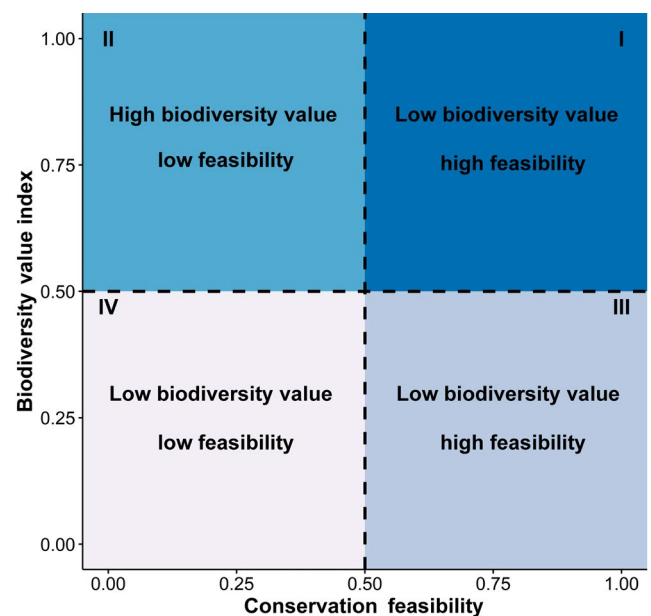

FIGURE 1 General prioritization model for identifying conservation projects with varying levels of biodiversity value and conservation feasibility. The vertical and horizontal lines that delimit the quadrants may be shifted to create larger or smaller quadrants as needed for an application

high biodiversity value and high conservation feasibility (quadrant I) are likely to be the highest priority for investment because conservation actions at these sites would have low sociopolitical costs but high benefits to biodiversity. Sites with high biodiversity value and low conservation feasibility (quadrant II) are of lower priority and represent areas with significant social–ecological trade-offs. Conservation actions at these sites would provide high benefits to biodiversity but may also have high sociopolitical costs. Sites in quadrant III are low cost, low reward: conservation actions may have low sociopolitical costs but also low benefits to biodiversity. Finally, sites with low biodiversity value and low conservation feasibility (quadrant IV) are high cost, low reward locations that are unlikely to be good choices for conservation investments.

By applying this prioritization framework across multiple future climate scenarios, decision-makers can explore how site-level conservation priority varies spatially and temporally. For example, if a site's positional variation is small across future climate scenarios (i.e. it remains within one quadrant), then there is low uncertainty in measures of biodiversity value and conservation feasibility across future climate scenarios. We also emphasize that the horizontal and vertical lines that delimit the quadrants are arbitrarily placed at the midpoint of each axis. They may be shifted to create large or smaller quadrants to fit the needs of a particular application.

Our prioritization framework represents a simple, flexible method for weighing the trade-offs between human and environmental aspects of a target conservation objective across future climate scenarios. After conservation objectives have been developed and potential target sites identified, decision-makers could use this prioritization framework as a coarse-scale filter to eliminate low priority sites from consideration. Because conservation planners typically weigh a complex set of incommensurable factors in addition to the two considered here in this framework, and many of those factors are difficult to quantify and measure at large spatial scales, our approach can reduce the cost and data needs of typical planning studies by reducing the number of sites that warrant more careful study. Adopting this prioritization framework during the assessment phase of conservation planning could benefit conservation outcomes and improve implementation success through (a) integrating social and ecological data to assess the conservation problem from a systems perspective, (b) identifying key sociopolitical factors that contribute to conservation feasibility (i.e. the likelihood of conservation implementation and success), (c) identifying site-level variation in conservation priority with a focus on identifying high priority sites and (d) identifying consistent outcomes under future climate scenarios (Ban et al., 2013; Guerrero & Wilson, 2017; Kujala et al., 2013; Moon et al., 2014).

2.2 | Study system

Our study focuses on the Red River in the south-central United States. The Red River is a semi-arid, drought-prone river where water availability is geographically skewed: western reaches are arid and can receive as little as 400 mm of rain per year, while eastern reaches can receive up to 1,600 mm (PRISM Climate Group, 2019). Additionally, the combination of consumptive societal water use and extreme thermal and flow regimes in this region can create harsh conditions for aquatic biota (DuBose et al., 2019; Matthews & Marsh-Matthews, 2017; Matthews et al., 2005). The Red River has a high density of reservoirs which were constructed for flood control and to provide water storage to meet societal water demands. In this study, we focus on river reaches downstream of 38 major reservoirs in the basin to identify reaches of high conservation priority under climatic and hydrologic uncertainty (Zamani Sabzi, Rezapour, et al., 2019).

2.3 | Downscaled climate projections

To explore how biodiversity value and conservation feasibility might vary across future climate scenarios, we used recent species distribution (Gill et al., 2020) and hydrologic models (Zamani Sabzi, Moreno, et al., 2019; Zamani Sabzi, Rezapour, et al., 2019) parameterized with basin-scale high-resolution climate projections (Bertrand & McPherson, 2018, 2019; Xue et al., 2016). Briefly, McPherson (2016) used statistical downscaling of global climate model outputs from the Coupled Model Inter-comparison Project Phase 5 (CMIP5) for both historical (1961–2010) and future (2010–2099) time series to estimate daily air temperature and precipitation across the basin at a 1/8° raster resolution for nine future climate scenarios. These nine climate scenarios resulted from taking all combinations of three general circulation models (GCMs; CCSM4, MIROC5 and MPI-ESM-LR) and three representative concentration pathways (RCPs; 2.6, 4.5 and 8.5 W/m^2). These nine scenarios represent a range of plausible air temperature and precipitation biases and climate sensitivities over the south-central United States (Sheffield et al., 2013). Other scenarios (e.g. those that include RCP 6.0) were deemed of lower value to decision-makers in the region and not investigated due to computational and personnel cost constraints (Bertrand & McPherson, 2019).

Air temperature and precipitation estimates for each of these nine climate scenarios were then used to fit a variable infiltration capacity (VIC) hydrologic model to simulate daily surface runoff, streamflow and reservoir storage for both historical and future time periods (Xue et al., 2016). VIC is a rainfall-runoff model that uses climate variable inputs (precipitation and temperature), estimates of infiltration and soil moisture, and reservoir storages to estimate evapotranspiration and surface runoff at a daily time step across the basin (Liang et al., 1996). Details of the VIC model calibration process for the Red River basin are given by Xue et al. (2016). VIC model outputs were then used to estimate future fish species distributions (Section 2.4; Gill et al., 2020) and reservoir storage and environmental flows (Section 2.6; Zamani Sabzi, Moreno, et al., 2019) for the 2040–2060 time series.

2.4 | Fish species distribution models

To determine the biodiversity value of river reaches downstream of the 38 major reservoirs in the basin, we used the predicted probability of occurrence outputs from a suite of species distribution models (SDMs) for 31 fish species native to the Red River (Gill et al., 2020). While there

are over 170 fish species native to this catchment, we focus on a suite of 31 species that represent a range of spawning modes, conservation concern and recreational/societal value based on input from regional fisheries managers (K. Kuklisnki and T. Spark, OK Department of Wildlife Conservation; and B. Matthews and E. Marsh-Matthews, U. Oklahoma). To build SDMs, Gill et al. (2020) used historical fish occurrence records and downscaled climatic and hydrologic data for the Red River basin (Bertrand & McPherson, 2019; Xue et al., 2016) to project fish distributions across a range of future climate scenarios using Maxent.

The outputs of the SDMs are gridded raster datasets in which the value associated with each raster grid cell is the predicted probability of occurrence of a species. To determine which species might benefit from environmental flow releases below each reservoir, we averaged the probability of occurrence values for the five grid cells (a total area of 144 km²) downstream of each reservoir for each species under each climate scenario (Figure 2). These five grid cells are a fraction of each fish species' full distributional range. Given the 1/8° resolution of the raster grid, the five grid cells downstream of each reservoir are approximately equal to 50 km of river channel length. This channel length is both the

minimum distance between the two closest reservoirs in the basin, and an approximate maximum distance downstream that reservoir water releases can affect flow regulation, thermal stratification and fish habitat (Kinsolving & Bain, 1993; Sinokrot et al., 1995). Though our prioritization focuses on fish, improvements to instream flows would also benefit other drought-sensitive regional biota (e.g. mussels; Vaughn et al., 2015).

2.5 | Biodiversity value index (BVI)

We developed an index to assign biodiversity value to each location based on species probability of occurrence in the downstream river reach and the conservation status of each species. In this index, each species' probability of occurrence at each river reach across each climate scenario is weighted by multiple conventions and directives that define criteria and conditions for species risk of endangerment (i.e. IUCN vs. NatureServe), and the spatial scale at which conservation status is being evaluated (i.e. state, national or global scale; Table 1). The BVI is calculated as:
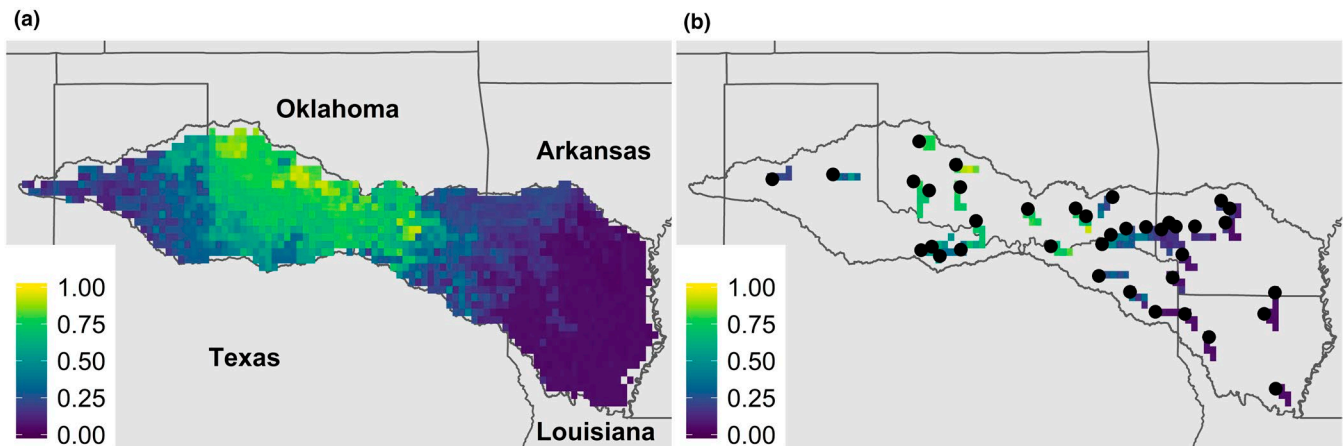

FIGURE 2 Map of the Red River basin depicting how predicted probabilities of occurrence from a species distribution model (a) were masked to 50-km river reaches below each reservoir (b). Colour represents predicted probability of occurrence values for one species. Values closer to 1 (yellow) indicate a high predicted probability of occurrence while values closer to 0 (purple) indicate a low probability of occurrence

TABLE 1 Data sources, criteria of endangerment, conservation status and weights used in creating the biodiversity value index (BVI)

IUCN Red List (R^a)		NatureServe Global Status (G)			IUCN Population Trend (T)		NatureServe State Status (S)		
Category	w_{ij}	Status	Category	w_{ij}	Trend	w_{ij}	Status	Category	w_{ij}
Extinct	0	GX	Presumed extinct	0	Unknown	0	SX	Presumed extinct	0
Extinct in wild	0	GH	Possibly extinct	0	Stable	0.5	SH	Possibly extinct	0
Critically endangered	1	G1	Critically imperilled	1	Decreasing	1	S1	Critically imperilled	1
Endangered	0.8	G2	Imperilled	0.8	Increasing	0	S2	Imperilled	0.8
Vulnerable	0.6	G3	Vulnerable	0.6			S3	Vulnerable	0.6
Near threatened	0.4	G4	Apparently secure	0.4			S4	Apparently secure	0.4
Least concern	0.2	G5	Secure	0.2			S5	Secure	0.2
Data deficient	0								
Not evaluated	0								

[a]R, G, T, S abbreviate the conventions used in the BVI.

$$BVI = \frac{\sum_i^{s=31} \sum_j^{n=4} p_i\,w_{ij}}{s \cdot n},$$

where s is the number of species considered, n is the number of conventions or directives used to derive endangerment weights w_{ij} and p_i is the probability of occurrence values for each species. The BVI varies between 0 (when the downstream river reach has low predicted probability of occurrences for all species with low conservation value) and 1 (when the downstream river reach has high predicted probability of occurrences for all species with high conservation value).

We used multiple conventions and directives to derive the conservation status weight to capture the multiple spatial scales at which conservation priorities operate (Bergerot et al., 2008).

For each convention or directive, we derive conservation status weights based on the risk of endangerment categories (Table 1). For example, we used risk of endangerment categories according to the International Union for the Conservation of Nature (IUCN) Red List of Threatened Species™ (IUCN, 2019), NatureServe® Global Conservation Status (Natureserve, 2019) and state rankings for the state in which the downstream river reach occurs (Table 2, accessed from the NatureServe® Explorer) to account for varying conservation status criteria at the global, national and state scale (Akçakaya et al., 2006; Halmy & Salem, 2015; Miller et al., 2007). The fourth category we use represents the population status of the species according to the IUCN Red List of Threatened Species™.

TABLE 2 Weights for each index term by species considered in the biodiversity value index (BVI)

Species	Common name	R_i[a]	G_i	T_i	S_i[b] (OK)	S_i (TX)	S_i (AR)	S_i (LA)
Ameiurus melas	Black bullhead	0.2	0.2	0.5	0.2	0.2	0.4	0.2
Cyprinella lutrensis	Red shiner	0.2	0.2	0.5	0.2	0.2	0.6	0.2
Cyprinodon rubrofluviatilis	Red River pupfish	0.2	0.2	0.5	0.2	0.4	0.0	0.0
Etheostoma collettei	Creole darter	0.2	0.4	0.5	0.6	0.0	0.4	0.4
Etheostoma radiosum	Orangebelly darter	0.2	0.2	0.5	0.2	0.6	0.4	0.0
Fundulus zebrinus	Plains killifish	0.2	0.2	0.0	0.4	0.2	0.0	0.0
Gambusia affinis	Western mosquitofish	0.2	0.2	0.5	0.2	0.2	0.4	0.2
Hybognathus placitus	Plains minnow	0.2	0.4	1.0	0.2	0.4	0.0	0.0
Ictalurus furcatus	Blue catfish	0.2	0.2	0.5	0.2	0.2	0.4	0.2
Lepomis cyanellus	Green sunfish	0.2	0.2	0.5	0.2	0.4	0.2	0.2
Lythrurus snelsoni	Ouachita shiner	0.2	0.6	0.5	0.8	0.0	0.0	0.0
Macrhybopsis australis	Prairie chub	0.6	0.6	0.0	0.4	0.0	0.0	0.0
Macrhybopsis hyostoma	Shoal chub	0.2	0.2	0.5	0.2	0.0	0.8	0.6
Macrhybopsis storeriana	Silver chub	0.2	0.2	0.5	0.4	0.6	0.6	0.4
Micropterus dolomieu	Smallmouth bass	0.2	0.2	0.5	0.2	0.0	0.4	0.0
Micropterus punctulatus	Spotted bass	0.2	0.2	0.5	0.2	0.2	0.4	0.2
Micropterus salmoides	Largemouth bass	0.2	0.2	0.5	0.2	0.2	0.4	0.2
Morone saxatilis	Striped bass	0.2	0.2	0.0	0.0	0.0	0.0	0.4
Notropis atherinoides	Emerald shiner	0.2	0.2	0.5	0.2	0.4	0.4	0.2
Notropis atrocaudalis	Blackspot shiner	0.2	0.4	0.5	1.0	0.6	0.6	0.6
Notropis bairdi	Red River shiner	0.4	0.4	0.0	0.6	0.6	0.0	0.0
Notropis boops	Bigeye shiner	0.2	0.2	0.5	0.2	0.0	0.4	0.6
Notropis ortenburgeri	Kiamichi shiner	0.0	0.2	0.0	0.6	0.0	0.8	0.0
Notropis perpallidus	Peppered shiner	0.6	0.6	1.0	0.8	0.0	0.8	0.0
Notropis potteri	Chub shiner	0.2	0.4	1.0	0.4	0.4	1.0	0.0
Notropis stramineus	Sand shiner	0.2	0.2	0.5	0.2	0.6	0.0	0.0
Notropis suttkusi	Rocky shiner	0.0	0.6	0.0	0.4	0.0	0.0	0.0
Percina copelandi	Channel darter	0.2	0.4	0.5	0.4	0.0	0.4	0.8
Percina pantherina	Leopard darter	0.8	0.8	0.5	1.0	0.0	1.0	0.0
Phenacobius mirabilis	Suckermouth minnow	0.2	0.2	0.5	0.4	0.4	1.0	1.0
Pteronotropis hubbsi	Bluehead shiner	0.4	0.6	1.0	1.0	1.0	0.6	0.8

[a]R, G, T, S abbreviate the conventions used in the BVI (see Table 1).

[b]NatureServe State-level weights for each state in the Red River Basin (OK, Oklahoma; TX, Texas; AR, Arkansas; LA, Louisiana).

2.6 | Conservation feasibility

To estimate the feasibility of meeting environmental flow targets in river reaches downstream of each reservoir, we used estimates of potential societal water satisfaction derived from a water-balance hydrologic model (Zamani Sabzi, Rezapour, et al., 2019). Briefly, the hydrologic model uses an optimization framework for allocating scarce water resources across the entire river basin in a way that balances societal (agricultural, industrial and municipal) and ecosystem (environmental flows) water needs. Using this model, Zamani Sabzi, Rezapour, et al. (2019) delineated Pareto trade-off curves that represent the mathematically optimal trade-offs between meeting societal and environmental flow targets across the basin. Estimates of societal water demands that underlie this analysis are based on the analysis of existing water rights and consumption across the basin (McPherson et al., 2016). Estimates of environmental flow demands are based on the retrospective analysis of historical flow regimes in each river reach (Zamani Sabzi, Rezapour, et al., 2019). In the model, societal and environmental water satisfaction values range from 0 (completely unsatisfied) to 1 (fully satisfied). For a given weight that determines the relative importance of meeting societal versus environmental flow targets, the model identifies a mathematically optimal distribution of water across the basin that jointly maximizes societal and ecosystem water satisfaction. Exploring a range of relative weights for societal versus environmental flow targets results in a Pareto trade-off curve between meeting societal versus environmental flow targets (Figure 3).

We estimated the conservation feasibility of meeting environmental flow targets below each reservoir by analysing individual Pareto trade-off curves at each reservoir. On each curve, we identified the highest possible societal water satisfaction value that could be achieved while fully meeting environmental flow goals. Conceptually, this point represents the sociopolitical challenges of meeting environmental flow goals below each reservoir. If it is possible to fully meet environmental flow targets while also maintaining high societal water satisfaction, then water conservation likely has high feasibility because it would entail little sociopolitical conflict. Conversely, in locations where it is not possible to maintain high societal water satisfaction while meeting environmental flow targets, water conservation likely has low feasibility because it would entail significant sociopolitical costs. In Lake Texoma, for example, the hydrologic model suggests that 100% satisfaction of both environmental and societal water targets is possible (Figure 3a). Thus, meeting environmental flow goals below this reservoir should have high sociopolitical feasibility because it would not require any reduction in societal water use. In Atoka Reservoir, however, fully meeting environmental flow targets would result in societal water satisfaction of 25% at most (Figure 3b). As a result, we estimate that meeting environmental flow goals below Atoka Reservoir has low sociopolitical feasibility because it would require a dramatic reduction in societal water use. Kemp Reservoir (Figure 3c) represents an extreme case in which environmental flow goals cannot be met even by complete elimination of societal water consumption.

2.7 | Identification of high conservation priority river reaches

We identified high conservation priority river reaches by applying our prioritization framework (Figure 1) across the nine climate
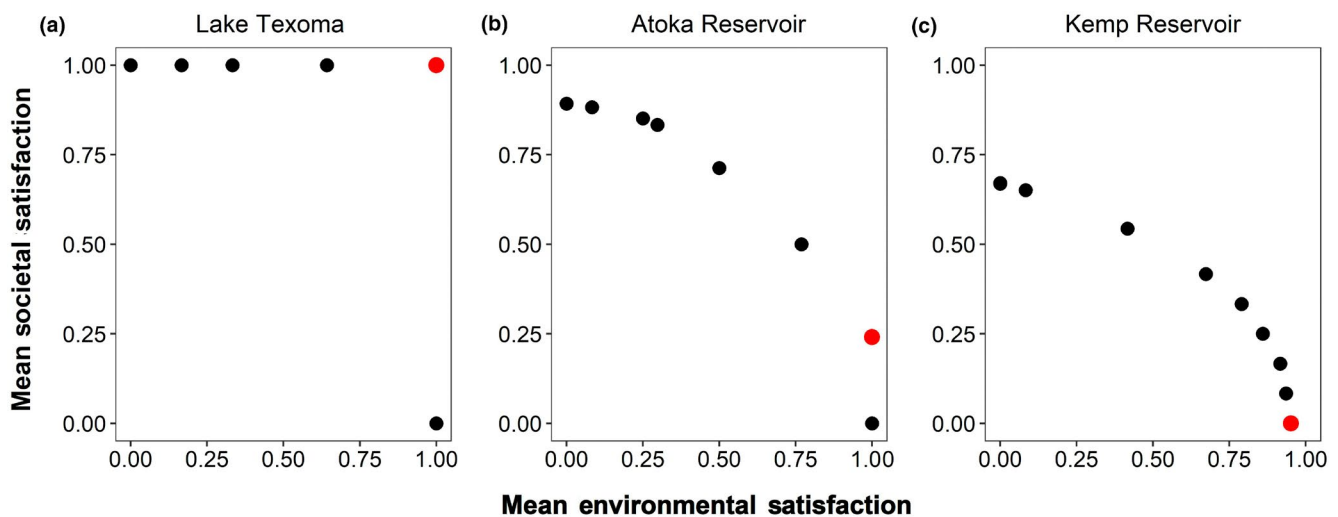

FIGURE 3 Examples of Pareto trade-off curves used to estimate conservation feasibility. Pareto trade-off curves were generated from a basin-wide hydrologic planning model that considered the satisfaction of societal (y-axis) and environmental (x-axis) water needs at each reservoir in the basin. Societal satisfaction levels are based on meeting water allocations for municipal and agricultural users, and environmental satisfaction levels are based on meeting environmental flow targets below each reservoir. Red points indicate the highest possible societal water satisfaction value that could be achieved when environmental flow goals are fully met. For example, at Lake Texoma (a), both societal and environmental water needs can be satisfied when environmental flows are prioritized. At Atoka Reservoir (b), only 25% of societal water needs can be met when environmental flows are prioritized. At Kemp Reservoir (c), none of societal water needs can be met when environmental flows are prioritized

scenarios detailed in Section 2.3. For each climate scenario, we plotted the biodiversity value (i.e. BVI scores) of each river reach against conservation feasibility (i.e. the highest possible societal water satisfaction when environmental flows are fully met). For each river reach, we quantified the proportion of future climate scenarios in which that reach fell into each of the four priority quadrants of Figure 1.

In applying this framework (Figure 1) to the Red River, we used a societal water satisfaction value of 0.50 as the breakpoint between high and low sociopolitical costs. Given that our goal was to use the prioritization framework as a coarse filter to identify sites for more detailed analysis, we chose to use a low breakpoint to retain a larger number of sites for further consideration. We acknowledge that sites with a societal water satisfaction of 0.50 likely have substantial sociopolitical costs associated with water conservation actions (e.g. payment for ecosystem services; Sone et al., 2019). Similarly, we used the median of the BVI values in each scenario as the breakpoint between high and low biodiversity values.

3 | RESULTS

The biodiversity value of river reaches (as measured by BVI) varied geographically and across future climate scenarios (Figure 4). For example, river reaches in the eastern portion of the basin (below Bistineau, Bayou D'Arbonne, Catahoula and Caddo reservoirs) had consistently low BVI values, river reaches in the central portion of the basin (below Arbuckle, Atoka, Broken Bow and McGee Creek reservoirs) had consistently high BVI values and river reaches in the western portion of the basin (below Buffalo, Greenbelt, Foss and Tom Steed reservoirs) had consistently low or medium BVI values. Overall, BVI scores ranged from 0.004 to 0.109 and indicate model (GCM)- and scenario (RCP)-specific variation. For example, for 22 of 38 river reaches, CCSM4 under RCP 4.5 produced the highest BVI values. Alternatively, for 28 of 38 river reaches, MPI-ESM-LR under RCP 8.5 produced the lowest BVI values.

The conservation feasibility of river reaches varied geographically and across future climate scenarios (Figure 4) For example, river

reaches in the eastern portion of the basin had consistently high feasibility values, river reaches in the central portion of the basin had consistently medium to high feasibility values and river reaches in the western portion of the basin had generally low feasibility. However, there was some variability across climate scenarios. The CCSM4 model under RCP 8.5 produced the lowest feasibility values while the MIROC5 and MPI-ESM-LR models under RCP 2.6 produced the highest feasibility values for most of the western reaches. Generally, there is considerably more uncertainty associated with the feasibility values relative to the BVI.

Because the biodiversity value and conservation feasibility of each site varied independently among future climate scenarios, site-level conservation priority varied among future climate scenarios (Figure 5). For example, the CCSM4 model column showed an increasing separation of river reaches across a west-east gradient under higher RCP's (i.e. most western sites were arranged towards low conservation feasibility and most eastern sites were arranged towards high conservation feasibility). The MIROC5 model column indicated a slight divergence in western reaches towards lower or higher conservation feasibility under higher emission scenarios. The MPI-ESM-LR model column indicated little variation across sites with increasing RCP's. Generally, across all GCMs and RCPs, reaches in arid western portions of the basin shifted the most between quadrants, while a subset of eastern reaches were consistently of high conservation priority as indicated by the line of blue sites on the right side of all plots.

Despite considerable uncertainty about future climate conditions, we found a subset of sites that had both high biodiversity value and high conservation feasibility across all future climate scenarios (Figure 6). Reaches in the east-central portion of the basin (below DeQueen, Broken Bow, Hugo and Sardis reservoirs) were consistently within quadrant I (i.e. they were high conservation priority because they had high biodiversity value and conservation feasibility). However, some reaches in the west-central region sometimes shifted to quadrant II (i.e. high biodiversity value but low conservation feasibility) because the difficulty of meeting environmental flow targets increased under some future climate scenarios. Reaches in the eastern portion of the basin were consistently within quadrant III
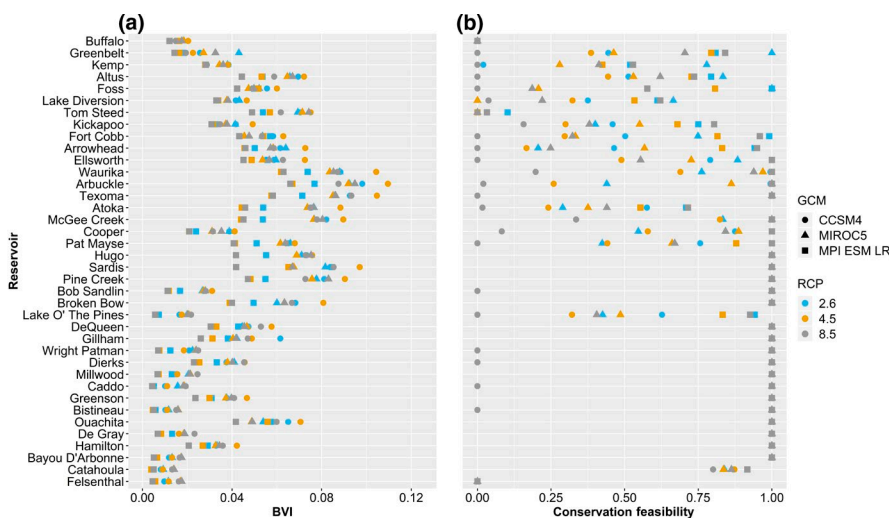

FIGURE 4 Biodiversity value index (BVI) values (a) and conservation feasibility values (b) across all river reaches below 38 major reservoirs in the Red River (United State). Each metric was calculated across nine future climate scenarios using three general circulation models (GCMs) and representative concentration pathways (RCPs). Reservoir names are ordered from West (Buffalo) to East (Felsenthal)

FIGURE 5 Quadrant plots to identify high conservation priority river reaches among river reaches below 38 major reservoirs in the Red River Basin (United States). Biodiversity value index (BVI) values (y-axis) were calculated from species distribution models and various conservation status weights. Conservation feasibility (x-axis) values were calculated from a basin-scale hydrologic model—extracting the highest possible value of societal water satisfaction when environmental flow targets can be fully achieved. Quadrant lines relate to those delineated in Figure 1. River reaches (points) were coloured by longitude
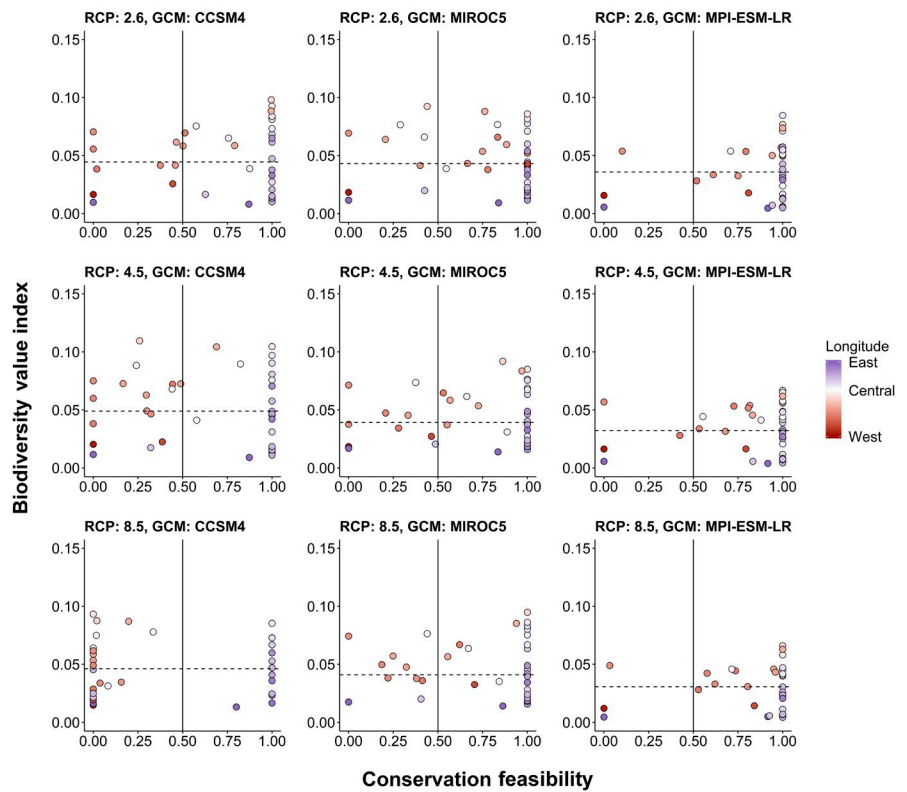

FIGURE 6 Map with stacked bar charts showing the proportion of times each river reach fell within each priority quadrant across all nine future climate scenarios. Colours correspond to the coloured quadrants delineated in the prioritization model in Figure 1 and the inset figure (i.e. HBV/HCF corresponds to the high biodiversity value/high conservation feasibility quadrant in Figure 1)
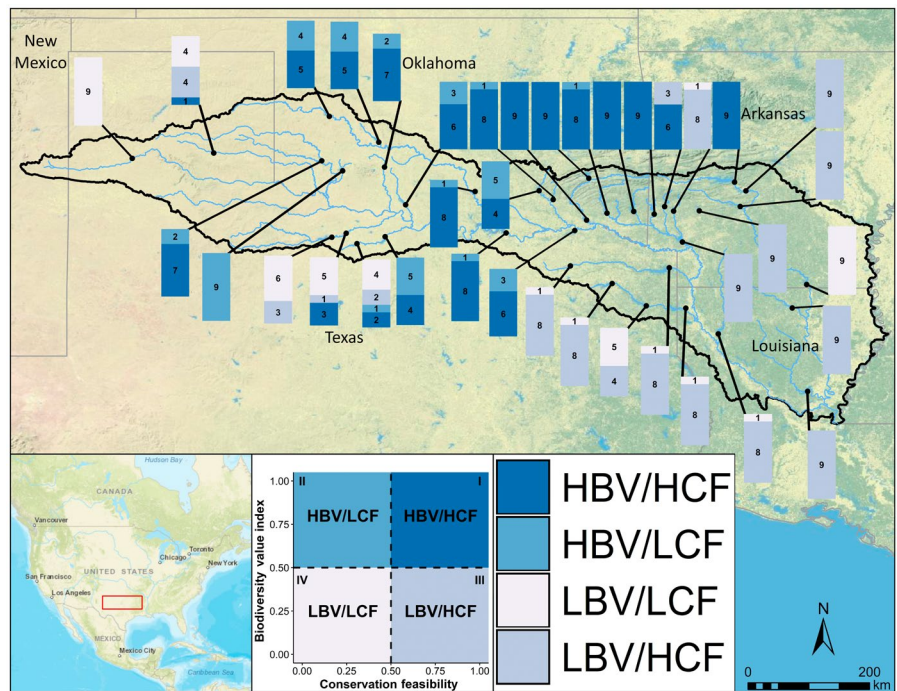
(i.e. high feasibility but low biodiversity value) and sometimes shifted to quadrant IV (i.e. they were low conservation priority because they had both low conservation value and low feasibility). The priority levels of reaches in the western portion of the basin had the highest variability across climate scenarios, to the extent that some individual reaches were placed into each of the four quadrants across the nine climate scenarios.

4 | DISCUSSION

Planning for biodiversity conservation under climatic uncertainty and sociopolitical constraints is challenging yet essential (Carvalho et al., 2011; Guerrero & Wilson, 2017; Lawler & Michalak, 2017). Given that decision-makers must choose among candidate conservation projects under climatic uncertainty and sociopolitical constraints,

our approach provides an example of how to integrate biodiversity, climate and sociopolitical considerations in a prioritization framework. Our work extends other two-dimensional prioritization frameworks (Guerrero & Wilson, 2017; Popejoy et al., 2018) by accounting for uncertainty in future climate conditions. By doing so, our prioritization framework allows decision-makers to identify a subset of candidate sites where biodiversity value and sociopolitical feasibility will be consistently high across future climate scenarios. Thus, our prioritization framework could accelerate implementation by focusing research and assessment resources on those sites where conservation actions are most likely to be valuable and feasible in all future scenarios.

Our case study on environmental flows in the Red River illustrates how our prioritization framework may be used to identify freshwater conservation priorities at the regional scale. We found significant geographic variation in conservation priorities across future climate scenarios basin wide (Figure 6). We found that a subset (5/38) of sites in the east-central portion of the basin could be identified as high priority, low climate risk sites. Using our case study approach, conservation practitioners and resource managers in the region could collaboratively focus attention and resources on these priority sites because they have a high biodiversity value and high likelihood of conservation success while being insensitive to climatic change. Many of these sites are in the Ouachita Mountains, part of the central interior highlands, an ecoregion with a high number of endemic species (Abell et al., 2000; Matthews & Robison, 1998). The Kiamichi River, for example, contains a high number of endemic and threatened fish, crayfish and freshwater mussel species (Cross et al., 1986; Vaughn & Pyron, 1995). Endemic species typically have higher risk of endangerment, and thus higher conservation priority, due to their limited geographic ranges. Ultimately, fish species' future distributions were driven by a combination of static landscape covariates (e.g. lithology and elevation) and dynamic climate-driven covariates that varied among future scenarios (e.g. mean flow and temperature during summer; Gill et al., 2020). The relative importance of covariates in driving species' distributions mirrors current knowledge on important factors driving stream fish distributions (Dodds et al., 2004; Perkin et al., 2015). In some cases, the biodiversity value of a site varied considerably among scenarios because species' occurrences at that site were sensitive to climatic factors (Gill et al., 2020). In other cases, probabilities of occurrence were consistently low across climate scenarios because species' occurrences were constrained by static landscape covariates (e.g. high salinity some western portions of the Red River basin; Winston et al., 1991).

While our prioritization framework identifies some sites as 'low conservation priority', these sites may warrant reconsideration if they align with local conservation priorities despite their low feasibility or broader value (Guerrero & Wilson, 2017). For example, sites in the western portion of the basin (in central Oklahoma and North Texas) sometimes fall within quadrant II (high biodiversity value, low conservation feasibility). Some of these sites could be considered more carefully for prioritization if they are of value to local stakeholders. Additionally, our framework assumes that the potential costs and benefits of conserving a site are independent of

actions taken at other sites. In reality, water conservation actions at upstream sites may increase downstream water availability, potentially altering the conservation feasibility and biodiversity value of downstream sites. Indeed, for many types of conservation actions, the costs and benefits of conservation projects may be contingent on where and when other projects are done (Moilanen et al., 2008; Neeson et al., 2015; Mills et al., 2014; Sundermann et al., 2011). While our framework does not account for project dependency, high priority sites could be more closely examined to determine how modifying dam operations would impact water availability across the reservoir network (Zamani Sabzi, Rezapour, et al., 2019).

Implementing environmental flow programs is a complex issue, especially in drought-prone river basins with increasing water-related conflicts (Arthington et al., 2006; Pittock et al., 2008; Poff, 2018; Poff & Matthews, 2013; Wheeler et al., 2018; Zamani Sabzi, Moreno, et al., 2019). For example, in Oklahoma, there is currently no mandate for water resource managers to incorporate environmental flows into their water management plans, and environmental flows are not considered a 'beneficial use' of water (OWRB, pers. comm.). Texas has an 'Instream Flows Program' for specific river basins, but there is no formal plan to establish flow requirements for the Red River basin. Because of high tensions across political boundaries (see Tarrant Regional Water District vs. Herrmann, 2013), increasing water demand and decreasing water availability, positive-sum solutions may seem improbable. While there are many factors to consider in the design and implementation of environmental flow policies and programs, we believe our findings may aid in identifying solutions to balance environmental flows and societal water needs. For example, the Oklahoma Water Resources Board's Instream Flows Advisory Group recently concluded a pilot study that found that implementing environmental flows in an ecologically and recreationally important sub-basin could be feasible under certain conditions (Oklahoma Water Resources Board, 2019). Other potential environmental flow programs or policies could include non-profit conservation organization programs (i.e. the Nature Conservancy's Sustainable Rivers Program), or the purchasing of water rights to allocate specifically to environmental flows (Connor et al., 2013; Wheeler et al., 2013). Additionally, since water conservation incentives are being established in this region (Oklahoma Water Resources Board, 2015), water governors, water resource managers and fisheries managers could explore where incentivizing water conservation may be most beneficial to both societal and ecosystem needs (Zamani Sabzi, Rezapour, et al., 2019). While there is a complex set of incommensurable factors that comprise the sociopolitical feasibility of increasing environmental flows, our coarse-scale approach is meant to help managers and planners filter among many sites to identify feasible conservation targets.

Our prioritization framework identifies sites that remain valuable and feasible across climate scenarios. Previous work by Popejoy et al. (2018) and Guerrero and Wilson (2017) prioritized sites for conservation by mapping the trade-offs between conservation feasibility and some relevant ecological measure, but without considering future climate uncertainty. Popejoy et al. (2018)

prioritized sites for freshwater mussel conservation in Texas by examining the trade-offs between mussel communities' similarity to the past and conservation feasibility. Guerrero and Wilson (2017) prioritized sites for improving connectivity of native vegetation clusters in Australia by examining the trade-offs between the ecological importance of the vegetation clusters and conservation feasibility in terms of stakeholder presence. Other applications of our framework could include conservation prioritizations of expanding terrestrial protected areas or establishing harvest restrictions in marine protected areas. In these applications, the biodiversity dimension of our framework could be calculated in terms of the species, processes or ecosystem services protected by a conservation action. Similarly, conservation feasibility may be estimated by examining landowners' willingness to sell land or participate in a land conservation program, or stakeholders' willingness to participate in harvest restrictions (Cohen & Foale, 2013; Deacon & Parker, 2009; Knight et al., 2011; McDonald et al., 2018).

Our framework highlights the importance of considering climate uncertainty in spatial conservation prioritizations. There are numerous studies that assess the impacts of climate change on range shifts, range overlap or species loss (Beaumont et al., 2011; Biber et al., 2020; Cheaib et al., 2012; Jones et al., 2013; Midgley et al., 2003). A consistent finding among these studies is that projections vary among models and scenarios, and across species, leaving decision-makers uncertain of the utility of planning approaches (Groves et al., 2012; Kujala et al., 2013; Schmitz et al., 2015). However, prioritization approaches that are inclusive of uncertainty may focus attention on sites that are consistently of high value across future scenarios (Lawler & Michalak, 2017) or penalize locations with high uncertainty (Kujala et al., 2013). Because resources for conservation planning are often limited, it is necessary then to explore approaches that are inclusive of uncertainty and examine the trade-offs among the biodiversity value and conservation feasibility of locations to identify areas that can achieve the largest conservation outcome at minimal risk (Reside et al., 2018).

5 | CONCLUSION

Our study presents a new conservation prioritization framework that enables conservation planners to identify high conservation priority, low climate risk sites. Overall, the novelty of our prioritization approach lies in allowing decision-makers to weigh the trade-offs among sites with varying levels of biodiversity value and conservation feasibility across future climate scenarios. We show that even under considerable climatic uncertainty, it is possible to identify sites that remain high priority across a range of future climate conditions. Our case study in the Red River basin highlights the complex challenges of conservation planning for freshwater biodiversity and water resource management under climatic uncertainty. Our framework could be expanded to a variety of different taxa and systems to identify similar targeted conservation projects.

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