Landscape Ecol
https://doi.org/10.1007/s10980-020-01190-7

RESEARCH ARTICLE

Incorporating social values and wildlife habitats for biodiversity conservation modeling in landscapes of the Great Plains

Amin Rastandeh · Meghann Jarchow · Morgan Carnes

Received: 18 March 2020 / Accepted: 29 December 2020

Abstract
Context Socioecological information should be properly employed in the process of spatial analysis, planning, and management in order to respond to complex and multidirectional biodiversity issues.
Objectives We conducted this study to map socioecological hotspots, where landscapes of social significance and wildlife habitats overlap, show to what extent and how the spatial distribution of social values (SVs) of people toward their landscapes interact with wildlife habitats in socioecological hotspots, simulate the potential for habitat degradation as a result of human activities linked to SVs, identify strategic areas for landscape restoration in socioecological hotspots, where both environmental conditions and SVs support the persistence and colonization of wildlife, and detect specific areas, where SVs of people may be contradictive leading to land-use disputes.
Methods We developed a model to show the potential for habitat degradation based on the spatial distribution of SVs associated with landscapes. We restricted our study to the Upper Missouri River Basin (UMRB) and focused on habitats of five keystone mammal species to assess the validity of our model.
Results Habitat loss, habitat subdivision, habitat dispersion, and habitat shrinkage can be four consequences of human activities for biodiversity in socioecological hotspots of the UMRB, however, the magnitude of impacts varies among landscapes and mammal species.
Conclusion Spatially explicit models to properly map SVs in relation to wildlife habitats are still associated with some uncertainties and limitations, and therefore, require further development. Change in SVs and public attitudes toward land use is essential to avoid further biodiversity loss in this region.

Keywords Socioecological hotspots · Social values · Wildlife habitats · Spatial modeling · Socioecological mapping

A. Rastandeh (✉) · M. Jarchow · M. Carnes
Department of Sustainability & Environment, University of South Dakota, Vermillion, SD, USA
e-mail: amin.rastandeh@usd.edu

Introduction

Landscapes and biodiversity crisis

The Anthropocene era is associated with an accelerating rate of biodiversity loss (Dirzo et al. 2014); with an unprecedented average decline of at least 68% in wildlife populations since 1970 (WWF 2020). The widespread change in natural patterns of landscapes, among other factors (Butchart et al. 2010), has given rise to habitat degradation worldwide (Forman 1995; Pereira et al. 2010). Agriculture (Tscharntke et al. 2012), urbanization (Elmqvist et al. 2013), and industrial development (Ramankutty et al. 2006) have caused widespread changes in the structure of wildlife habitats (e.g. the size, shape, spatial connectivity, etc.). Historically, settler colonialism has also accelerated the process of these changes; more specifically, in North America (Ramankutty and Foley 1999; Goldewijk and Ramankutty 2004), Latin America (Meade 2016), Australia (Bradshaw 2012), and New Zealand (Ewers et al. 2006). Anthropogenic climate change is another powerful catalyst for affecting natural patterns of landscapes and consequently, wildlife habitats, leading to species extinction (Opdam and Wascher 2004; Mantyka-Pringle et al. 2015; Urban 2015; Beever et al. 2017). Up to 60% of the change in the world's land cover is attributed to direct human activities (Song et al. 2018). Cumulatively, these factors have adverse impacts on wildlife habitats, threaten the survival of a wide range of species, and alter their interactions with key components of ecosystems (e.g. soil, water, plant communities, and food sources; Turner 1989; Forman 1995).

Landscapes and social values

Landscape social values (SVs), potential conflicts and synergies among them, are believed to be a growing concern among a diverse range of stakeholders across the world (Lapointe et al. 2019; Brown et al. 2020; Włodarczyk-Marciniak et al. 2020). Methodological plurality can help science and practice to find feasible, site-specific solutions for sustainability issues in landscapes of different socioeconomic background. In this context, cross-disciplinary research on SVs is increasingly becoming an important target to advance environmental sustainability (Kenter et al. 2019). Because of the complex and evolving relationship between humans and nature over time (Riechers et al. 2020), and due to the increasing recognition of the linkages among social and ecological systems in landscapes, there is a growing interest in developing new methods through which social and ecological information can be properly integrated in the process of spatial analysis, planning, and management in order to respond to the current biodiversity and land use challenges (Alessa et al. 2008; Sherrouse et al. 2011; Ban et al. 2013; Bagstad et al. 2016; Guerrero and Wilson 2017; Karimi and Adams 2019; Okamoto et al. 2020; Zhang et al. 2020).

Mapping the SVs people assign to landscapes is of particular importance for the success of science-based conservation implementation (Bryan et al. 2011) and developing an understanding of how and why people value different aspects of ecological systems is deemed to be essential to mitigate potential conflicts among stakeholders (Ives and Kendal 2014). This can inform impact assessment of land-use plans on biodiversity and identify priorities for land-use conflict (Karimi and Hockings 2018). Despite this, standard regional-scale spatially explicit models to accurately link SVs to human activities, landscapes, and biodiversity still requires further development. Therefore, while existing methods such as Public Participation Geographic Information Systems—PPGIS (Sieber 2006), Social Values for Ecosystem Services—SolVES (Sherrouse et al. 2011), Landscape Values Mapping—LVM (Biedenweg et al. 2019), should be expanded, new models should be innovated concurrently to provide an in-depth understanding of societal dimensions of landscapes to reduce the risk of habitat degradation, increase the likelihood of habitat conservation, and identify portions of the land to target for habitat conservation, more specifically in socioecological hotspots, a term that we used in this study to define certain areas, where landscapes of social significance and wildlife habitats overlap. In the context of this study, socioecological hotspots are certain areas of landscapes that are attractive to people for multiple land uses, and simultaneously of ecological importance for landscape restoration and/or biodiversity conservation. This definition does not imply that the human-wildlife confluence does not occur in areas outside of socioecological hotspots. Urbanized landscapes, for example, are believed to be among the most dynamic areas in terms of the human-wildlife confluence (Alvey 2006; Elmqvist et al. 2013; Rastandeh 2018). In this study, however, we used field-collected SV data and then, integrated it with habitat suitability information to address our study objectives across the study region, without specific focus on the nature of landscapes (i.e. urban vs. rural; intact vs. modified, etc.).

Understanding how SVs associated with landscapes are distributed in space is a key step to inform ecosystem services analysis, land-use planning, and strategic landscape management (Sherrouse et al. 2011; Bagstad et al. 2016; Hu et al. 2019). The importance of research on human-wildlife conflict has been articulated in the current literature (Madden 2004; Nyhus 2016; Frank et al. 2019); to date, however, less attention has been paid to the integration of SVs and biodiversity data when land-use issues are contested at the regional scale (Brown et al. 2014). Perhaps one of the main reasons for this negligence, has been the lack of accurate and detailed data, as well as applicable methods to depict social dimensions of the land. While the current literature focuses more on the rate of land-cover change to quantify habitat loss, we conducted this study to show how the spatial distribution of SVs can be employed to simulate the potential for change in the structure of wildlife habitats or shift in the spatial extent of areas suitable for species colonization/persistence; and simultaneously detect opportunities for local- and regional-scale landscape restoration and/or biodiversity conservation. In this respect, we restricted our study to the Upper Missouri River Basin (UMRB; ca. 746,000 Km^2), as part of the Great Plains of the United States, mainly covered by Montana, North Dakota, South Dakota, and Wyoming, to fulfil our study objectives: (1) to map socioecological hotspots (2) to show to what extent and how the spatial distribution of SVs of people toward their landscapes interact with wildlife habitats in mapped socioecological hotspots, (3) to simulate the potential for habitat degradation as a result of human activities linked to SVs, (4) to identify strategic areas for landscape restoration in socioecological hotspots, where both environmental conditions and SVs support the persistence and colonization of wildlife, and (5) to detect specific areas, where SVs of people may be contradictive leading to land-use disputes.

Methodology

We developed a model to show the potential for habitat degradation based on the spatial distribution of SVs associated with landscapes. The central assumption was that SVs are predictive of land-use conflict (Brown et al. 2020), may trigger a broad spectrum of human activities, ranging from development to conservation, and determine how the land should be controlled and used in the future. We did not use this model to measure the risk or rate of land-cover change but to envision how the structure of wildlife habitats may be altered by SVs in socioecological hotspots of the region through various mechanisms in ways that reduce and/or change the spatial extent of suitable areas for species colonization/persistence.

Study region

Euro-American settlement began at the second half of the nineteenth century (Lampard 1955) in the United States, including the UMRB (Fig. 1). In comparison with other parts of the United States, population density in this region is still very low (Stoy et al. 2018). There are only 23 human population centers with a population of more than 10,000 people across the region (US Census Bureau 2018). Agricultural landscapes of the region are critically important to produce human food and livestock feed (Jarchow et al. 2020). For example, respectively 30%, 13%, 11%, and 9% of wheat, soybean, cattle, and corn productions of the United States are supplied in this region (Stoy et al. 2018). Commonplace human activities including settlement development, farming, ranching, grazing, hunting and mining have substantially affected wildlife species, leading to widespread habitat degradation and biodiversity loss over the last two centuries across the region (Samson and Knopf 1994; Jarchow et al. 2020).

Data preparation

Land cover

We used the dataset North American Land Cover (spatial resolution: 30 m; Commission for Environmental Cooperation 2020) to measure the current composition of land cover classes across the UMRB.

Wildlife habitats

We used the dataset Gap Analysis Project Species Habitat Maps (GAP; U.S. Geological Survey 2018) to illustrate the habitat distribution of mammal species in the UMRB. Its developers recommend the use of this dataset for national, regional or statewide biodiversity and habitat conservation planning. While it provides very useful spatial information about parts of the land that contain suitable conditions for vertebrates to occur, fine-scale features have not been considered in GAP. This dataset has been recognized as a reliable source of information for regional-scale biodiversity

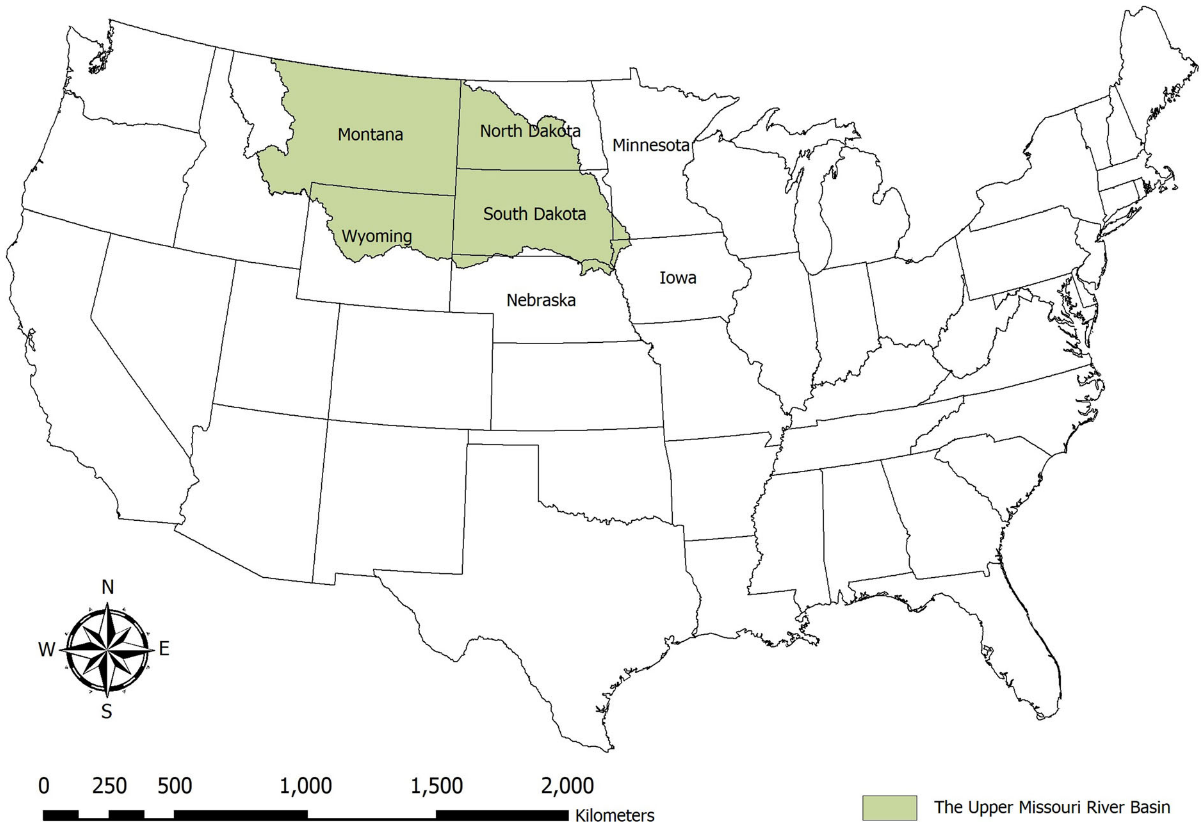

Fig. 1 The Upper Missouri River Basin

research across the United States (Gergely et al. 2019; Tarr 2019; Dietz et al. 2020). To build this dataset, species habitat suitability has been evaluated not only based on existing land cover classes, but also other determinant climatic and environmental parameters. (U.S. Geological Survey 2018). We selected five mammal species from GAP in order to validate the accuracy and capability of our model (Table 1). We considered four basic factors to sample the most relevant species for this study: (1) habitat requirements, (2) species dispersal patterns, (3) ecological mutualism relationships, and (4) cultural importance. Globally, mammal species increase habitat heterogeneity and significantly contribute to biodiversity at various spatial scales (Lacher et al. 2019). The five species we selected, in particular, can play significant roles in supporting vital ecosystem services in the region and for this reason, they are identified as keystone species in the literature. These keystone species also function as a representative set of habitat requirements for other taxa, can affect ecosystems (Mills et al. 1993), and are of critical importance to the structure of ecosystems (Collier et al. 1997). Some, if not all, of these species can be concurrently considered as flagship and/or umbrella species (Walpole and Leader-Williams 2002; Wilkins et al. 2019), as they are well known among peoples of the UMRB, and are species of cultural importance, as well (cf. Garibaldi and Turner 2004). For each species, we extracted areas that can act as its primary habitat using GAP (U.S. Geological Survey 2018). We termed these areas as "wildlife habitats". Although the chosen species may be present or absent in their habitats at the present time, they have suitable conditions under which species can persist and/or colonize.

Social values

During 2018 and 2019, we conducted a survey of 1009 residents from 22 cities across the UMRB. The aim was to identify where and how the land is valued by people in this region. As one part of this survey, we

Table 1 Mammals sampled for this study and their ecological roles in ecosystems of the UMRB based on (Naiman 1988; Knapp et al. 1999; Kotliar et al. 1999; Miller et al. 2000; Gökbulak 2002; Goheen and Swihart 2003; Kotliar et al. 2006; Rosas et al. 2008; Allred et al. 2011; Hoogland 2013; Nummi and Holopainen 2014; Hale and Koprowski 2018)

Common name	Scientific name	Ecological roles
American beaver	*Castor canadensis*	Water purification
		Erosion control
		Habitat provision for some species
American red squirrel	*Tamiasciurus hudsonicus*	Seed dispersal
		Soil fertilization
Bison	*Bison bison*	Seed dispersal
		Soil bioturbation
		Habitat provision for some species
Black-tailed prairie dog	*Cynomys ludovicianus*	Soil bioturbation
		Seed germination
		Habitat provision for some species
		Food provision for some predators
White-tailed prairie dog	*Cynomys leucurus*	Soil bioturbation
		Seed germination
		Habitat provision for some species
		Food provision for some predators

asked each participant to choose up to four SVs from a list of 11 major SVs (i.e. Aesthetics, Agriculture, Community, Conservation, Cultural, Development, Economy, Energy, Recreation, Spiritual, and Tourism; Jarchow et al. 2018; Carnes 2019) and indicate locations where his/her chosen values would like to occur on the land, within a radius of 80 km from the place of his/her residence. We defined these SVs with respect to commonplace human activities in the region, as described above (cf. Jarchow et al. 2018, 2020), and with particular attention to the results of 33 semi-structured interviews with stakeholders about SVs and landscapes of the UMRB (Carnes 2019). We built a dataset of 5046 georeferenced locations (GRLs), representing landscapes of social significance in the region. Each GRL corresponds to one specific SV and thereby, its corresponding human activities. In this study, we used an impact-sensitive classification of SVs based on previous research in the region (Jarchow et al. 2018; Stoy et al. 2018; Carnes 2019; Jarchow et al. 2020) and condensed them into five groups (Table 2). Landscapes of the UMRB are subject to change depending on the nature of these five groups of SVs. Although the impacts of these groups differ, in all cases, except for Group 5, human activities can adversely affect biodiversity through different mechanisms, including environmental (e.g. air, water, soil, noise, and light) pollutions, land-cover change, habitat fragmentation (e.g. building and road construction), and human-induced disturbances (e.g. fires, pest, and weed dispersal; WWF 2018, 2020). Therefore, even those SVs that seem neutral to biodiversity (e.g. Group 3; Tradition), can motivate a range of human activities that lead to disturbance in wildlife habitats (e.g. the spread of pest animals, weed plant species, fires, and environmental pollutions).

We applied a radius of 5 km from the center of each GRL, termed the Radius of Human Activity (RHA; 7850 ha). We created 5046 RHAs corresponding to 5046 GRLs. We considered each RHA as a specific area within which a specific range of human activities will be dominant. Human activities often have far-reaching impacts on abiotic and biotic resources and their impacts are not limited within the locations, where people are physically present. Any change in these resources (e.g. soil degradation, water pollution, land cover conversion) can be detrimental to biodiversity (WWF 2020). According to Benítez-López et al. 2010, Barbosa et al. 2020, and Mendes et al. 2020, under normal conditions, the environmental impacts of human activities on mammals can spread up to 5 km, if not more, from the origin of impacts. We considered each GRL as an origin of the impacts of a certain SV, linked to a specific range of human activities (Table 2), on wildlife habitats and assumed its corresponding RHA as the spatial extent of its possible impacts. This method enabled us to transform

Table 2 An impact-sensitive classification of SVs applied in this study based on the nature of human activities in the UMRB (cf. Stoy et al. 2018; Carnes 2019; Jarchow et al. 2020)

Human activities	Main SVs associated with landscapes	# of GRLs
Group 1: Development	Economic investment	1110
	Residential and commercial development	
	Employment	
	Energy extraction	
Group 2: Recreation	Outdoor recreation	1289
	Tourism in historical or cultural sites	
Group 3: Tradition	Aesthetics	943
	Identity and family traditions	
	Sacred places and religion	
Group 4: Agriculture	Farming	1009
	Ranching	
Group 5: Conservation	Responsibility to land	695

GRLs to measurable surfaces based on the spatial distribution of each group of SVs and to detect areas that are subject to human activities associated with five groups of SVs.

Overlay analysis

We quantified the spatial overlap between landscapes of social significance and wildlife habitats using ArcMap 10.5. We mapped the overlap between RHAs and wildlife habitats to identify socioecological hotspots, where landscapes of social significance and wildlife habitats overlap (Fig. 2). We preliminarily used four landscape metrics (McGarigal 2015) and the software program FRAGSTATS 4.2 (McGarigal et al. 2012) to quantify the structure of wildlife habitats overlapping with landscapes of social significance (Table 3). We also used these landscape metrics to characterize areas subject to various human activities, as well as the overlap between Conservation and other SVs in socioecological hotspots. The number of patches (PN) can be increased as a result of landscape fragmentation (Botequilha Leitao and Ahern 2002; McGarigal et al. 2012). An increase in PN can reduce the risk of disturbances (e.g. wildfire, flood, hurricane, disease outbreak; Botequilha Leitao et al. 2006). At the same time, landscape fragmentation limits species movement (Forman 1995; Dramstad et al. 1996; Botequilha Leitao et al. 2006). Patch size is the most important component of landscape pattern affecting biomass, nutrient storage, productivity, and species composition and diversity (McGarigal 2015). Total Area (AREA), Largest Patch (LP), and Mean Patch Size (MPS) can quantify habitat availability in terms of patch size. These landscape metrics are species-specific and scale-dependent; therefore, their values should be interpreted with respect to the spatial ecology of species under study.

Landscape patterns and wildlife habitats

We simulated the impact of human activities on the current structure of wildlife habitats in the GIS environment through mapping and modeling the spatial distribution of the overlap between socially-valued human activities and wildlife habitats. We did not specifically calculate the nature and power of human impacts on wildlife habitats because human land use activities differ in terms of the potential effects that they have on the wildlife habitats they overlap. Instead, we developed and used a regional-scale spatially-explicit model to depict the spatial distribution of human activities in relation to wildlife habitats. We made three basic assumptions to run this simulation: (1) although different in terms of the nature and magnitude of impacts on wildlife habitats, human activities linked to SVs in Groups 1, 2, 3, and 4 (i.e. Development, Recreation, Tradition, and Agriculture) have the potential to increase the risk of wildlife habitat degradation in different ways; (2) human activities linked to SVs in Group 5 (i.e. Conservation) will not increase the extent of wildlife habitats, mainly because environmental requirements to provide suitable conditions for species are not necessarily met in areas that people value the land for conservation practices; however, where wildlife

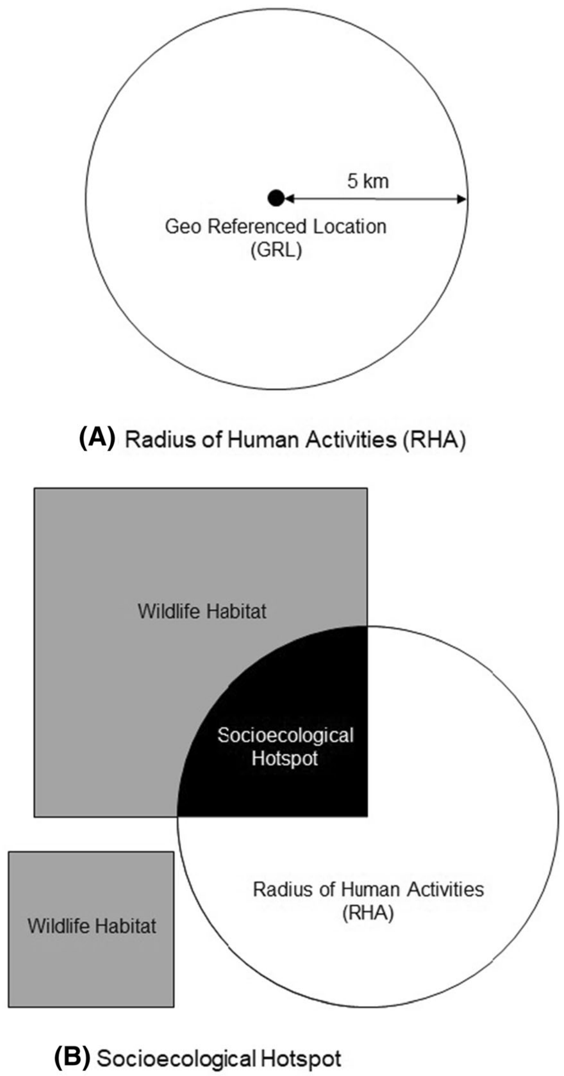

Fig. 2 The spatial concepts used in this study; **a** Radius of Human Activity (RHA) created based on a 5 km buffer from the position of each georeferenced location (GRL), representing a portion of the land subject to the impacts of human activities (i.e. landscapes of social significance); **b** a socioecological hotspot (black), where human activities and wildlife habitats overlap

habitats and RHAs linked to Conservation overlap, the chance of dedicating the land for landscape restoration and/or biodiversity conservation is high because both environmental and social factors support such practices; (3) where RHAs linked to Conservation overlap with other human activities, land-use disputes are expected.

We used six class-level landscape metrics in FRAGSTATS 4.2 (McGarigal et al. 2012) to quantify

Table 3 Landscape metrics used to quantify the structure of wildlife habitats overlapping with landscapes of social significance

Metrics	Unit	Abbreviation/range
Total area	Hectares	AREA ≥ 0
Number of patches	None	PN ≥ 0
Largest patch	Hectares	LP > 0
Mean patch size	Hectares	MPS > 0

the structure of wildlife habitats across the UMRB before and after applying the potential impact of SVs on wildlife habitats in socioecological hotspots (Table 4). To prepare accurate inputs for FRAGSTATS, we rasterized the maps generated in the process of overlay analysis using a cell size of 50 m. we applied the eight-cell-neighborhood rule to run our model in FRAGSTATS. We computed four characteristics of landscape transformation: (1) habitat loss, (2) habitat dispersion, (3) habitat subdivision, and (4) habitat shrinkage (Fig. 3). We used Class Area Proportion (CAP) to quantify the total area of wildlife habitats and habitat availability across the region; Area-weighted Mean Radius of Gyration (GYRATE_AM) to compute habitat shrinkage; Aggregation Index (AI) and Patch Cohesion Index (COHESION) to quantify habitat dispersion against habitat aggregation; and Patch Density (PD) and Effective Mesh Size (MESH) to quantify to what extent wildlife habitats may be subject to subdivision.

The chosen landscape metrics are mainly size-independent, highly recommended for in-depth landscape analysis, and central to the study of habitat loss and landscape fragmentation (Botequilha-Leitao et al. 2006; Cushman et al. 2008; Kupfer 2012; McGarigal et al. 2012). Detailed descriptions of landscape metrics and their applications to landscape quantification are provided in (McGarigal 2015).

Results

Land cover

Of 19 land cover classes defined in the dataset North American Land Cover (Commission for Environmental Cooperation 2020), 11 classes occur in the UMRB

Table 4 Landscape metrics used to quantify the potential impacts of SVs on the structure of wildlife habitats in the UMRB before and after applying the impacts of human activities on wildlife habitats

Metrics	Unit	Abbreviation/range
Class area proportion	Percent	$0 \leq$ CAP ≤ 100
Patch density	Number per 100 ha	PD > 0
Effective mesh size	Hectares	Ratio of cell size to area $\leq$ MESH $\leq$ total area
Aggregation index	Present	$0 \leq$ AI ≤ 100
Patch cohesion index	None	$0 <$ COHESION < 100
Area-weighted mean radius of gyration	Meters	GYRATE_AM > 0

Area-weighted Mean Radius of Gyration (GYRATE_AM) is equivalent to Correlation Length (CL). More information is available in McGarigal 2015

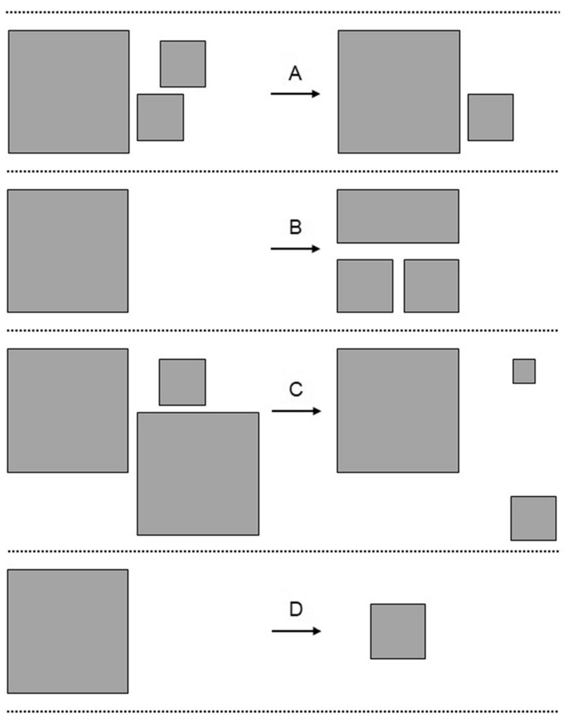

Fig. 3 Four basic concepts used to characterize landscape transformation and habitat degradation in this study; **a** Habitat Loss; **b** Habitat Subdivision; **c** Habitat Dispersion; **d** Habitat Shrinkage. Note that these phenomena are not standalone in practice. For example, in C, Habitat Dispersion is associated with Habitat Shrinkage

Table 5 Land cover classes in the UMRB measured based on the dataset North American Land Cover (Commission for Environmental Cooperation 2020)

Class	% of the UMRB
Grassland	47
Cropland	25
Shrubland	14
Needleleaf forest	8
Urban and built-up	2
Water	< 2
Wetland	< 2
Barren	< 1
Broadleaf deciduous forest	< 1
Mixed forest	< 0.5
Snow and ice	< 0.1

(Table 5). Grassland, Cropland, Shrubland, and Needleleaf Forest are respectively the most dominant land cover classes in the region. Areas covered by the class Urban and Built-up also encompass some 2% of the land. Other strategic land cover classes, including Wetland, Water, Broadleaf Deciduous Forest, and Mixed Forest, cover only a small fraction of the region.

Wildlife habitats

Currently, areas with suitable conditions for the black-tailed prairie dog occupy at least 15.41% (± 2%) of the UMRB. For other mammals, the coverage is under 10% of the region (Table 6). A very small area of the region can function as primary habitat for the bison. Habitat requirements for the American beaver are met in less than 4% of the region. Habitats of this species are extremely scattered. Habitats of the American red squirrel and white-tailed prairie dog occupy 8.51% and 6.33% of the UMRB, respectively. In certain areas

Table 6 The extent of suitable areas as wildlife habitat for mammals in the UMRB

Species	% of the UMRB
American beaver	3.48
American red squirrel	8.51
Bison	0.56
Black-tailed prairie dog	15.41
White-tailed prairie dog	6.33

of the UMRB, wildlife habitats converge and overlap, and therefore, these mammals can potentially coexist.

Landscapes of social significance

A large portion of the UMRB was covered by the RHAs (Table 7). This geographical extent represents landscapes of social significance that are valued for Agriculture (% of the UMRB: 5.36%), Recreation (4.12%), Conservation (3.71%), Tradition (3.38%), and Development (3.34%).

Socioecological hotspots

Landscapes of social significance overlie a large portion of wildlife habitats (ca. 3,474,870 ha; 4.65% of the UMRB). As defined, we regarded this part of the UMRB as socioecological hotspots. The values of four landscape metrics for wildlife habitats overlapping with human activities in socio-ecological hotspots are provided in Table 8. The overlap between wildlife habitats and human activities in socioecological hotspots ranges from 14.11 to 2.79% (Fig. 4). This chart provides quantitative information about the risk of encroachment to wildlife habitats imposed by each group of human activities in the UMRB. These results combined with information depicted in Figs. 5 and 6 suggest that wildlife habitats in socio-ecological hotspots of the UMRB can be widely influenced by multifarious human activities; however, the magnitude of impacts on species differs depending on the nature of human activities. This information should be interpreted with particular attention to the spatial ecology of wildlife species, as well as the nature of human activities. For example, as depicted in Table 8; Fig. 5, the American beaver has a high number of habitat patches overlaid by human activities (PN_{AB}: 565,243). The opposite, the values of this landscape metric for habitats of the bison and white-tailed prairie dog are very low (PN_{B}: 29,413, PN_{WTPD}: 52,884). Habitats of the black-tailed prairie dog are highly fragmented, specifically in areas where SVs for Agriculture are dominant (PN_{BTPD}: 215,969). In Fig. 6, the difference between the values of MPS for the white-tailed prairie dog and other species is considerable, specifically in areas where habitats of this species have overlap with human activities linked to Development and Agriculture. These findings are species-specific and have different implications for the conservation and/or restoration of these species and their habitats depending on the objectives of projects.

Table 7 Landscapes of social significance measured by RHAs for five groups of SVs in the UMRB

Core activity	RHA (ha)	Percent	% of the UMRB
Development	2,496,801	16.79	3.34
Recreation	3,075,988	20.68	4.12
Tradition	2,527,261	16.99	3.38
Agriculture	3,999,214	26.89	5.36
Conservation	2,772,032	18.65	3.71
Total*	14,871,296*	100	19.91

*This area includes overlaps between RHAs, not the total size of landscapes of social significance

Landscape pattern analysis

Habitat degradation within socioecological hotspots could have far-reaching consequences for the composition and configuration of wildlife habitats, and consequently biodiversity across the UMRB (Figs. 7, 8, and 9). We found that habitat loss, habitat subdivision, habitat dispersion, and habitat shrinkage are four consequences of human activities for wildlife habitats in the UMRB; however, the magnitude and spatial extent of these impacts vary among species (Table 9).

The amount of habitat loss ranges from 17.85% in habitats of the bison to 7.89% in habitats of the white-tailed prairie dog (Table 10). We observed the highest changes in the values of CAP and PD in habitats of the bison. Habitats of the black-tailed prairie dog had the greatest degree of shrinkage ($GYRATE\_AM_{BTPD}$: − 30.52%) and subdivision ($MESH_{BTPD}$: − 46.39%).

Table 8 The values of four landscape metrics (cf. Table 3) to quantify the structure of wildlife habitats overlaid by human activities in socio-ecological hotspots

Species—SVs	AREA (ha)	PN	LP (ha)	MPS (ha)
AB				
Development	93,770.83	94,757	3932.41	0.98
Recreation	145,330.57	123,849	7235.81	1.17
Tradition	112,737.66	94,239	2845.92	1.19
Agriculture	150,719.45	144,988	4552.11	1.03
Conservation	115,868.73	107,410	6646.93	1.07
Total	618,427.24	565,243		
ARS				
Development	183,175.54	43,237	28,803.69	4.23
Recreation	633,388.11	73,567	109,806.12	8.61
Tradition	464,537.46	61,626	57,480.45	7.54
Agriculture	188,019.65	64,922	43,521.02	2.89
Conservation	572,691.95	72,148	211,651.69	7.93
Total	2,041,812.71	315,500		
B				
Development	19,194.78	3418	3519.37	5.61
Recreation	15,012.07	2755	2860.08	5.44
Tradition	58,615.92	10,992	3053.21	5.33
Agriculture	25,131.41	2879	5407.86	8.72
Conservation	44,829.45	9369	6551.83	4.78
Total	162,783.63	29,413		
BTPD				
Development	546,251.18	154,464	27,395.32	3.53
Recreation	520,866.03	156,223	23,999.46	3.34
Tradition	480,917.05	140,680	22,940.67	3.42
Agriculture	902,352.42	215,969	81,130.52	4.17
Conservation	547,618.08	140,874	41,893.81	3.89
Total	2,998,004.76	808,210		
WTPD				
Development	157,762.21	6263	43,175.26	25.18
Recreation	180,184.71	17,510	22,188.58	10.29
Tradition	134,943.42	9973	14,716.78	13.53
Agriculture	132,137.59	5941	35,546.86	22.24
Conservation	162,257.46	13,197	50,081.57	12.29
Total	767,285.39	52,884		

Mammals have been coded as follows: American Beaver (AB), American Red Squirrel (ARS), Bison (B), Black-tailed Prairie Dog (BTPD), White-tailed Prairie Dog (WTPD)

We also observed the highest level of dispersion in habitats of the American beaver (AI_{AB}: − 0.74%; $COHESION_{AB}$: − 0.73%), which can extremely limit species movement. We found that except for habitats of the white-tailed prairie dog, human activities increase habitat subdivision in all wildlife habitats. This change can restrict the movement of mammals in the UMRB while may simultaneously act as a means to protect them against natural and human-induced disturbances. Although habitat loss was modeled to occur for the white-tailed prairie dog (CAP_{WTPD}: − 7.89%), we found no further dispersion in habitats of this species ($COHESION_{WTPD}$: 0).

Conflict of interests among SVs

Parts of socioecological hotspots are potentially subject to conflict of interests among SVs. Our model

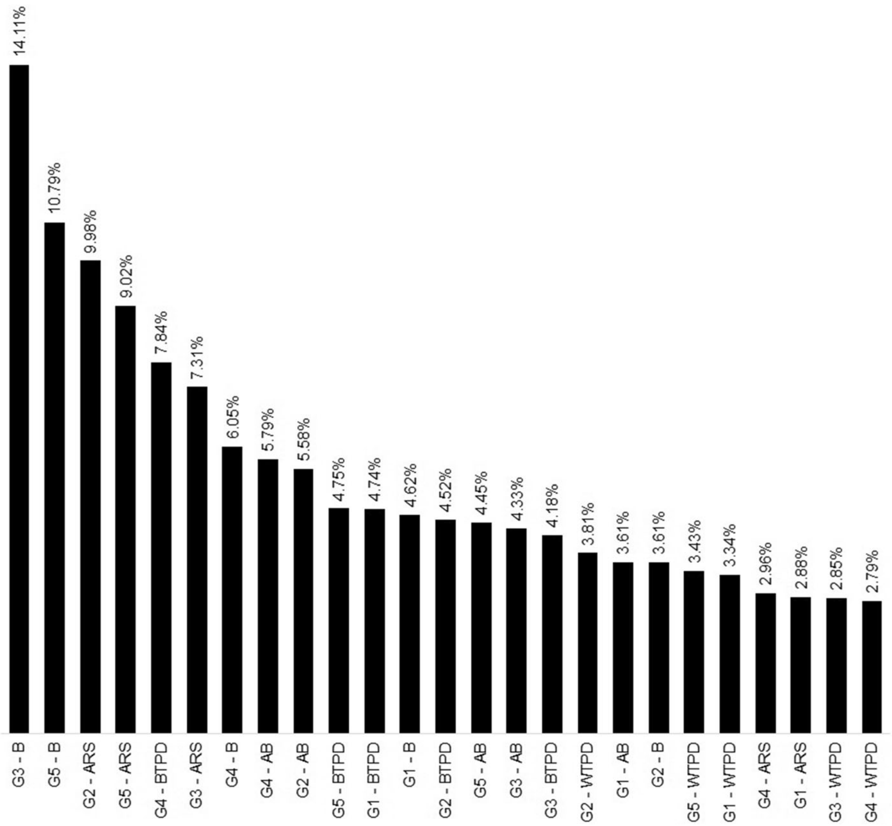

Fig. 4 The percentage of overlap between wildlife habitats and human activities in socioecological hotspots. Human activities linked to SVs have been coded as follows: Development (G1), Recreation (G2), Tradition (G3), Agriculture (G4), and Conservation (G5). Mammals have been coded as follows: American Beaver (AB), American Red Squirrel (ARS), Bison (B), Black-tailed Prairie Dog (BTPD), and White-tailed Prairie Dog (WTPD)

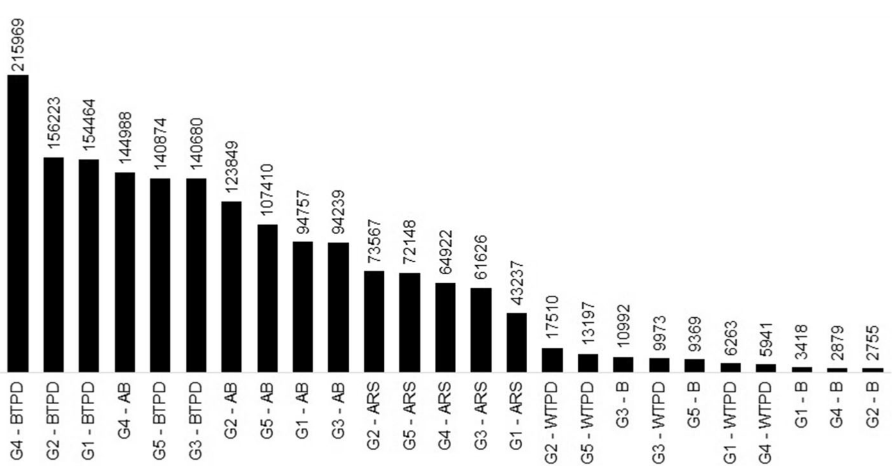

Fig. 5 The number of habitat patches overlaid by human activities in socioecological hotspots. Human activities linked to SVs have been coded as follows: Development (G1), Recreation (G2), Tradition (G3), Agriculture (G4), and Conservation (G5). Mammals have been coded as follows: American Beaver (AB), American Red Squirrel (ARS), Bison (B), Black-tailed Prairie Dog (BTPD), and White-tailed Prairie Dog (WTPD)

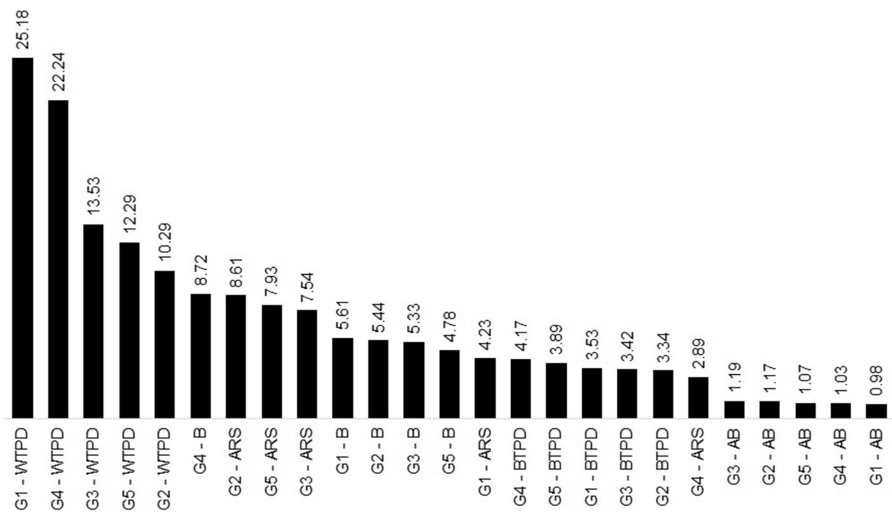

Fig. 6 The mean patch size (MPS; ha) of habitat patches overlaid by human activities in socioecological hotspots. Human activities linked to SVs have been coded as follows: Development (G1), Recreation (G2), Tradition (G3), Agriculture (G4), and Conservation (G5). Mammals have been coded as follows: American Beaver (AB), American Red Squirrel (ARS), Bison (B), Black-tailed Prairie Dog (BTPD), and White-tailed Prairie Dog (WTPD)

revealed that in about one-third of areas identified as socioecological hotspots, SVs and their corresponding human activities are contradictive, more specifically where SVs related to Conservation and other human activities (i.e. Development, Recreation, Tradition, and Agriculture; Table 2) overlap (Fig. 8). Of the total area people value for Conservation (i.e. 3.71% of the UMRB; Table 7), 49.13% is in socioecological hotspots, meaning that about half of the land has an overlap with wildlife habitats, a potential social support for landscape restoration and/or biodiversity conservation; however, about three-fourth of this overlap (985,493 ha, 72.35%) was concurrently valued by people for other SVs (Fig. 8). These values quantify areas that are subject to conflict of interests between Conservation (G5) and other SVs (G1 to 4). In particular, the values of MPS for G1 to 4 (2.25 ha), G5 (2.63 ha), and Overlap (2.42 ha) show the potential for conflict of interests among human activities linked to SVs. This information is central to study habitat size for species, as well as the availability and abundance of habitats in landscapes of the UMRB.

Discussion

A new spatially explicit model

In this study, we developed a region-wide spatially explicit model to identify socioecological hotspots across an extensive mosaic of the Great Plains, widely influenced by different human activities. We used the model to envision the potential for habitat degradation resulting from human activities linked to SVs of people. We also provided a robust spatially explicit basis to identify the potential for conflict of interests in terms of SVs of people (i.e. Conservation and other SVs). The information that we obtained from this spatial modeling provides an insight into the spatial distribution of SVs, the extent of socioecological hotspots, potentials for habitat degradation in the future, opportunities for landscape restoration and/or biodiversity conservation, and the risk of conflict of interests among SVs in certain parts of the region. It establishes a robust platform for making inclusive decisions on land-use and resource management based on an integrated dataset of social and ecological data. For example, the results of this study indicate that while about half of the UMRB is covered by grasslands, a very small fraction of the land can function as wildlife habitat for grassland mammals (i.e. the bison, black- and white-tailed prairie dog; Table 6) and a

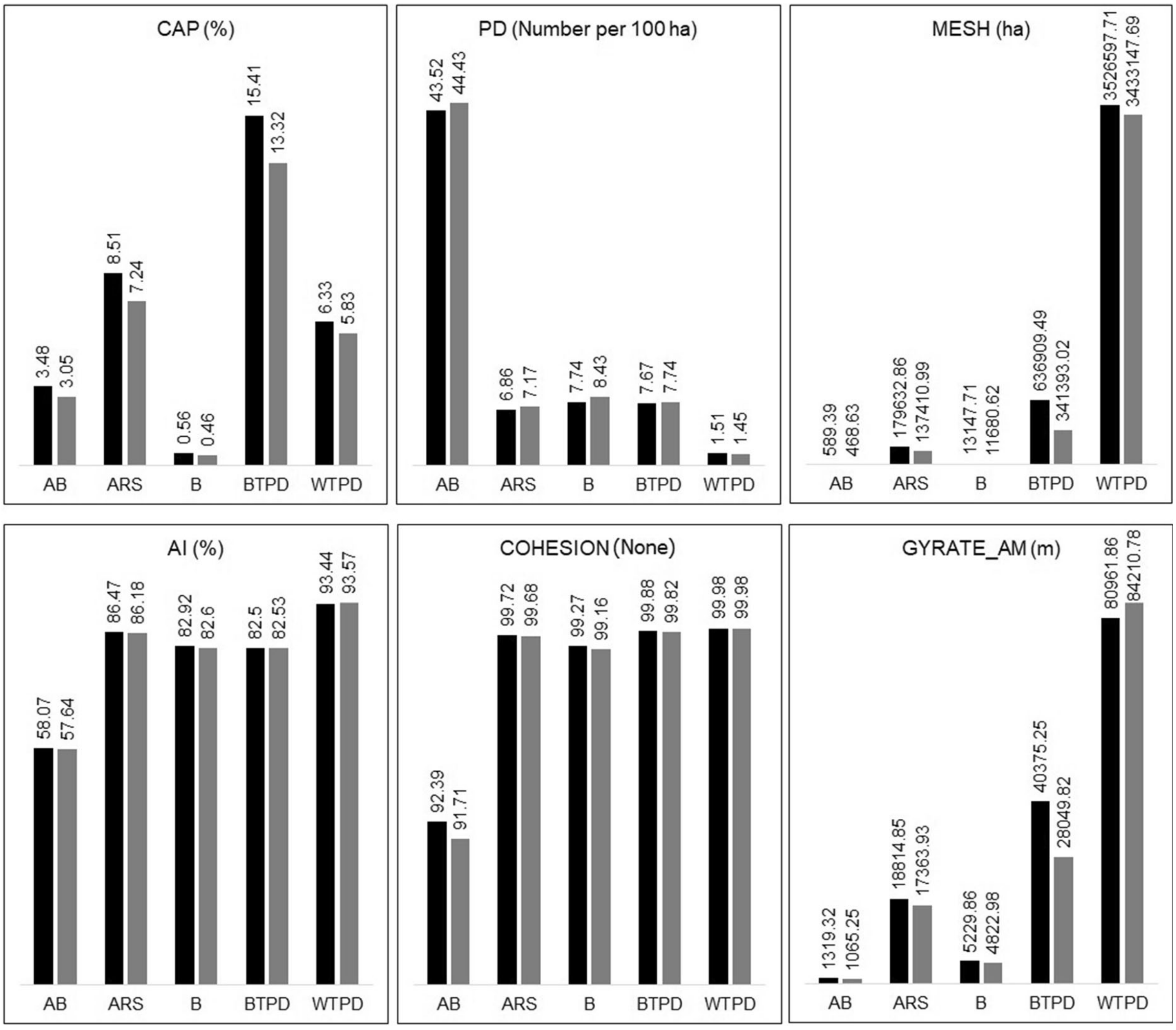

Fig. 7 The values of Class Area Proportion (CAP), Patch Density (PD), Effective Mesh Size (MESH), Aggregation Index (AI), Patch Cohesion Index (COHESION), and Area-weighted Radius of Gyration (GYRATE_AM) for habitats of American Beaver (AB), American Red Squirrel (ARS), Bison (B), Black-tailed Prairie Dog (BTPD), and White-tailed Prairie Dog (WTPD) across the UMRB in the current (black) and simulated (grey) situations. The charts show how human activities (i.e. Development, Recreation, Tradition, and Agriculture) in socioecological hotspots can change the values of landscape metrics for wildlife habitats in the whole region

relatively large part of this fraction is subject to the impacts of human activities (Table 8). Utilizing landscape metrics in FRAGSTATS enabled us to quantify the structure of wildlife habitats before and after applying the impacts of SVs on wildlife habitats. As previously discussed by others (Forman 1995; Botequilha-Leitao and Ahern 2002; McGarigal et al. 2012), the values of landscape metrics should be interpreted based on the context and phenomenon under study. Although we provide examples of possible interpretations here to explain how this model can contribute to a better understanding of the spatial relationship between SVs and biodiversity, we strongly believe that these interpretations remain species- and site-specific, and therefore, should not be generalized to the whole.

We observed that landscape metrics accurately quantified habitat loss, subdivision, dispersion, and shrinkage for habitats of five keystone species. As such, the model functioned as a tool to evaluate how

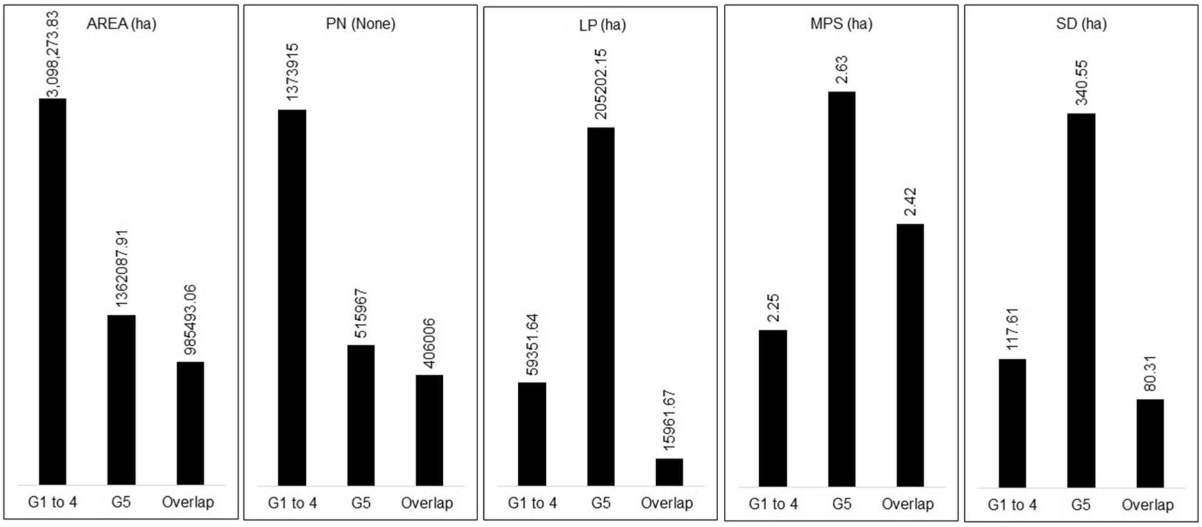

Fig. 8 The values of Total Area (AREA), Patch Number (PN), Largest Patch (LP), Mean Patch Size (MPS) and Standard Deviation (SD) for areas subject to Development (G1), Recreation (G2), Tradition (G3), Agriculture (G4), and Conservation (G5) in socioecological hotspots. The overlap between areas people value for Development, Recreation, Tradition, and Agriculture (G1 to 4) and parts of the land that are valued for Conservation (G5) is characterized by AREA, PN, LP, MPS, and SD

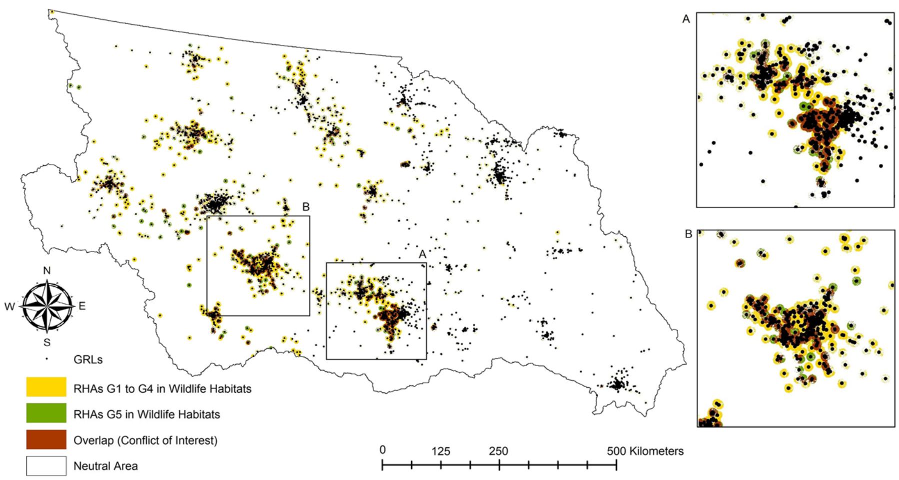

Fig. 9 The spatial distribution of overlaps between human activities (Radius of Human Activities; RHAs) and wildlife habitats in socioecological hotspots. Areas overlaid by human activities related to Development, Recreation, Tradition, and Agriculture (yellow) may negatively affect the structure of wildlife habitats. Areas overlaid by human activities related to Conservation (green) may have the social support for landscape restoration in wildlife habitats. Where human activities related to Development, Recreation, Tradition, and Agriculture overlap with human activities related to Conservation (red), the potential for conflict of interests for land-use is higher. Neutral Areas (white) are expanses of the land that are not considered as socioecological hotspots. There is no overlap between human activities and wildlife habitats in this area. Georeferenced locations (GRLs; black) represent areas where people value the land based on a range of human activities classified in Table 2. Note that in each RHA, not the whole area overlaps with expanses of wildlife habitats (cf. Fig. 10)

SVs may influence biodiversity. Human perceptions of good governance and social impacts are among strong predictors of increasing public support for the longevity of conservation practices (Bennett et al. 2019), human perceptions about the impacts of conservation on the livelihood can determine how local communities decide on land-uses (Bennett and Dearden 2014), and education and community co-

Table 9 The difference between the values of landscape metrics Class Area Proportion (CAP), Patch Density (PD), Effective Mesh Size (MESH), Aggregation Index (AI), Patch Cohesion Index (COHESION), and Area-weighted Radius of Gyration (GYRATE_AM)) before and after applying the impacts of human activities linked to SVs on wildlife habitats

Species	Difference between the current and simulated landscapes; Increase (+), Decrease (−)					
	CAP	PD	MESH	AI	COHESHION	GYRATE_AM
AB	− 0.43	+ 0.91	− 120.76	− 0.43	− 0.68	− 254.07
ARS	− 1.27	+ 0.31	− 42,221.87	− 0.29	− 0.04	− 1450.92
B	− 0.10	+ 0.69	− 1467.09	− 0.32	− 0.11	− 406.88
BTPD	− 2.09	+ 0.07	− 295,516.47	+ 0.03	− 0.06	− 12,325.4
WTPD	− 0.50	− 0.06	− 93,450.02	+ 0.13	0	+ 3248.92

Mammals have been coded as follows: American Beaver (AB), American Red Squirrel (ARS), Bison (B), Black-tailed Prairie Dog (BTPD), White-tailed Prairie Dog (WTPD)

Table 10 The percentage of change in the values of landscape metrics after applying the impacts of human activities linked to SVs on wildlife habitats

Species	% of change; Increase (+), Decrease (−)					
	CAP	PD	MESH	AI	COHESHION	GYRATE_AM
AB	− 12.35	+ 2.09	− 20.48	− 0.74	− 0.73	− 19.25
ARS	− 14.92	+ 4.51	− 23.51	− 0.33	− 0.04	− 7.71
B	− 17.85	+ 8.91	− 11.16	− 0.38	− 0.11	− 7.78
BTPD	− 13.56	+ 0.91	− 46.39	+ 0.04	− 0.06	− 30.52
WTPD	− 7.89	− 3.97	− 2.65	+ 0.14	0	+ 4.01

Mammals have been coded as follows: American Beaver (AB), American Red Squirrel (ARS), Bison (B), Black-tailed Prairie Dog (BTPD), and White-tailed Prairie Dog (WTPD)

management have been suggested to mitigate land-use disputes in areas of conservation importance (Liu et al. 2010). In this respect, our model and its outputs have the power to inform land-use policy and resource management at the local scale, as well. Fig. 10, for example, illustrates how spatial information obtained in this study can be used to inform stakeholders about the risk of habitat loss resulting from Development. Such derivatives can be extrapolated from the outputs depending on the scale and nature of the issue under consideration.

Mammal diversity in ecosystems can be used as an indicator to determine the impacts of humans on biodiversity (Morrison et al. 2007). Between 1900 and 2015, more than 170 mammal species have lost $\geq$ 30% of their geographic ranges and more than 40% of species have experienced > 80% range shrinkage, worldwide (Ceballos et al. 2017). Urbanization and agriculture are among the main drivers of biodiversity loss worldwide (WWF 2020). Although the UMRB is not considered as an urbanized region (Stoy et al. 2018), land conversion, mainly from grassland and wetland ecosystems to cropland (Wright and Wimberly 2013; Wimberly et al. 2018), has been an underlying ecological issue in the region since the 1850s, and there is new evidence to show the rate of urban growth in parts of the region is increasing, as well (US Census Bureau 2018). Therefore, the impacts of Development and Agriculture on mammals are higher compared to Recreation and Tradition in the UMRB. In this study, however, we assumed that human activities linked to Recreation and Tradition would have profound impacts on wildlife habitats, even if these impacts can be indirect and/or slower. Outdoor recreation is associated with a wide range of environmental issues (Mieczkowski 1995; Holden 2016). For example, even if land-cover change is properly controlled, noise pollutions force mammals

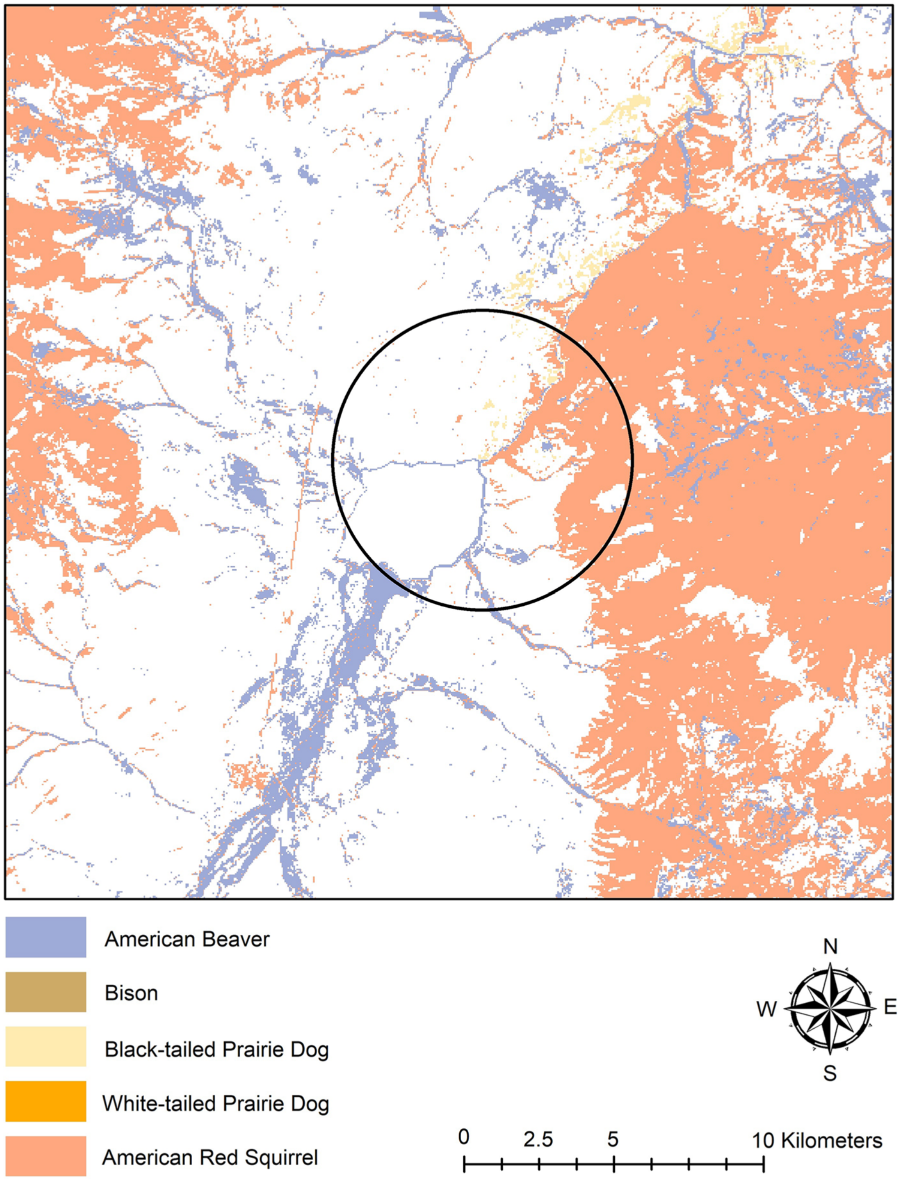

Fig. 10 A local-level example of mapping SVs for envisioning the potential for habitat degradation as a result of human activities. The wildlife habitat data was retrieved from GAP (U.S. Geological Survey 2018). This map depicts a RHA related to Development overlapping with wildlife habitats. Landscape metrics used in this study to quantify the impacts of Development on structure of wildlife habitats show the risk of further habitat degradation in certain areas of the UMRB (cf. Fig. 7 and Fig. 8)

to leave their habitats (Slabbekoorn et al. 2018) and light pollutions have widespread impacts on many of them (Gaston et al. 2013). Human infrastructures to support recreational activities also increase the risk of landscape fragmentation, weed and pest dispersal (Pickering and Mount 2010) and fires (Barros et al. 2015). Human activities associated with Tradition also adversely affect wildlife habitats through the spread of invasive species (Naylor et al. 2001; Radosevich et al. 2007), landscape fragmentation (Jordan 1993; Aldrich et al. 2006; Sorice et al. 2012), and the extraction of natural resources (Hessburg and Agee 2003). A meta-analysis of 76 studies of 62 mammal species world-wide showed a strong effect of different human

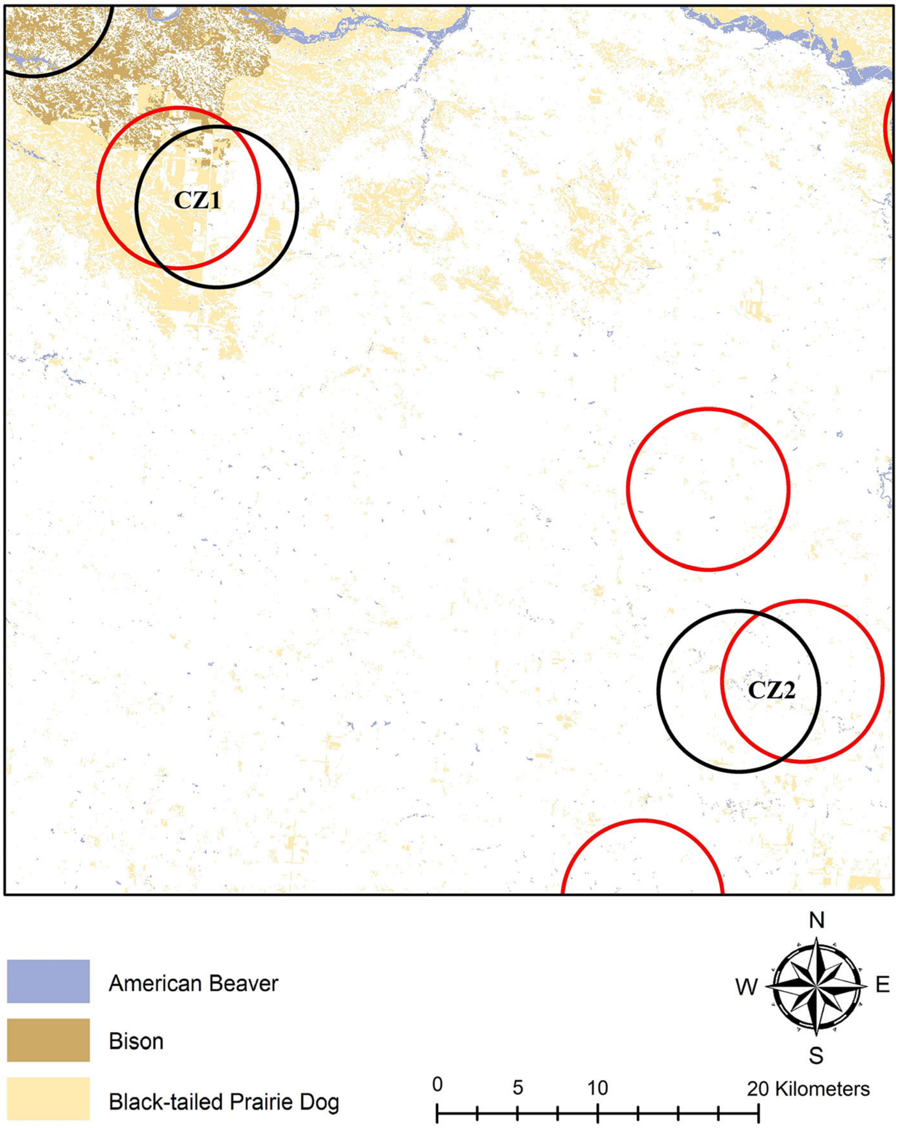

Fig. 11 A local-scale example of conflict of interests between human activities, where two RHAs linked to two different SVs overlap: Development (red) vs. Conservation (black). The wildlife habitat data was retrieved from GAP (U.S. Geological Survey 2018). There exist two conflict zones (CZ1 and CZ2) on this map. The overlap in CZ1 has a higher impact on wildlife habitats whereas in CZ2, the level of threat to biodiversity is considerably lower because the amount of wildlife habitat in CZ2 is small

activities, including hiking, walking, and hunting, on daily patterns of wildlife activity (Gaynor et al. 2018). These activities are linked to Recreation and Tradition in the region (cf. Arthun and Holechek 1982; Flores 1991; Isenberg 2020).

Implications for biodiversity conservation

Habitat degradation

We observed 25 interactions between human activities and wildlife habitats in socioecological hotspots of the UMRB (Fig. 4). More interactions are also possible when two or more than two RHAs overlap. Irrespective of these overlaps between human activities and wildlife habitats, there is also a high chance of overlap between two or more RHAs. As an example, Fig. 11 shows the capacity of our model to detect the potential for conflict of interests among human activities at the local scale, where RHAs of different human activities overlap. Such applications of the model are sensitive to the spatial ecology of species of interest, land-use legacies, human population density, and study scales (cf. Pacione 2009).

One important action to take in the face of these issues is to increase the multifunctionality of landscapes, perhaps because landscape multifunctionality could have disproportionate benefits on biodiversity conservation (Schulte et al. 2017). Multifunctional landscapes can mitigate the impacts of human activities on biodiversity through strategic placement of conservation practices within agricultural landscapes (Meurk and Swaffield 2000; Jarchow et al. 2015). In the UMRB, about one-fourth of the land is covered by croplands (Table 5). In addition to agricultural lands, the concept of multifunctional landscapes has been extended to describe how the land can be shared between people and wildlife in urbanized landscapes (Rosenzweig 2003).

The changes that we detected in the values of landscape metrics suggest that human activities linked to SVs can lead to landscape transformation through habitat loss, subdivision, dispersion, and shrinkage; however, the magnitude of changes was not the same for all mammal species. For example, habitats of the bison could be heavily encroached by human activities (Tables 8, 10; Fig. 4). Habitats of the American beaver and black-tailed prairie dog were also subject to the risk of habitat loss and a high level of exposure to human activities. If the land is used in the future based on SVs, the structure of patches of vegetation suitable for the persistence and colonization of these three mammals is expected to be adversely affected across socioecological hotspots (cf. Tables 2, 8, 9, 10; Fig. 7).

We observed that the American beaver is extremely sensitive to human activities, perhaps because this species is completely confined to water bodies and riparian zones (Collen and Gibson 2000). In the process of landscape simulation, the change in the value of GYRATE_AM for habitats of this species indicated that its habitats may be severely influenced by human activities. Habitats of the American red squirrel is less likely to be influenced by SVs linked to Recreation or Tradition. In addition, a very small portion of the land, which can provide habitat for this species, overlaps with areas valued for Development and Agriculture (Fig. 4). Except for PD, the values of other landscape metrics that we used to compute its habitats declined after applying the impacts of SVs (Fig. 7). The American red squirrel can contribute to the natural regeneration of forest ecosystems of the UMRB (i.e. Needleleaf; Broadleaf Deciduous; Mixed; Table 5; Goheen and Swihart 2003). Thus, habitat degradation can have adverse consequences for both this species and forest ecosystems of the region in the long run.

Conservation and restoration

To rewild (Foreman 2004; Manning 2011) the Great Plains of the United States, landscape restoration is essential. Habitat requirements of keystone species should be considered as an integral part of this process. Bison positively impact the diversity of arthropods (Nickell et al. 2018) and plant communities, including forbs and graminoids of tallgrass prairie landscapes (Rosas et al. 2008). In areas covered by the class Grassland (47% of the UMRB; Table 5), colonies of black- and white-tailed prairie dogs can increase vegetation heterogeneity (Gervin et al. 2019) and support biodiversity through the provision of habitat and food source for a wide range of invertebrates, amphibians, reptiles, birds, and mammals (Hoogland 2013). These mammals also contribute to soil bioturbation (Table 1). The presence of the American beaver is positively correlated with a higher level of plant and bird species richness (Cooke and Zack 2008; Law et al 2017); enhances water quality (Demmer and Beschta 2008); and leads to greater diversity in wetlands (Law et al. 2019), forests (Nummi and Kuuluvaine 2013) and streams (Gibson and Olden 2014). The importance of this species is so high that some researchers call for restoring its habitats in urban landscapes (Bailey et al. 2019).

In this study, approximately 2.7 million ha of the land that people valued for Conservation have an overlap with wildlife habitats (Table 7); however, a large portion of this overlap is also valued for other human activities (Fig. 8). The remaining areas have the capacity to serve as strategic zones for landscape restoration and/or biodiversity conservation, mainly because they contain suitable environmental

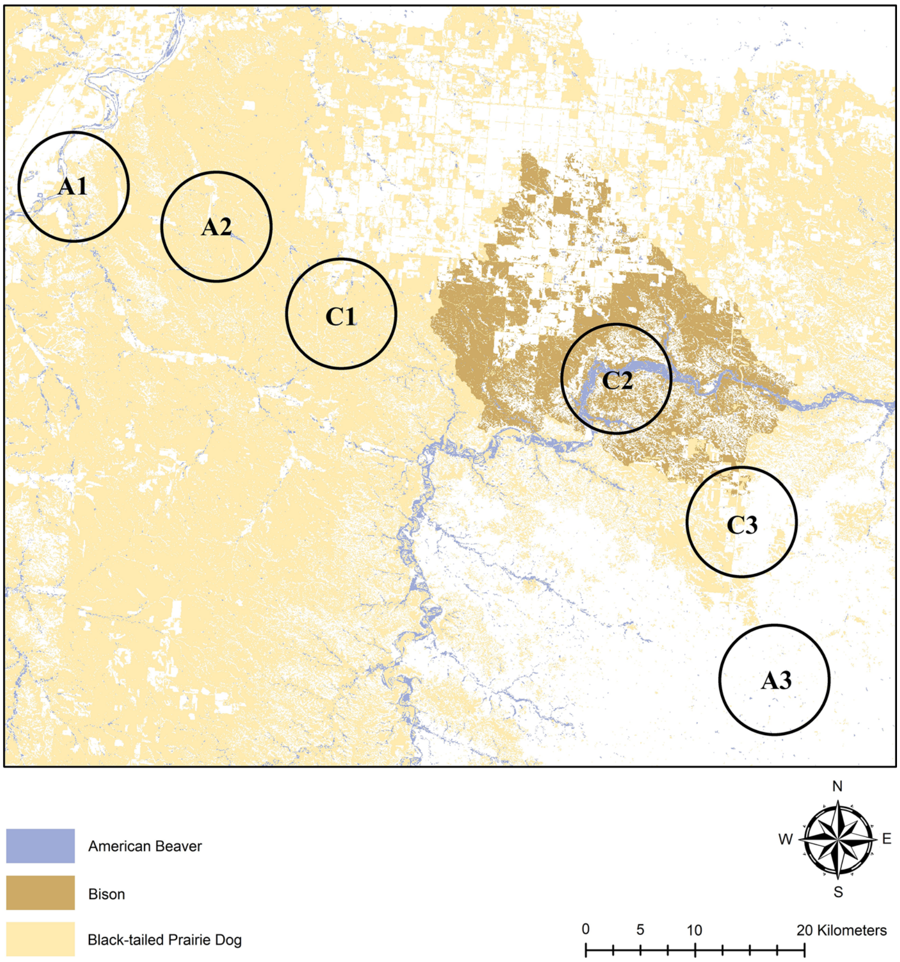

Fig. 12 Six RHAs mapped to show the impacts of Agriculture (A) and Conservation (C) on wildlife habitats. The wildlife habitat data was retrieved from GAP (U.S. Geological Survey 2018). According to this map, two RHAs related to Agriculture overlie expanses of biodiversity significance, where the land simultaneously has the capacity to function as habitat for keystone species (i.e. A1 and A2). The location of the third RHA (i.e. A3), however, is less harmful to biodiversity

conditions for species persistence and colonization, and at the same time, also have the social support needed to implement conservation and restoration practices. Previous research shows that public support can enhance the odds of success in landscape restoration and/or biodiversity conservation practices (Norton 2016; Sullivan and Molles 2016; Butler and Schultz 2019) and change in public attitudes toward environmental and land-use issues is essential to build a proper platform for biodiversity conservation through landscape restoration (Novacek 2008; Manfredo 2017). Our model developed an in-depth understanding of the spatial distribution of social support for landscape restoration and/or biodiversity conservation in socioecological hotspots of the UMRB. Fig. 12, for example, shows six RHAs linked to Agriculture and Conservation in a mosaic of the UMRB. In this mosaic, areas valued for Conservation can provide opportunities for restoration and/or biodiversity conservation. Areas that wildlife species coexist should take precedence for conservation/restoration. In this part of the UMRB, priority must be given to C2, C3, and C1, respectively.

Limitations

We put forward four important limitations associated with this study that should be considered in future research.

First, our model and its outputs were sensitive to the spatial scale of land cover data (cell size: 30 m), the thresholds defined in FRAGSTATS (e.g. RHA: 5 km, cell size: 50 m), as well as the spatial ecology of chosen species derived from the literature. The nature and magnitude of impacts imposed by human activities on the structure of wildlife habitats are not the same. In addition, not all species respond to these impacts in the same way. Particular attention should be placed on these issues to avoid the misinterpretation of results. Where the spatial scale and/or species of interest are different, these issues should be cautiously considered. Second, given the scale of study region, the model should be checked at different scales to avoid the misinterpretation of data. For example, after applying the impacts of human activities on wildlife habitats, the values of GYRATE_AM and AI increased in habitats of the white-tailed prairie dog. However, based on a local-scale monitoring of the current spatial distribution of suitable habitats for this species, we found that in the process of landscape simulation, some dispersed and isolated habitat patches were eliminated in a way that the values of these landscape metrics increased. Third, the accuracy of interpretations at the local scale depends, to a large extent, on the spatial and ecological knowledge of stakeholders who may have different, if not conflicting, interests and priorities. Fourth, the methods that we used to map SVs determined the spatial distribution of locations that people value for specific human activities. We sampled 22 human population centers across four states of Montana, North Dakota, South Dakota, and Wyoming to provide the most accurate picture of the locations that people value their landscapes for particular human activities. Despite this, we still assume that the location of sampled human population centers, as well as the place of residence could partly influence the spatial distribution of SVs.

Conclusion

In this study, we developed and used a spatially explicit model to provide an integrated picture of possible interactions between landscapes of social significance and wildlife habitats in a large mosaic of the Great Plains of the United States. If used as predictors of land-use disputes, our model showed SVs of people toward their landscapes have the potential to act as drivers of landscape transformation through habitat loss, habitat subdivision, habitat dispersion, and habitat shrinkage; however, the magnitude of impacts varied among human activities and species across landscapes of the UMRB. The results of this study raised concerns about the future of biodiversity in the UMRB and concurrently depicted opportunities for landscape restoration and/or biodiversity conservation, where social support and environmental conditions are well aligned. As discussed, the model that we used to map SVs in relation to wildlife habitats was associated with some uncertainties and limitations, and therefore, further research is required for its development. Concurrently, change in SVs and public attitudes toward land use is essential to avoid further biodiversity loss in this region.

In light of this study, we define three questions for future research: (1) how can the capacity of this spatially explicit model be improved to address more complex issues related to the coexistence of human and wildlife in socioecological hotspots?; (2) how can this model be used as a catalyst to inform a wide spectrum of stakeholders and make a multi-scale change, from community to region, in SVs and attitudes toward land-use issues?; (3) how can the model be used by other researchers in other parts of the world to evaluate its capacity and relevance to the current socioecological challenges?

Acknowledgements This work was supported by the National Science Foundation through the EPSCoR Track II cooperative agreement OIA-1632810 and NSF DBI-1560048. We thank Darius Semmens, Selena Ahmed, Julia Haggerty for feedback on the survey instrument and methods; Dylan Lewis, Tanner Hall, Jacinda Maassen, Jade Muller-Smit, Pamela Jackson, Daniel Whirlwind Soldier Petite, Mitchell Houska, Lori Peterson, Brad Frazier, Ann McGehrin, Rebecca Sistad for assistance conducting the surveys; and Andrew Baltensperger and Andrew Sechrist for assistance with the survey data management.

References

Aldrich SP, Walker RT, Arima EY, Caldas MM, Browder JO, Perz S (2006) Land-cover and land-use change in the Brazilian Amazon: smallholders, ranchers, and frontier stratification. Econ Geogr 82(3):265–288

Alessa LN, Kliskey AA, Brown G (2008) Social–ecological hotspots mapping: a spatial approach for identifying coupled social–ecological space. Landsc Urban Plan 85(1):27–39

Allred BW, Fuhlendorf SD, Hamilton RG (2011) The role of herbivores in Great Plains conservation: comparative ecology of bison and cattle. Ecosphere 2(3):1–17

Alvey AA (2006) Promoting and preserving biodiversity in the urban forest. Urban For Urban Green 5(4):195–201

Arthun D, Holechek JL (1982) The North American bison History. Rangel Arch 4(3):123–125

Bagstad KJ, Reed JM, Semmens DJ, Sherrouse BC, Troy A (2016) Linking biophysical models and public preferences for ecosystem service assessments: a case study for the Southern Rocky Mountains. Reg Environ Change 16(7):2005–2018

Bailey DR, Dittbrenner BJ, Yocom KP (2019) Reintegrating the North American beaver (*Castorcanadensis*) in the urban landscape. Wiley Interdiscip Rev Water 6(1):e1323

Ban NC, Mills M, Tam J, Hicks CC, Klain S, Stoeckl N, Bottrill MC, Levine J, Pressey RL, Satterfield T, Chan KM (2013) A social–ecological approach to conservation planning: embedding social considerations. Front Ecol Environ 11(4):194–202

Barbosa P, Schumaker NH, Brandon KR, Bager A, Grilo C (2020) Simulating the consequences of roads for wildlife population dynamics. Landsc Urban Plan 193:103672

Barros A, Monz C, Pickering C (2015) Is tourism damaging ecosystems in the Andes? Current knowledge and an agenda for future research. Ambio 44(2):82–98

Beever EA, Hall LE, Varner J, Loosen AE, Dunham JB, Gahl MK, Smith FA, Lawler JJ (2017) Behavioral flexibility as a mechanism for coping with climate change. Front Ecol Environ 15(6):299–308

Benítez-López A, Alkemade R, Verweij PA (2010) The impacts of roads and other infrastructure on mammal and bird populations: a meta-analysis. Biol Cons 143(6):1307–1316

Bennett NJ, Dearden P (2014) Why local people do not support conservation: community perceptions of marine protected area livelihood impacts, governance and management in Thailand. Mar Policy 44:107–116

Bennett NJ, Di Franco A, Calò A, Nethery E, Niccolini F, Milazzo M, Guidetti P (2019) Local support for conservation is associated with perceptions of good governance, social impacts, and ecological effectiveness. Conserv Lett 12(4):e12640

Biedenweg K, Williams K, Cerveny L, Styers D (2019) Is recreation a landscape value? Exploring underlying values in landscape values mapping. Landsc Urban Plan 185:24–27

Botequilha-Leitao A, Ahern J (2002) Applying landscape ecological concepts and metrics in sustainable landscape planning. Landsc Urban Plan 59(2):65–93

Botequilha-Leitao A, Miller J, Ahern J, Mc Garigal K (2006) Measuring landscapes: a planner's handbook. Island Press, Washington, D.C.

Bradshaw CJ (2012) Little left to lose: deforestation and forest degradation in Australia since European colonization. J Plant Ecol 5(1):109–120

Brown G, Weber D, De Bie K (2014) Assessing the value of public lands using public participation GIS (PPGIS) and social landscape metrics. Appl Geogr 53:77–89

Brown G, Reed P, Raymond CM (2020) Mapping place values: 10 lessons from two decades of public participation GIS empirical research. Appl Geogr 116:102156

Bryan BA, Raymond CM, Crossman ND, King D (2011) Comparing spatially explicit ecological and social values for natural areas to identify effective conservation strategies. Conserv Biol 25(1):172–181

Butchart SH, Walpole M, Collen B, Van Strien A, Scharlemann JP, Almond RE, Baillie JE, Bomhard B, Brown C, Bruno J, Carpenter KE (2010) Global biodiversity: indicators of recent declines. Science 328(5982):1164–1168

Butler WH, Schultz CA (2019) A new era for collaborative forest management: policy and practice insights from the Collaborative Forest Landscape Restoration Program. Routledge, Abingdon

Carnes M (2019) Land-use change and social values in micropolitan communities in the Upper Missouri River Basin. Master thesis. University of South Dakota, Vermillion

Ceballos G, Ehrlich PR, Dirzo R (2017) Biological annihilation via the ongoing sixth mass extinction signaled by vertebrate population losses and declines. Proc Nat Acad Sci 114(30):E6089–E6096

Collen P, Gibson RJ (2000) The general ecology of beavers (*Castor* spp.), as related to their influence on stream ecosystems and riparian habitats, and the subsequent effects on fish–a review. Rev Fish Biol Fish 10(4):439–461

Collier KJ, Parkyn SM, Rabeni CF (1997) Koura: a keystone species. Water Atmos 5(1):18–20

Commission for Environmental Cooperation (2020) 2010 Land Cover of North America at 30 Meters, North American Land Change Monitoring System, Ed. 2.0. 2020. Available online at: http://www.cec.org/north-american-environmental-atlas/land-cover-2010-landsat-30m

Cooke HA, Zack S (2008) Influence of beaver dam density on riparian areas and riparian birds in shrubsteppe of Wyoming. West N Am Nat 68(3):365–373

Cushman SA, McGarigal K, Neel MC (2008) Parsimony in landscape metrics: strength, universality, and consistency. Ecol Ind 8(5):691–703

Demmer R, Beschta RL (2008) Recent history (1988–2004) of beaver dams along Bridge Creek in central Oregon. Northwest Sci 82(4):309–318

Dietz MS, Belote RT, Gage J, Hahn BA (2020) An assessment of vulnerable wildlife, their habitats, and protected areas in the contiguous United States. Biol Cons 248:108646

Dirzo R, Young HS, Galetti M, Ceballos G, Isaac NJ, Collen B (2014) Defaunation in the Anthropocene. Science 345(6195):401–406

Dramstad WE, Olson JD, Forman RTT (1996) Landscape ecology principles in landscape architecture and land-use planning. Harvard University Graduate School of Design and Island Press, Cambridge

Elmqvist T, Fragkias M, Goodness J, Güneralp B, Marcotullio PJ, McDonald RI, Parnell S, Schewenius M, Sendstad M, Seto KC, Wilkinson C (2013) Urbanization, biodiversity and ecosystem services: challenges and opportunities: a global assessment. Springer, Dordrecht

Ewers RM, Kliskey AD, Walker S, Rutledge D, Harding JS, Didham RK (2006) Past and future trajectories of forest loss in New Zealand. Biol Cons 133(3):312–325

Flores D (1991) Bison ecology and bison diplomacy: the southern plains from 1800 to 1850. J Am Hist 78(2):465–485
Foreman D (2004) Rewilding North America: a vision for conservation in the 21st century. Island Press, Washington, D.C.
Forman RT (1995) Land Mosaics: The ecology of landscapes and regions. The ecological design and planning reader. Island Press, Washington, D.C.
Frank B, Glikman JA, Marchini S (eds) (2019) Human-wildlife interactions: turning conflict into coexistence, vol 23. Cambridge University Press, Cambridge
Garibaldi A, Turner N (2004) Cultural keystone species: implications for ecological conservation and restoration. Ecol Soc 9(3):1
Gaston KJ, Bennie J, Davies TW, Hopkins J (2013) The ecological impacts of nighttime light pollution: a mechanistic appraisal. Biol Rev 88(4):912–927
Gaynor KM, Hojnowski CE, Carter NH, Brashares JS (2018) The influence of human disturbance on wildlife nocturnality. Science 360(6394):1232–1235
Gergely KJ, Boykin KG, McKerrow AJ, Rubino MJ, Tarr NM, Williams SG (2019) Gap Analysis Project (GAP) terrestrial vertebrate species richness maps for the conterminous U.S.: U.S. Geological Survey Scientific Investigations Report 2019–5034, 99. https://doi.org/10.3133/sir20195034
Gervin CA, Bruun HH, Seipel T, Burgess ND (2019) Presence of both active and inactive colonies of prairie dogs contributes to higher vegetation heterogeneity at the landscape scale. Am Midland Nat 181(2):183–194
Gibson PP, Olden JD (2014) Ecology, management, and conservation implications of North American beaver (Castor canadensis) in dryland streams. Aquat Conserv Mar Freshw Ecosyst 24(3):391–409
Goheen JR, Swihart RK (2003) Food-hoarding behavior of gray squirrels and North American red squirrels in the central hardwoods region: implications for forest regeneration. Can J Zool 81(9):1636–1639
Gökbulak F (2002) Effect of American bison (*Bisonbison* L.) on the recovery and germinability of seeds of range forage species. Grass Forage Sci 57(4):395–400
Goldewijk KK, Ramankutty N (2004) Land cover change over the last three centuries due to human activities: the availability of new global data sets. GeoJournal 61(4):335–344
Guerrero AM, Wilson KA (2017) Using a social–ecological framework to inform the implementation of conservation plans. Conserv Biol 31(2):290–301
Hale SL, Koprowski JL (2018) Ecosystem-level effects of keystone species reintroduction: a literature review. Restor Ecol 26(3):439–445
Hessburg PF, Agee JK (2003) An environmental narrative of inland northwest United States forests, 1800–2000. For Ecol Manag 178(1–2):23–59
Holden A (2016) Environment and tourism. Routledge Oxon, New York
Hoogland J (ed) (2013) Conservation of the black-tailed prairie dog: saving North America's western grasslands. Island Press, Washington, D.C
Hu M, Li Z, Wang Y, Jiao M, Li M, Xia B (2019) Spatio-temporal changes in ecosystem service value in response to land-use/cover changes in the Pearl River Delta. Resour Conserv Recycl 149:106–114
Isenberg AC (2020) The destruction of the bison: an environmental history, 1750–1920. Cambridge University Press, Cambridge
Ives CD, Kendal D (2014) The role of social values in the management of ecological systems. J Environ Manag 144:67–72
Jarchow ME, Liebman M, Dhungel S, Dietzel R, Sundberg D, Anex RP, Thompson ML, Chua T (2015) Trade-offs among agronomic, energetic, and environmental performance characteristics of corn and prairie bioenergy cropping systems. Gcb Bioenergy 7(1):57–71
Jarchow M, Carnes M, Semmens D (2018) Mapping social values toward land use in the Upper Missouri River Basin. Presented at A Community on Ecosystem Services, Washington, D.C.
Jarchow M, Swanson D, Kerby J (2020) North American grasslands as multifunctional landscapes. In: Leal Filho W, Azul A, Brandli L, Lange Salvia A, Wall T (eds) Life on land. Encyclopedia of the UN sustainable development goals. Springer, Cham, pp 1–20
Jordan TG (1993) North American cattle-ranching frontiers: origins, diffusion, and differentiation. University of New Mexico Press, Albuquerque, p 147208
Karimi A, Adams VM (2019) Planning for the future: combining spatially-explicit public preferences with tenure policies to support land-use planning. Land Use Policy 82:497–508
Karimi A, Hockings M (2018) A social-ecological approach to land-use conflict to inform regional and conservation planning and management. Landsc Ecol 33(5):691–710
Kenter JO, Raymond CM, Van Riper CJ, Azzopardi E, Brear MR, Calcagni F, Christie I, Christie M, Fordham A, Gould RK, Ives CD (2019) Loving the mess: navigating diversity and conflict in social values for sustainability. Sustain Sci 14(5):1439–1461
Knapp AK, Blair JM, Briggs JM, Collins SL, Hartnett DC, Johnson LC, Towne EG (1999) The keystone role of bison in North American tallgrass prairie: *Bison* increase habitat heterogeneity and alter a broad array of plant, community, and ecosystem processes. Bioscience 49(1):39–50
Kotliar NB, Baker BW, Whicker AD, Plumb G (1999) A critical review of assumptions about the prairie dog as a keystone species. Environ Manag 24(2):177–192
Kotliar NB, Miller BJ, Reading RP, Clark TW, Hoogland JL (2006) The prairie dog as a keystone species. Island Press, Washington, D.C.
Kupfer JA (2012) Landscape ecology and biogeography: rethinking landscape metrics in a post-FRAGSTATS landscape. Prog Phys Geogr 36(3):400–420
Lacher TE Jr, Davidson AD, Fleming TH, Gómez-Ruiz EP, McCracken GF, Owen-Smith N, Peres CA, Vander Wall SB (2019) The functional roles of mammals in ecosystems. J Mammal 100(3):942–964
Lampard EE (1955) The history of cities in the economically advanced areas. Econ Dev Cult Change 3(2):81–136
Lapointe M, Cumming GS, Gurney GG (2019) Comparing ecosystem service preferences between urban and rural dwellers. Bioscience 69(2):108–116

Law A, Gaywood MJ, Jones KC, Ramsay P, Willby NJ (2017) Using ecosystem engineers as tools in habitat restoration and rewilding: beaver and wetlands. Sci Total Environ 605:1021–1030

Law A, Levanoni O, Foster G, Ecke F, Willby NJ (2019) Are beavers a solution to the freshwater biodiversity crisis? Divers Distrib 25(11):1763–1772

Liu J, Ouyang Z, Miao H (2010) Environmental attitudes of stakeholders and their perceptions regarding protected area-community conflicts: a case study in China. J Environ Manag 91(11):2254–2262

Madden F (2004) Creating coexistence between humans and wildlife: global perspectives on local efforts to address human–wildlife conflict. Hum Dimens Wildl 9(4):247–257

Manfredo MJ, Bruskotter JT, Teel TL, Fulton D, Schwartz SH, Arlinghaus R, Oishi S, Uskul AK, Redford K, Kitayama S, Sullivan L (2017) Why social values cannot be changed for the sake of conservation. Conserv Biol 31(4):772–780

Manning R (2011) Rewilding the West: restoration in a prairie landscape. University of California Press, Berkeley

Mantyka-Pringle CS, Visconti P, Di Marco M, Martin TG, Rondinini C, Rhodes JR (2015) Climate change modifies risk of global biodiversity loss due to land-cover change. Biol Cons 187:103–111

McGarigal K (2015) FRAGSTATS Help. http://www.umass.edu/landeco/research/fragstats/documents/fragstats.help.4.2.pdf

McGarigal K, Cushman SA, Ene E (2012) FRAGSTATS v4: spatial pattern analysis program for categorical and continuous maps. Computer software program produced by the authors at the University of Massachusetts, Amherst. http://www.umass.edu/landeco/research/fragstats/fragstats.html

Meade TA (2016) History of modern Latin America: 1800 to the present. Wiley, Hoboken

Mendes CP, Carreira D, Pedrosa F, Beca G, Lautenschlager L, Akkawi P, Bercê W, Ferraz KM, Galetti M (2020) Landscape of human fear in Neotropical rainforest mammals. Biol Cons 241:108257

Meurk CD, Swaffield SR (2000) A landscape ecological framework for indigenous regeneration in rural New Zealand-Aotearoa. Landsc Urban Plan 50(1–3):129–144

Mieczkowski Z (1995) Environmental issues of tourism and recreation. University Press of America, London

Miller B, Reading R, Hoogland J, Clark T, Ceballos G, List R, Forrest S, Hanebury L, Manzano P, Pacheco J, Uresk D (2000) The role of prairie dogs as a keystone species: response to Stapp. Conserv Biol 14(1):318–321

Mills LS, Soulé ME, Doak DF (1993) The keystone-species concept in ecology and conservation. Bioscience 43(4):219–224

Morrison JC, Sechrest W, Dinerstein E, Wilcove DS, Lamoreux JF (2007) Persistence of large mammal faunas as indicators of global human impacts. J Mammal 88(6):1363–1380

Naiman RJ (1988) Animal influences on ecosystem dynamics. Bioscience 38(11):750–752

Naylor RL, Williams SL, Strong DR (2001) Aquaculture–a gateway for exotic species. Science. https://doi.org/10.1126/science.1064875

Nickell Z, Varriano S, Plemmons E, Moran MD (2018) Ecosystem engineering by bison (*Bison bison*) wallowing increases arthropod community heterogeneity in space and time. Ecosphere 9(9):e02436

Norton DA, Young LM, Byrom AE, Clarkson BD, Lyver POB, McGlone MS, Waipara NW (2016) How do we restore New Zealand's biological heritage by 2050? Ecol Manag Restor 17(3):170–179

Novacek MJ (2008) Engaging the public in biodiversity issues. Proc Natl Acad Sci 105(Supplement 1):11571–11578

Nummi P, Holopainen S (2014) Whole-community facilitation by beaver: ecosystem engineer increases waterbird diversity. Aquat Conserv Mar Freshw Ecosyst 24(5):623–633

Nummi P, Kuuluvaine T (2013) Forest disturbance by an ecosystem engineer: beaver in boreal forest landscapes. Boreal Environ Res 18(suppl A):13–24

Nyhus PJ (2016) Human–wildlife conflict and coexistence. Annu Rev Environ Resour 41:143–171

Okamoto DK, Poe MR, Francis TB, Punt AE, Levin PS, Shelton AO, Armitage DR, Cleary JS, Dressell SC, Jones R, Kitka H (2020) Attending to spatial social–ecological sensitivities to improve trade-off analysis in natural resource management. Fish Fish 21(1):1–12

Opdam P, Wascher D (2004) Climate change meets habitat fragmentation: linking landscape and biogeographical scale levels in research and conservation. Biol Cons 117(3):285–297

Pacione M (2009) Urban geography: a global perspective. Routledge, New York

Pereira HM, Leadley PW, Proença V, Alkemade R, Scharlemann JP, Fernandez-Manjarrés JF, Araújo MB, Balvanera P, Biggs R, Cheung WW, Chini L (2010) Scenarios for global biodiversity in the 21st century. Science 330(6010):1496–1501

Pickering C, Mount A (2010) Do tourists disperse weed seed? A global review of unintentional human-mediated terrestrial seed dispersal on clothing, vehicles and horses. J Sustain Tour 18(2):239–256

Radosevich SR, Holt JS, Ghersa CM (2007) Ecology of weeds and invasive plants: relationship to agriculture and natural resource management. Wiley, Hoboken

Ramankutty N, Foley JA (1999) Estimating historical changes in land cover: North American croplands from 1850 to 1992. Glob Ecol Biogeogr 8(5):381–396

Ramankutty N, Graumlich L, Achard F, Alves D, Chhabra A, DeFries RS, Foley JA, Geist H, Houghton RA, Goldewijk KK, Lambin EF (2006) Global land-cover change: recent progress, remaining challenges. In: Lambin EF, Geist H (eds) Land-use and land-cover change. Springer, Berlin, Heidelberg, pp 9–39

Rastandeh A (2018) Urban biodiversity in an era of climate change: Towards an optimised landscape pattern in support of indigenous wildlife species in urban New Zealand. PhD thesis. Victoria University of Wellington, Wellington

Riechers M, Balázsi Á, Abson D, Fischer J (2020) The influence of landscape change on multiple dimensions of human–nature connectedness. Ecol Soc 25(3):3

Rosas CA, Engle DM, Shaw JH, Palmer MW (2008) Seed dispersal by *Bisonbison* in a tallgrass prairie. J Veg Sci 19(6):769–778

Rosenzweig ML (2003) Win-win ecology: how the Earth species can survive in the midst of human enterprise. Oxford University Press, Oxford

Samson F, Knopf F (1994) Prairie conservation in North America. Bioscience 44(6):418–421

Schulte LA, Niemi J, Helmers MJ, Liebman M, Arbuckle JG, James DE, Kolka RK, O'Neal ME, Tomer MD, Tyndall JC, Asbjornsen H (2017) Prairie strips improve biodiversity and the delivery of multiple ecosystem services from corn–soybean croplands. Proc Natl Acad Sci 114(42):11247–11252

Sherrouse BC, Clement JM, Semmens DJ (2011) A GIS application for assessing, mapping, and quantifying the social values of ecosystem services. Appl Geogr 31(2):748–760

Sieber R (2006) Public participation geographic information systems: a literature review and framework. Ann Assoc Am Geogr 96(3):491–507

Slabbekoorn H, McGee J, Walsh EJ (2018) Effects of man-made sound on terrestrial mammals. In: Slabbekoorn H, Dooling R, Popper A, Fay R (eds) Effects of anthropogenic noise on animals. Springer Handbook of Auditory Research, vol 66. Springer, New York

Song XP, Hansen MC, Stehman SV, Potapov PV, Tyukavina A, Vermote EF, Townshend JR (2018) Global land change from 1982 to 2016. Nature 560(7720):639–643

Sorice MG, Kreuter UP, Wilcox BP, Fox WE III (2012) Classifying land-ownership motivations in central, Texas, USA: a first step in understanding drivers of large-scale land cover change. J Arid Environ 80:56–64

Stoy PC, Ahmed S, Jarchow M, Rashford B, Swanson D, Albeke S, Bromley G, Brookshire ENJ, Dixon MD, Haggerty J, Miller P (2018) Opportunities and trade-offs among BECCS and the food, water, energy, biodiversity, and social systems nexus at regional scales. Bioscience 68(2):100–111

Sullivan JJ, Molles LE (2016) Biodiversity monitoring by community-based restoration groups in New Zealand. Ecol Manag Restor 17(3):210–217

Tarr NM (2019) Demonstrating a conceptual model for multi-species landscape pattern indices in landscape conservation. Landsc Ecol 34(9):2133–2147

Tscharntke T, Clough Y, Wanger TC, Jackson L, Motzke I, Perfecto I, Vandermeer J, Whitbread A (2012) Global food security, biodiversity conservation and the future of agricultural intensification. Biol Cons 151(1):53–59

Turner MG (1989) Landscape ecology: the effect of pattern on process. Annu Rev Ecol Syst 20(1):171–197

Urban MC (2015) Accelerating extinction risk from climate change. Science 348(6234):571–573

US Census Bureau (2018) American Community Survey: 2014–2018 ACS 5-Year Data Profile. Available from: https://www.census.gov/acs/www/data/data-tables-and-tools/data-profiles

U.S. Geological Survey (2018) Gap Analysis Project, U.S. Geological Survey—Gap Analysis Project Species Habitat Maps CONUS 2001: U.S. Geological Survey data release, Available from: https://doi.org/10.5066/F7V122T2

Walpole MJ, Leader-Williams N (2002) Tourism and flagship species in conservation. Biodivers Conserv 11:543–547

Wilkins K, Pejchar L, Garvoille R (2019) Ecological and social consequences of bison reintroduction in Colorado. Conserv Sci Pract 1(2):e9

Wimberly MC, Narem DM, Bauman PJ, Carlson BT, Ahlering MA (2018) Grassland connectivity in fragmented agricultural landscapes of the north-central United States. Biol Cons 217:121–130

Włodarczyk-Marciniak R, Frankiewicz P, Krauze K (2020) Socio-cultural valuation of Polish agricultural landscape components by farmers and its consequences. J Rural Stud 74:190–200

Wright CK, Wimberly MC (2013) Recent land use change in the Western Corn Belt threatens grasslands and wetlands. Proc Natl Acad Sci 110(10):4134–4139

WWF, Grooten M, Almond REA (eds) (2018) Living Planet Report 2018: Aiming Higher. WWF, Gland

WWF, Almond REA, Grooten M, Petersen T (eds) (2020) Living planet report 2020—bending the curve of biodiversity loss. WWF, Gland

Zhang W, Yu Y, Wu X, Pereira P, Borja MEL (2020) Integrating preferences and social values for ecosystem services in local ecological management: a framework applied in Xiaojiang Basin, Yunnan province, China. Land Use Policy 91:104339

Publisher's Note Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.