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Grounding simulation models with qualitative case studies: Toward a holistic framework to make climate science usable for US public land management

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ABSTRACT

Policies directing agencies and public land managers to incorporate climate change into management face several barriers. These stem, in part, from a disconnect between the information that is produced and the information needs of local resource managers. A disproportionate focus on the natural and physical sciences in climate vulnerability and adaptation assessment obscure understandings of complex social systems and the interactions and feedbacks in social-ecological systems. We use a qualitative case study of bison management on Department of the Interior-managed and tribal lands to explore how a social-science driven Determinants and Analogue Vulnerability Assessment (DAVA) can inform ecological response models, specifically simulation models that account for multiple drivers of change. First, we illustrate how a DAVA approach can help to: 1) identify key processes, entities, and interactions across scales; 2) document local impacts, indicators, and monitoring efforts of drought and climate; and 3) identify major tradeoffs and uncertainties. We then demonstrate how qualitative narratives can inform simulation models by: 1) prioritizing model components included in modeling efforts; 2) framing joint management and climate scenarios; and 3) parameterizing and evaluating model performance. We do this by presenting a conceptual joint agent-based/state-and-transition simulation modeling framework. Simulation models can represent multiple interacting variables and can identify surprising, emergent outcomes that might not be evident from qualitative analysis alone, and we argue that qualitative case studies can ground simulation models in local contexts and help make them more structurally realistic and useful. Together, these can provide a step toward developing actionable climate change adaptation strategies.

1. Introduction

Several policies created in the last decade directed public land managers to integrate considerations of climate change into management (DOI Order 3226, 2001; DOI Order 3289, 2009; EO 13514, 2009; EO 13653, 2013; EO 13693, 2015). Yet, a number of barriers limit implementation of climate adaptation actions on the ground, such as cross jurisdictional differences, lack of agency

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direction, conflicting mandates, insufficient time and funding, lack of information at temporal and spatial scales that are relevant to managers, and failure to consider the decision context of local managers, among others (Archie et al., 2014, 2012; Bierbaum et al., 2013; Jantarasami et al., 2010; Kemp et al., 2015; McNeeley, 2012).

Further, the Department of the Interior's (DOI) Climate Adaptation Plan acknowledged the impacts of climate change on biophysical resources, as well as human livelihoods and economies (Department of the Interior, 2014a). In addition, the National Park Service (NPS) Climate Change Response Strategy recognized that effective decision-making under climate change must consider the social, political, economic, and cultural contexts under which decision-making occurs (National Park Service, 2010). In this vein, several recent reviews have emphasized the utility of incorporating social science in natural resource management for characterizing complex social-ecological interactions in local contexts, and identifying realistic, effective, and sustainable management initiatives (Ban et al., 2013; Bennett et al., 2017; McNeeley et al., 2017; Miller et al., 2012; Sharp et al., 2014). Yet, the natural and physical sciences have dominated inputs to vulnerability and adaptation science (e.g., Glick et al., 2011), which leaves out important understandings of complex human systems, and the interactions and feedbacks among these linked social-ecological systems (Ban et al., 2013; McNeeley et al., 2017; Ojima et al., 2013).

These barriers to climate change adaptation are further complicated by the high degree of uncertainty associated with future climate change impacts in local places, and the ways in which social and ecological systems will interact under change (Miller and Morisette, 2014). Therefore, developing actionable adaptation strategies in public lands management requires a broader engagement with social science in general, but specifically for integrative social-ecological assessments.

To address this need, we explore linkages between qualitative and quantitative approaches by analyzing how qualitative case studies – using a combination of approaches that assess the contextual determinants of, and analogues for, vulnerability and adaptive capacity in local places (Ford et al., 2010; Füssel and Klein, 2006; Glantz, 1996, 1991; McNeeley and Shulski, 2011; Smit and Pilifosova, 2003; Smit and Wandel, 2006), which we refer to here as a Determinants and Analogue Vulnerability Assessment (DAVA) – can inform ecological response models, specifically simulation models that account for multiple drivers of change. Simulation models can represent complex, multi-scale social-ecological dynamics (An et al., 2014; Axelrod, 1997), such as the combined effects of climate, disturbances, and management alternatives under future, uncertain change (Creutzburg et al., 2015; Jarnevich et al., 2015; Miller et al., 2017; Symstad et al., 2017; Yospin et al., 2015). Yet, simulation models are most useful when they consider local management priorities and decision contexts (Miller and Morisette, 2014). Qualitative vulnerability assessments are place-based studies that work closely with resource managers to document the biophysical and social factors that determine vulnerability and adaptive capacity in local contexts (hence "determinants"), and often use past impacts and responses as analogues for impacts and responses that are plausible in the future (hence "analogues") (Ford et al., 2010; Smit and Pilifosova, 2003; Smit and Wandel, 2006). Thus, simulation models can represent multiple interacting variables and can identify surprising, emergent outcomes that might not be evident from qualitative analysis alone, and we argue that qualitative case studies can ground simulation models in local contexts and help make them more structurally realistic and useful.

We build on initial results of a qualitative comparative case study of bison management in southwest South Dakota and endangered fish recovery in northwest Colorado (McNeeley et al., 2016). Herein, we focus in more detail on the bison management case study, specifically with regards to the National Park Service (NPS) context, and the multi-scale, multi-jurisdictional relationships in which they are embedded. First, we illustrate how a DAVA approach can help to: 1) identify key processes, entities, and interactions across scales; 2) document local impacts, indicators, and monitoring efforts; and 3) determine major tradeoffs and uncertainties (Section 5.1). We then demonstrate how a DAVA can help prioritize the key components included in modeling efforts, frame the development of relevant management and climate scenarios, and support model parameterization and evaluation (Section 5.2). We present a conceptual joint agent-based/state-and-transition simulation modeling framework that can address key results from the DAVA approach.

2. Conceptual framework

Protected areas (PA) cannot be effectively managed based solely on ecological principles and apart from the surrounding environment in which they are embedded (Cumming et al., 2015; Hansen et al., 2014). Instead, sustainable management can only be achieved by recognizing PAs as social-ecological systems (Cumming et al., 2015 and the special issue therein; Miller et al., 2012).

Social-ecological systems are complex adaptive systems characterized by feedbacks, non-linearity, emergence, path dependence, and thresholds that interact across multiple spatial and temporal scales (Berkes et al., 2003). Ostrom's (2009) social-ecological system framework for PA management consists of four components: the resource system (e.g., NPS park unit); resource units (e.g., vegetation); governance system (e.g., management plans); and the actors involved (e.g., staff). Interactions occur within and between scales, from the sub-PA level, to the PA, which is further embedded in regional to international social, political, economic, and environmental settings (Cumming et al., 2015; Ostrom, 2009).

PAs are part of regional zones of social-ecological interactions (Cumming et al., 2015; Hansen et al., 2014; Mathevet et al., 2016). PA boundaries are often designated based on administrative considerations and/or aesthetic values, and not necessarily on ecological integrity (Hansen et al., 2011; Palomo et al., 2014). Therefore, management actions inside a PA impact, and are impacted by, responses outside the PA. For instance, land use decisions and land tenure arrangements (e.g., development) made outside park boundaries can impact PAs through disruption of ecological flows (e.g., water, disturbance) (DeFries et al., 2010; Hansen et al., 2011). Additionally, PAs are commonly adjacent to public and private landholders, all of which operate under diverse objectives and mandates (Bennett and McGinnis, 2008). This can lead to conflicts about how resources in a PA should be managed, which is particularly germane to PA management on U.S. public lands where local and extra-local stakeholders have a voice in land management decisions (Beier et al., 2009; Trosper, 2003). PAs are also embedded in national to global processes, such as climate change, policies, and tourism economies (Cumming et al., 2015).

The interactions and feedbacks between spatial scales are overlain by temporal scales of interactions, which can result in scale mismatches (Maciejewski et al., 2015). Spatial scale mismatches occur when PA boundaries are too small to effectively manage resources. In rangelands, fragmentation tightens the coupling between wildlife and vegetation, and can result in ecosystem degradation as wildlife continue to graze on preferred areas, and/or congregate at isolated water points (Hobbs et al., 2008). Temporal mismatches occur when the time required to take management action is not coincident with changes in ecological conditions (Crowder et al., 2006).

The interactions between social-ecological system components within and across scales suggest that system-wide behavior cannot be assessed by examining each component alone (Bennett and McGinnis, 2008; Liu et al., 2015). Below, we introduce some of the social and ecological factors that combine to impact bison management in the case study.

3. Case study Background: Bison management as a complex social-ecological system under change

Bison management in this region of the High Plains is complicated by myriad climate and social-ecological dynamics (Fig. 1). Future changes in these dynamics and their effects on bison exhibit a high degree of uncertainty, making planning difficult. Climate change projections suggest that the High Plains will see further warming and may experience increases in annual precipitation (Polley et al., 2013; Shafer et al., 2014). It is not well understood if net primary productivity and forage quality would increase as a function of increased precipitation and a longer growing season, or decrease as the benefits of increased precipitation are offset by warming temperatures, CO_2 fertilization, and increased evapotranspiration (Polley et al., 2013). Further, significant hydrological changes in snowpack are suggested, including alterations to the timing of precipitation events and early runoff (Ojima et al., 2015). Changes to the timing and seasonality of precipitation have the potential for devastating impacts to range condition as spring and early summer precipitation drives forage production (Smart et al., 2007).

The High Plains region has experienced several severe droughts in recent decades, and climate projections suggest an increase in the frequency and severity of droughts, which poses significant risks to managing bison (Ojima et al., 2015). For instance, bison preferentially graze around the edges of prairie dog colonies (Coppock et al., 1983), and prairie dog colonies expand during prolonged drought periods (Derner et al., 2006). Therefore, drought may not only lead to reductions in forage availability, but could also change the spatial arrangement of preferred grazing habitat. Wildfire also can significantly impact the amount of available forage available for bison and other grazers, and bison preferentially graze recently burned patches (Coppock and Detling, 1986). It is unclear the degree to which drought will increase or decrease fire likelihood (Allred et al., 2011; Amberg et al., 2012). Additionally, bison and other ungulates exploit woody and riparian areas more often under hotter and drier conditions (Allred et al., 2011). The congregation of ungulates in these areas promotes soil compaction (Amberg et al., 2012), and can further contribute to resource scarcity in fragmented landscapes (Hobbs et al., 2008).

Bison management must also consider a range of social, political, and institutional factors. Southwest South Dakota is a patchwork of public and private land ownership (Fig. 1). Private landowners and ranchers live around, and in some cases depend on, public lands managed by the NPS, Bureau of Land Management, and United States Forest Service. The DOI conserves bison in this region, and both Pine Ridge and Rosebud Reservations manage bison herds (Department of the Interior, 2014b).

Managing bison, especially under frequent and recurring drought, is challenging for several reasons. Bison in the NPS system are confined to fenced enclosures with limited water availability. These enclosures restrict long-range dispersal for adapting to climate variability, and therefore place pressure on forage and water availability for bison and other wildlife in the PA. For instance, at Wind Cave National Park (WICA) bison are housed within an approximately 28,000 acre enclosure in the park boundaries, with a recommended herd size of 350–500 animals (Department of the Interior, 2014b). Water availability at WICA is limited to only a few reliable sources of surface water streams (e.g., Beaver Creek, Highland Creek), as well as six developed springs distributed throughout the park (Wildlife Biologist, Personal Comm.). The surface water streams sink underground inside the park and charge the lakes in the cave system. WICA recently acquired an additional 5500 acres of rangeland in the southeast corner of the park. The bison herd at Badlands National Park (BADL) roams within an approximately 64,000 acre enclosure that falls predominately within the boundaries of the Badlands Wilderness Area (Fig. 1; Amberg et al., 2012; Department of the Interior, 2014b). Water availability within the bison range is limited as there are no perennial streams in the fenced enclosure (Department of the Interior, 2014b). There is only one significant spring that flows into a large tank for bison and several smaller seeps and springs, which are located in the northwest portion of the Badlands Wilderness Area (Park Ecologist, Personal Comm.). There are several water-holding structures that are scattered in other parts of the bison range, though many run dry during drought. The recommended herd size is 600–700 bison (Department of the Interior, 2014b). Currently, an environmental assessment is ongoing at BADL to consider expanding the bison range by more than 20,000 acres.

Additionally, park managers partner with the Intertribal Buffalo Council (ITBC) and Oglala Sioux Parks and Recreation Authority (OSPRA) to coordinate bison round-ups.[1] However, logistical, funding, and institutional constraints can at times limit the transfer of surplus animals to maintain recommended grazer densities (McNeeley et al., 2016).

[1] During the interviews, managers at Wind Cave National Park mentioned that as of 2013, the park had signed an agreement with The Nature Conservancy (TNC) to deliver all surplus bison from the park to TNC and their partners.

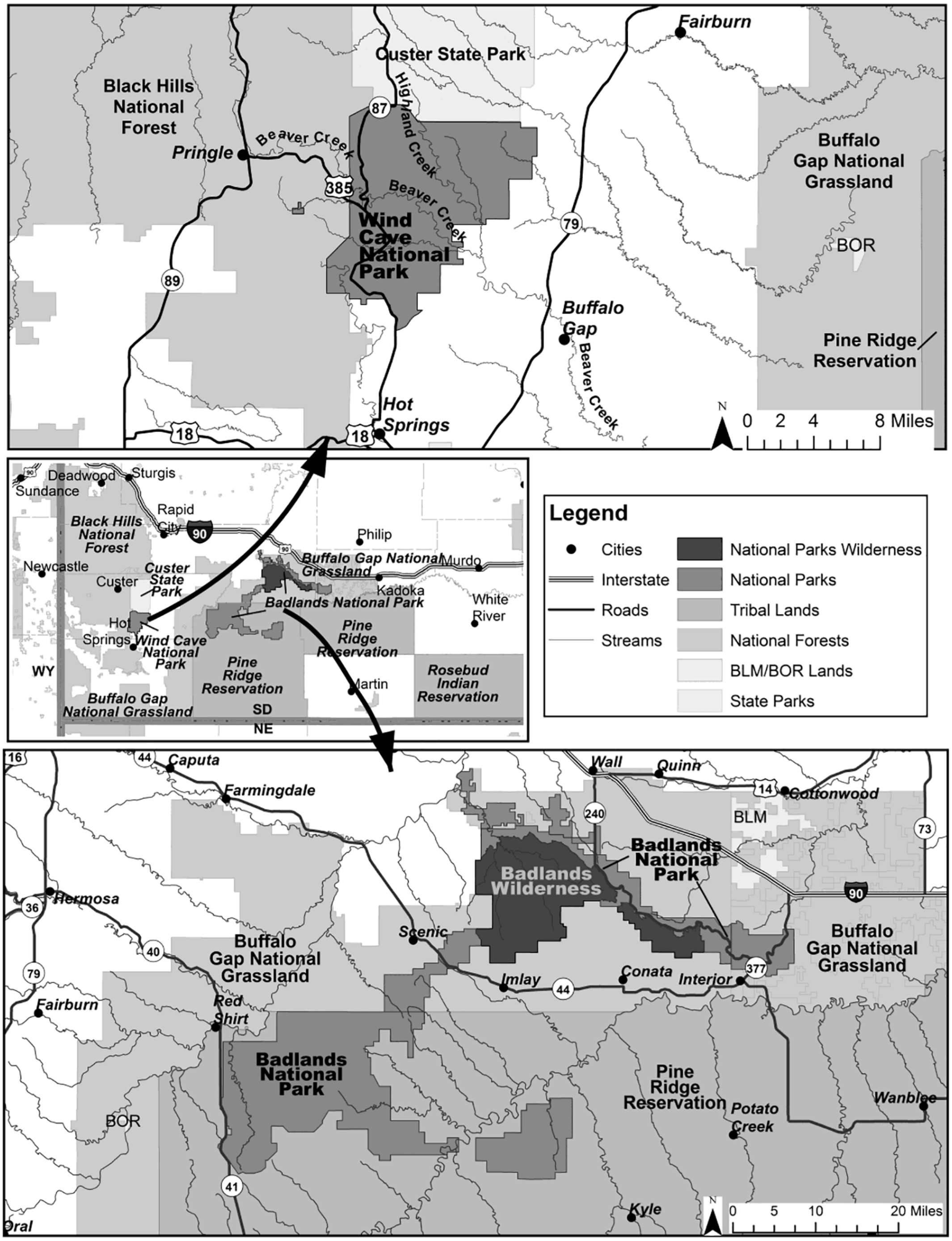

Fig. 1. Maps of southwest South Dakota case study.

4. Methodological approach

4.1. Case study data collection and analysis

The case study design and analysis was framed by a determinants and analogue approach to vulnerability assessment (DAVA). Vulnerability and adaptive capacity are dynamic and ever-changing, i.e., the vulnerability of local communities/groups is context-, place-, and time-dependent (O'Brien et al., 2007; Smit and Wandel, 2006). As such, in a determinants approach, the causal factors that determine local vulnerability and response capacities are identified in close consultation with local land and resource managers

(Füssel and Klein, 2006; McNeeley, 2014; Smit and Pilifosova, 2003; Smit and Wandel, 2006). An analogue approach documents local drivers, impacts, and responses to past drought, and those barriers that inhibited past drought response as an analogue for plausible impacts and response capacities in the future (Ford et al., 2010; Glantz, 1996, 1991; McNeeley and Shulski, 2011).

We conducted semi-structured key informant interviews (n = 11) with land and resource managers[2] in public lands managed by the DOI and tribal lands in southwest South Dakota. Interviews focused on drought risk and responses (Appendix A). Questions pertinent to this DAVA approach considered: the timing and seasonality of drought; severe drought years and impacts/responses; indicators, triggers, and information sources for drought monitoring and assessment; and the capacities and barriers to drought response. Data were collected using purposive sampling, which is a non-random sampling technique used to sample informants who are knowledgeable about the subject (Bernard, 2006). Key informants included managers from the NPS, Natural Resources Conservation Service (NRCS), ITBC, OSPRA, Rosebud Reservation/Sinte Gleska University (SGU), and Pine Ridge Reservation.

We followed a grounded theory method to analyze interviews. Grounded theory is an iterative discovery process, which consists of coding, memo-writing, and constant comparison of similarities and differences within and between informants (Corbin and Strauss, 2008; Glaser and Strauss, 1967). Each transcript was read in its entirety, during which open and axial codes were iteratively applied to text segments by one coder. Open coding is used to summarize the information in, and meaning of, isolated segments of text, while axial coding is the process of relating concepts to one another and creating higher-order categories (Corbin and Strauss, 2008). We used several analytical functions in Atlas.ti, a qualitative software program, to analyze transcripts. Code groundedness (i.e., the number of times a code occurs) was used to identify key management targets and issues (e.g., bison management, drought years of importance), while co-occurrence (i.e., when two codes occur in the same text segment), complex code queries, and network analyses were used to identify the social, ecological, and climate factors related to bison management. We relied on knowledge and observations of local managers to inform our results.

4.2. Linking case study results to a simulation modeling framework

We proposed a conceptual joint agent-based/state-and-transition simulation modeling framework to best address the key components and interactions across scales that were identified in the empirical case study results. Agent-based models (ABMs) and state-and-transition simulation models (STSMs) are two types of models used to simulate social-ecological phenomena. ABMs are composed of autonomous agents that represent individual or collective entities (e.g., people, wildlife), which interact with one another and the environment (generally a gridded landscape that can be static or dynamic) according to a set of rules. ABMs can represent decision-making, individual-level variation, and mobility, for example (Berry et al., 2002, see special issue). STSMs are stochastic models that use discrete timesteps and spatial cells to represent landscape dynamics. During each timestep, each simulation cell is classified according to a set of vegetation states that can change according to a set of transition pathways (Daniel et al., 2016). The likelihood of each transition depends on state class, age, and time since previous transition, and can vary spatially and temporally. STSMs project changes in the landscape (state classes and their attributes, such as biomass) over time, including ranges of uncertainty. STSMs can integrate diverse data sources, and represent a variety of ecosystems, processes, and management alternatives.

Both ABMs and STSMs can conveniently explore "what-if" scenarios and trade-offs in management decisions before taking costly or detrimental actions (Axelrod, 1997; Erlien et al., 2006; Peck, 2004), as well as possible climate futures and uncertainties in climate projections (Miller and Morisette, 2014). Together, ABMs and STSMs offer unique strengths in simulating social-ecological systems, and tools to dynamically link them are being developed (Miller and Frid, Unpublished results).

For this paper, we first developed a conceptual model of the joint ABM/STSM modeling framework to illustrate key variables and interactions pertinent to the bison management case study. We then focus our results on how the rich, qualitative narratives informed by a DAVA approach and grounded theory can help to: 1) prioritize model components included in modeling efforts; 2) frame joint management and climate scenarios; and 3) parameterize and evaluate model performance. Case study results were supplemented with a review of the simulation modeling literature to identify relevant model approaches and design concepts, along with empirical studies from the region that document bison ecology and interactions with other components germane to this study. This helped to identify the types of models, design concepts, and data available to address major components, processes, and interactions identified through the case study analysis, while also identifying gaps and needs moving forward.

5. Results

5.1. Case study results

We organize the case study results around three themes related to bison management: a) identifying key processes, entities, scales, and interactions; b) documenting local impacts, indicators, and monitoring efforts; and c) determining management tradeoffs and uncertainties.

[2] We recognize the term "manager" can have specific meanings in agency lexicon. For the purposes of this study, we use the term to describe individuals who have a role in land or natural resource management (e.g., technicians, biologists, ecologists, etc.), and use "informants" interchangeably with "managers."

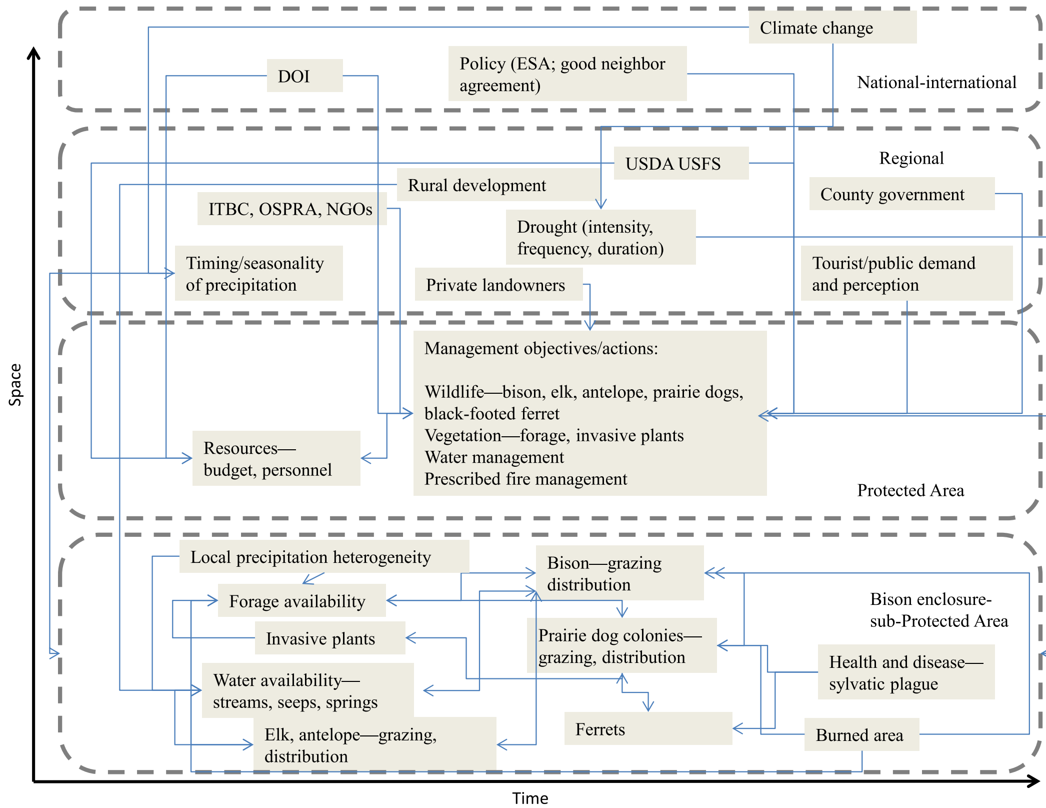

Fig. 2. Conceptual diagram of the entities, processes, and interactions within and across spatio-temporal scales that impact bison management in southwest South Dakota. Adapted from Cumming et al. (2015). Abbreviations: DOI = Department of the Interior; ESA = Endangered Species Act; USDA = US Department of Agriculture; USFS = US Forest Service; ITBC = Intertribal Buffalo Council; OSPRA = Oglala Sioux Parks and Recreation Authority; NGO = non-governmental organization.

5.1.1. Identifying key processes, entities, scales, and interactions

A host of entities, processes, and interactions in social-ecological components within and across scales must be considered for bison management (Fig. 2). In the bison enclosure, bison mobility and distribution impacts, and is impacted by, forage and water availability. Although grazers, such as antelope and elk, can leave the fenced enclosures during drought, which many informants observed during past droughts, these grazers can compete for forage and water. At the PA level, limited funding, time, and personnel were prominent factors in the ability of managers to set management targets, priorities, and objectives for bison and other wildlife, vegetation, and fire management (Fig. 2). These institutional resources were considered a primary barrier to current management goals and long-term drought planning. The constraints to manage for multiple objectives at the PA level were further embedded in regional and international factors. For instance, grazer densities were defined according to forage allocation models described in management plans, but were largely dependent on personnel and funding to coordinate round-ups, as well as agreements between NPS park managers and tribal/non-governmental institutions regarding the number of bison to surplus each year. Also, increased population and housing development northwest of WICA impacted surface water availability inside the bison enclosure during normal and dry years (McNeeley et al., 2016).

Prescribed fire management is a tool that can maintain suitable habitat for grazers as bison discriminately graze in recently burned areas (Fig. 2). However, prescribed fire management was constrained by the interactions between drought, inter-agency partnerships, county government decisions, and mismatches in the allocation of funds and changing ecological conditions (Fig. 2). During drought, managers were constrained in their ability to conduct prescribed burns for several reasons, one of which is that burning during drought can impact forage productivity in subsequent years. Additionally, WICA and BADL are part of the Great Plains Interagency Dispatch Center, which is a partnership that supports wildfire and prescribed fire management on public lands throughout South Dakota and Nebraska. Local managers were sometimes unable to conduct prescribed burns because personnel were unavailable. Further, a wildland fire coordinator at BADL described how county-wide burn moratoriums during drought can impact decisions to do a prescribed burn. Although the park office need not abide by county bans, they often do so to maintain good neighbor agreements. In 2012, WICA procured US Forest Service funds to conduct a prescribed burn. However, conditions were too dry in the fall for a burn, and by the time managers were ready to burn the fiscal year ended and the money was no longer available to them. This highlights an important mismatch in temporal scales; funding streams for prescribed fire management did not coincide with changing conditions on the ground.

Bison also depend on preferred grazing habitats at the edge of prairie dog towns. Informants reported that prairie dog colonies ebb and flow as a function of temperature and precipitation, grazing intensity, and fire behavior, and in doing so the location and extent of prairie dog colonies can directly impact forage availability for bison and other grazers, and can impact bison distribution (Fig. 2). Prairie dogs create early seral[3] vegetation conditions, and as they expand their range in search of forage, prairie dogs create areas with bare ground prone to wind erosion during drought. Managing for early seral vegetation was described as difficult, because on one hand it is a result of healthy prairie dog populations and it supports a variety of wildlife dependent on that vegetation, including for instance the endangered black footed ferret that managers are federally mandated to protect. On the other hand, it is susceptible to non-native plant colonization, which affects habitat quality, forage availability, and wildlife populations, including bison (Fig. 2).

Informants described that managing both invasive plants and native prairie dogs is complicated due to diverse, and often conflicting, management objectives and priorities of adjacent landholders and the broader public (Fig. 2). For instance, the presence of prairie dogs is considered an indicator of bad land management and early seral vegetation is considered undesirable by some ranchers on the Buffalo Gap National Grassland adjacent to BADL. Therefore, WICA and BADL both uphold good neighbor agreements to control and treat prairie dog colonies and invasive plants so that they do not extend outside the park. Further, the relatively recent emergence of sylvatic plague caused by *Yersinia pestis* in prairie dog and black footed ferret populations in the region has resulted in local mortality in these populations (Fig. 2). This has created uncertainty regarding how drought and plague will interact to impact these populations and the habitats, communities, and species they support, and specifically the ways in which forage resources will be affected vis-à-vis local prairie dog colony persistence or mortality. Therefore, the combination of local climate, ecological processes, and social factors associated with prairie dog and invasive species management can impact the amount, quality, and distribution of forage for bison.

Managers also described the temporal and spatial patterns of climate and weather that drive forage production in the region, with relevant temporal scales ranging from seasonal to decadal. Locals reported a high degree of spatial heterogeneity in precipitation. Spring and summer thunderstorms are highly localized, and differences in the duration and intensity of storms have implications for forage productivity; slow-wetting events are more beneficial than shorter pulses of precipitation (Fig. 2). While winter precipitation was considered important, spring-early summer (April-June) precipitation was described as the primary driver of forage productivity, which is consistent with local studies (Smart et al., 2007). Fall conditions can impact prescribed fire management. A rangeland manager reported that annual primary productivity is a function of a temporal lag in precipitation; productivity depends on the current and previous year precipitation. Informants also described how differences in the frequency, intensity, and duration of past droughts, and recovery between droughts can result in differential impacts on forage and water availability (McNeeley et al., 2016).

5.1.2. Documenting local impacts, indicators, and monitoring efforts

Informants documented a host of local drought impacts, indicators, and monitoring efforts to assess drought progression and impacts, and to trigger decision-making. Local observations of impacts from droughts were framed mostly in the context of two previous drought periods; the persistent 2002–2007 drought and the more recent, short-lived but extreme 2012 drought which was accompanied by record high heat. For instance, reductions in forage quality and quantity during both the 2012 drought and the longer 2002–2007 drought led to significant impacts on forage availability, bison, and elk. Managers also observed loss of vegetation along streams leading to bare ground and soil erosion issues, partly because of ungulates congregating in these areas.

However, the 2002–2007 drought was described as more severe (McNeeley et al., 2016). Managers observed reductions in the reproductive capacity of bison and elk, which was attributed to reduced forage quality and quantity, and WICA staff made unprecedented inquiries about water rights and delivery in the bison enclosure. Additionally, informants from WICA reported the spread of the non-native plant White horehound (*Marrubium vulgare*) on prairie dog towns. Prairie dog towns are high-disturbance landscapes, and White horehound expanded from a very small acreage to over 800 acres by the end of the 2002–2007 drought. The invasion of White horehound caused soil erosion issues, and changed the spatial distribution and amount of quality forage and habitat for prairie dogs, bison, and elk. For instance, one manager observed that prairie dogs can't clip White horehound, which therefore forced those colonies to expand out in search of adequate forage and habitat. While some informants mentioned other invasive plant species of concern in the context of drought, such as Canada thistle (*Cirsium arvense)*, Smooth brome grass (*Bromus inermis*), Star thistle and knapweed (*Centaurea* sp.), among others, one grassland ecologist at BADL observed the emergence of the novel species, Spurge flax (*Thymelaea passerina*) in 2008, and Yellow sweet clover (*Melilotus officinalis*) in 2009/2010 following the persistent drought period and during a wetter moisture regime.

Fire dynamics and impacts to water resources were also reported. A fire coordinator at BADL reported that fire risk is higher during dry periods following wetter cycles, but as drought conditions persist, the conditions favorable to fire are reduced as a function of reduced fuel loads, especially in a grazed system. Indeed, a decrease in fire activity was observed during the 2002–2007 persistent drought at WICA. At BADL, dry conditions, coupled with socio-economic factors (e.g., funding constraints), also precluded the ability to conduct a prescribed burn in 2006 and 2012. At WICA, surface water from the perennial Beaver Creek retreated upstream to reduce water availability from the stream by approximately half in 2007.

Informants described key indicators, tools, and monitoring efforts that are being used to document drought impacts. For instance, they used broad-level climate summaries such as the US Drought Monitor. Yet, managers were more responsive to local climate,

[3] Seral stage refers to a developmental stage or status in an ecological succession. Early seral is the first stage of secondary successional development following disturbance. This stage is characterized by the absence or relatively low abundance of species that represent the potential natural community. It could also refer to a state where evidence of a life form layer is absent (Hall et al., 1995).

especially precipitation trends at critical points in the season, forage productivity, and animal condition and behavior as visual cues of landscape state. One informant used the South Dakota Drought Tool, which uses local precipitation data, weighted by spring and fall precipitation and by precipitation amounts in the previous and current year, to develop estimates of current productivity, projected "peak" productivity, and recommended grazing densities.[4] Also, managers described using bison mobility as visual cues of forage and water availability in the BADL bison enclosure. This included monitoring areas where bison congregate during drought, such as riparian areas and stock dams, and more specifically at a prairie dog town at the north end of the Badlands Wilderness (Fig. 1), known locally as Roberts. This 1000 acre prairie dog town has several dams that bison rely on when water is short.

Managers at WICA, BADL, and SGU all monitor forage requirements and grazer densities. Elk and bison numbers in park units were in some cases double the recommended densities. For instance, carrying capacity for elk is recommended at 232–475 animals at WICA, though one manager mentioned that the current population is around 900. Further, while recommended carrying capacity for bison at BADL is 600–700, managers estimated that numbers at the time of the interviews were 1000–1200 bison.

Key informants described three drought-specific long-term monitoring efforts. In response to past drought impacts, managers at WICA started monitoring vegetation annually to document impacts of drought and congregation of ungulates to riparian vegetation and forage productivity in rangelands. South Dakota State University manages the Cottonwood Range and Livestock Field Station northeast of BADL, where researchers have been conducting studies since the 1940 s to better understand the impacts of stocking density and drought on forage. SGU has used precipitation and temperature data over a 120-year historic period to relate spring green-up and greenness to drought.

5.1.3. Determining management trade-offs and uncertainties

Based on the results in 5.1.1 and 5.1.2, several locally-informed management concerns, trade-offs, and uncertainties were identified. These included:

1. Climate variability and change: What would be the impacts to forage and water if the persistent drought from 2002 to 2007 lasted for another couple years, or if conditions present during the abnormally hot and dry 2012 extreme drought carried on for just another year (McNeeley et al., 2016)? In this regard, the impacts to water and forage resources from the increased persistence and reduced recovery between droughts, and changes to the timing and seasonality of precipitation in the future were of concern.
2. Prescribed fire management: What would be the implications for biomass availability should prescribed fire management options be removed, or constrained, as a function of climate and socio-economic factors?
3. Prairie dogs and invasive plants: What is the combined effect of drought conditions, plague dynamics, and social factors (e.g., treatment/control) on the distribution of prairie dog colonies and invasive plants? And in turn, what are the cumulative impacts on forage amount and distribution?
4. Bison management
   a. Bison carrying capacity: What are the impacts on forage and water if bison round-ups do not occur due to institutional, funding, and/or personnel constraints? And, what are the added drought impacts to carrying capacity within the enclosure (McNeeley et al., 2016)?
   b. Range expansion: The development of additional rangeland could potentially circumvent the need to deliver surplus bison every year. Questions considered whether expanded range should be grazed year around, seasonally, or held in reserve for when needed, and how many additional animals could, or should, be supported.
5. Water management: What is right number and best locations to develop additional water infrastructure to better distribute bison, minimize impacts to forage, and provide ample water for grazers? What are the combined impacts of drought and domestic water extraction upstream on water resources inside bison enclosures?

5.2. Linking case study results to simulation modeling: A conceptual ABM/STSM modeling framework

The results presented in Section 5.1 demonstrated the complexity of bison management in southwest South Dakota. We identified a host of processes and entities that interact across scales to affect bison management, drought impacts, and several management tradeoffs and uncertainties to consider. Simulation models can represent multiple drivers of change across scales and scenarios, and therefore are an appropriate tool in which to address the factors that impact bison management.

Here, we introduce a joint ABM/STSM framework to address key factors that emerged from the case study results. Fig. 3 illustrates a conceptual model of the joint ABM/STSM. Bison movement, grazing and uptake, and demography are represented in this model using an ABM (Fig. 1; red boxes). Modeled bison (i.e., agents) interact in the environment as a function of surface and ground water availability (Fig. 3; blue boxes), as well as in response to available biomass, and the state classes associated with prairie dog colonies, fire, and invasive plants modeled using STSM (Fig. 3; green boxes). Inputs/outputs are shared within and between models (dashed and solid arrows, respectively), and each of the sub-models are impacted to varying degrees by climate and management scenarios. Key response variables of interest include forage condition, water availability, and bison populations. Modeling water availability would require a separate hydrological model coupled to this joint ABM/STSM modeling framework. For brevity, we limit our results to the ABM/STSM sub-models.

As with any model-building process, it is important to strike a balance between complexity and simplicity. To distill results

[4] The South Dakota Drought Tool can be accessed at: https://www.nrcs.usda.gov/wps/portal/nrcs/main/sd/technical/landuse/pasture/

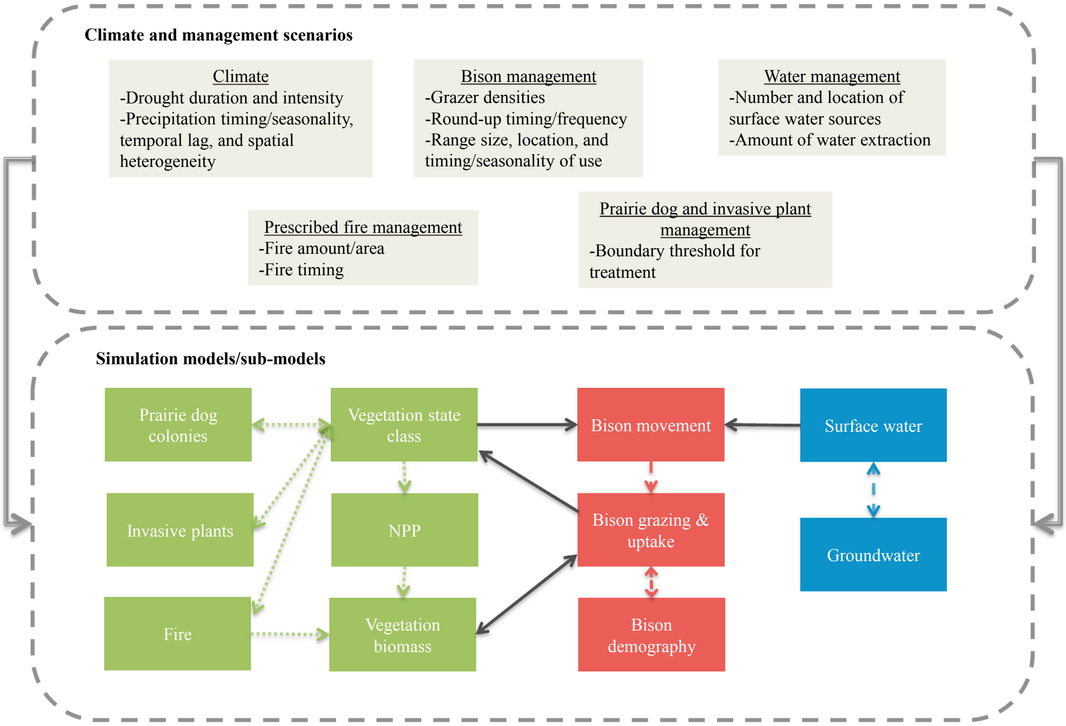

Fig. 3. Model structure including simulation models/sub-models and climate and management scenarios. Bison are modeled using ABM (red boxes), where bison interact with modeled vegetation dynamics (STSM; green boxes) and water availability (a separate hydrologic model; blue boxes). Inputs/outputs are shared within and between models (dashed and solid arrows, respectively). Climate and management scenarios feedback to affect interactions and outcomes within model structure. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 1
Components included in model structure. Proposed model structure includes several components from empirical qualitative results (Components listed in italics derived from Fig. 2) that are explicitly and implicitly modeled. Decisions were made to not represent some components to focus model structure on primary response variables and timescales of interest for this study (i.e., forage production, water availability, and bison population/carrying capacity) and to reduce model complexity.

Model inclusion	Components
Explicit	• *Bison* – demography, mobility, forage consumption • *Climate* – drought, timing/seasonality of precipitation, local precipitation heterogeneity • *Management objectives/actions* – prescribed fire, bison population management, water management, prairie dog and invasive plant management • *Forage availability* – biomass, net primary production • *Burned area* • *Prairie dog colonies* – vegetation state change, affected by climate, contraction and expansion (contagion) • *Invasive plants* – vegetation state change, affected by climate, spread (dispersal kernel) • *Water availability* – a separate hydrological model would be needed
Implicit	• *Elk, antelope, other grazers* – forage consumption • *Prairie dogs* – demography, forage consumption • *Private landowners, tourist/public demand and perception, Policy, good neighbor agreements* – influences prairie dog/invasive plant management as a boundary threshold for treatment • *Resources* (*budget, personnel*) – influences round-ups • Institutional arrangements (*ITBC, OSPRA, NGOs*) – influences round-up timing/amount • *Rural development* – domestic water extraction outside park • Institutional arrangements (*USDA USFS, County government, good neighbor agreements*) – influences prescribed fire management • *Climate change* – influences timing/seasonality of precipitation and drought regimes
Not included	• *Ferrets* – does not directly influence biomass/water availability for bison • *Health and disease-sylvatic plague* – can influence forage availability/composition indirectly through prairie dog populations but would necessitate additional agent class in ABM framework and increase complexity substantially; recent to area and little known regarding transmission rate, climate controls (e.g., Eads and Hoogland, 2017, 2016; Eads et al., 2016) • *DOI* management directives – influence *Resources*, and *Management objectives/actions*, which are implicitly/explicitly modeled

Table 2
ABM sub-models. Includes description of inputs from qualitative results that can help prioritize model components, and insights to help parameterize and evaluate model performance. Example data and models, and description of additional needs are examples based on informant interviews and researcher experience, and are not intended to be exhaustive lists.

Simulation sub-models	Inputs from qualitative data	Example data/models	Additional inputs/needs
Bison movement	• Interactions between forage availability, water availability (streams, stock ponds), preferred grazing areas (burned areas, prairie dog colonies) • Grazing timing (e.g., Roberts prairie dog town, riparian areas during drought)	• Modeling wildlife movement – (Boone and Galvin, 2014; Tang and Bennett, 2010; McLane et al., 2011; Dumont and Hill, 2004) • Bison, prairie dog, fire, vegetation interactions – (Coppock et al., 1983; Coppock and Detling, 1986; Krueger, 1986) • Geospatial layers of enclosures and vegetation – (Miller et al., 2017; Symstad et al., 2017)	• Bison movement driven by forage amount and/or composition?
Bison grazing & uptake	• Primary grazers offtake (e.g., bison, prairie dogs, antelope, elk) • Grazing timing (e.g., elk and antelope dispersal outside park)	• Bison/plant interactions – (Fahnestock and Detling, 2002)	• # antelope, elk, prairie dogs in study region/jurisdictions • Bison/wildlife daily forage and water consumption requirements
Bison demography	• Bison densities (e.g., recommended/current; alternating grazing regime) • Decreased reproductive capacity (2002–2007)	• (Wockner et al., 2015; Plumb et al., 2009)	• Cow/calf ratio • Min/max size • Reproductive rate

presented in Section 5.1 down to key model components and response variables of interest, Table 1 presents an example approach for implicitly representing or excluding different components. In the next sections, we describe each of the sub-models and scenarios with a focus specifically on how results from Section 5.1 informs model prioritization, scenario development, and model parametrization and evaluation. We also present pertinent modeling examples and design concepts, and empirical data on bison interactions in the region from the literature, along with additional questions and/or inputs that could prove useful moving forward. Example data and models, and description of additional needs are examples based on informant interviews and researcher experience, and are not intended to be exhaustive lists.

5.2.1. Bison movement, grazing, and demography in an ABM

Bison movement, grazing, and demography are represented in an ABM (Fig. 3; red boxes). Each of the ABM sub-models are described below, with a focus on how qualitative results from Section 5.1 inform the model framework (See also Table 2).

The bison movement sub-model is dictated by forage and water availability, as well as preferred vegetation state classes associated with prairie dog colonies and burned areas (Table 2; Fig. 3). Bison movement is also dictated by weather conditions, i.e., bison move to preferred locations (e.g., Roberts prairie dog town) and riparian areas during drought. Modeling wildlife movement in an ABM context has several advantages. ABMs are built around design concepts such as learning, adaptation, and sensing (Grimm et al., 2010). Simulated bison could learn to remember where water and key forage resources are located, adapt to changing conditions in the distribution of preferred grazing habitats or water availability by sensing neighboring locations (i.e., patches or cells) with higher productivity, and alter grazing regime in response to thirst (Boone and Galvin, 2014; Dumont and Hill, 2004; McLane et al., 2011; Tang and Bennett, 2010). Several empirical studies have explored bison, prairie dog colony, fire, and vegetation interactions, for example the timing and use of vegetation by bison near prairie dog colonies (e.g., Coppock et al., 1983; Coppock and Detling, 1986; Krueger, 1986), and others have developed geospatial layers of vegetation types within bison enclosures in the region (Miller et al., 2017; Symstad et al., 2017), both of which could help initialize and parameterize the model. Further information on whether forage amount or composition drives bison movement would be needed, for example.

The bison grazing and uptake sub-model considers forage use (or “offtake”) from bison, but also forage consumption by prairie dogs, antelope, and elk, among others (Table 2). Fahnestock and Detling (2002) provided empirical evidence of plant-bison-prairie dog interactions, and specifically provided estimates of aboveground biomass on sites grazed by bison and sites not grazed by bison. Further, while bison graze year-around, antelope and elk can, and did, disperse out of enclosures in search of forage during drought, therefore the proportional offtake from grazers would be augmented contingent upon weather conditions. This can be done using a conditional statement in NetLogo, an agent-based programming language and modeling environment. Estimates of grazer densities across the study region, and accurate estimates of daily forage and water requirements for each would be needed.

In the bison demography sub-model, the recommended and current grazer densities of bison and alternating grazing regimes identified by locals provide more realistic parameters for forage requirements and offtake (Table 2), and provide bounding conditions for scenarios (Section 5.2.3). The observed reduction in bison reproductive capacity during the persistent 2002–2007 drought could provide a qualitative evaluation of model performance. Additional needs to consider are precise estimates of the cow/calf ratio, minimum and maximum size, and reproductive rate of bison within enclosures to more accurately determine forage and water requirements, and carrying capacity (Plumb et al., 2009; Wockner et al., 2015).

Table 3

STSM sub-models. Includes description of inputs from qualitative results that can help prioritize model components, and insights to help parameterize and evaluate model performance. Example data and models, and description of additional needs are examples based on informant interviews and researcher experience, and are not intended to be exhaustive lists. Abbreviations: CRLFS = Cottonwood Range and Livestock Field Station; SDDT = South Dakota Drought Tool.

Simulation sub-models	Inputs from qualitative data	Example data/models	Additional inputs/needs
NPP & biomass	• Monitoring efforts (e.g., CRLFS) • Tools (e.g., SDDT) relating biomass production to climate, grazing	• Stock/flow module – (Daniel et al., 2017a; Miller et al., 2017) • Vegetation response to climate – (Smart et al., 2007) • SDDT – https://www.nrcs.usda.gov/wps/portal/nrcs/main/sd/technical/landuse/pasture/	• Vegetation response to novel climatic conditions
Invasive plants	• Species of concern (*Marrubium vulgare, Cirsium arvense, Thymelaea passerina, Melilotus officinalis, Bromus inermis, Centaurea* sp.) • State change: *Marrubium vulgare* invasion in early seral during 2002–2007 drought; increase in *Thymelaea passerina* in 2008; increase in *Melilotus officinalis* in 2009	• Dispersal kernels – (Frid et al., 2013; Frid and Wilmshurst, 2009; Miller et al., 2017)	• Habitat suitability and spread of invasive plants • Palatability of invasive plants
Prairie dog colonies	• Dispersal a function of vegetation state and composition, grazing intensity, climate, fire • Expansion during 2002–2007 drought	• Geospatial layer of prairie dog colonies for initialization – (Miller et al., 2017) • Dispersal – (Cincotta et al., 1987; Garrett and Franklin, 1988; Dalsted et al., 1981; Hoogland, 2001) • Vegetation state and composition – (Krueger, 1986; Archer et al., 1987)	• Rate of expansion during drought
Fire	• Fire dynamics under persistent drought (2002–2007) • Dry/wet cycles	• Vegetation impacts – (Miller et al., 2015; Daniel et al., 2017b; Halofsky et al., 2013; Blankenship et al., 2015; Costanza et al. 2015)	• Area burned in drought, normal years

5.2.2. Characterizing vegetation state and transitions in STSM: Interactions between bison, prairie dog colonies, invasive plants, and fire

Biomass, and the state classes associated with prairie dog colonies, fire, and invasive plants are represented using STSM (Fig. 3; green boxes). We describe each of the STSM sub-models, and focus our results on how the qualitative results from Section 5.1 inform the model framework (See also Table 3).

The NPP and biomass sub-model is affected by grazing, climate, and changes in vegetation state class associated with fire, invasive plants, and prairie dog colonies (Fig. 3). In this context, it is possible to estimate above-ground biomass using the stock and flow module in ST-Sim, a software tool for STSM, which offers a way to track the pools or stocks of continuous state variables (e.g., biomass, litter) and the flows or changes in those stocks (e.g., primary production, decomposition, grazing, fire) over space and time (Table 3; Daniel et al., 2017a; Miller et al., 2017). Ongoing monitoring and assessment efforts at the Cottonwood Range and Livestock Field Station, and South Dakota Drought Tool, for example, could help to parameterize or validate the stock and flow module, and to relate biomass production to grazing pressure and climate (Table 3). Further, STSMs can apply the design concept of memory, such that biomass is estimated as a function of accumulated precipitation during the previous year and during key seasons, consistent with local studies and observations reported here (Smart et al., 2007). While the relationship between forage productivity and climate is relatively well understood, response to novel, future conditions is less understood, which therefore warrants additional consideration (Table 3).

Invasive plants can create species and state class changes, which in turn affects forage availability for bison and other grazers (Fig. 3). In STSMs, the spread of invasive plant species can be modeled using dispersal kernels (e.g., Frid et al., 2013; Frid and Wilmshurst, 2009; Miller et al., 2017). Local observations of invasive plants that were of key concern are useful for prioritization (Table 3). Observations of local impacts, including the spread of White horehound during the 2002–2007 drought, and an increase in Spurge flax following the persistent drought (Table 3), provide a starting point for understanding the climatic controls on non-native invasive plants to parameterize and evaluate the model. Miller et al. (2017) developed an STSM to simulate Canada thistle invasion under a variety of management and climate alternatives in BADL and surrounding region. Yet, additional work is needed to simulate the habitat suitability and dispersal potential of other species of concern identified here. Further, additional work is needed to determine the palatability of invasive plant species for bison.

Prairie dog colonies directly impact, and are impacted by, vegetation state class and composition (Fig. 3). Colonies could be represented in an STSM using spatial contagion, where colonies (or more precisely the state classes that colonies create) expand and contract as a function of vegetation state and composition. Miller et al. (2017) developed a geospatial layer of prairie dog colony locations that would be useful to set initial conditions, while the observed expansion of prairie dog colonies during the 2002–2007 persistent drought could be useful for model evaluation (Table 3). Additional information on prairie dog dispersal as a function of climatic and non-climatic factors (e.g., Cincotta et al., 1987; Dalsted et al., 1981; Derner et al., 2006; Garrett and Franklin, 1988; Hoogland, 2001), rate of expansion during drought, and effects of prairie dog colonies/grazing on vegetation state and composition

Table 4
Climate and management scenarios. Qualitative narrative/results framed the scenarios and parameters to vary based on key issues, timescales, and interactions of importance to managers.

Climate & management scenarios	Qualitative narrative	Parameter variation
Climate	• Variation in drought onset, frequency, persistence, and recovery differentially impact resources • Timing and seasonality, temporal lag, and spatial heterogeneity in precipitation are critical inputs to consider	• Historical record • Quick onset drought • Persistent drought • Recovery period • Proportion of precipitation falling during spring vs winter
Bison management	• Grazer densities are above recommended carrying capacity • Round-ups occur, but can be constrained by funding and institutional constraints • Range expansion considered – Questions considered grazing timing, whether range should be held in reserve for drought, and whether to increase herd size or maintain current densities	• Grazer densities – recommended, current, high • Round-ups – vary frequency of occurrence • Range expansion – grazing year-round, seasonal, or as reserve for drought
Water management	• Additional water development to provide water for bison and better distribute bison to minimize impacts to forage • Consider the impact of residential water extraction on water availability	• Vary number and location of wells/dams in enclosure • Water extraction – pre-, post-, and projected development
Prescribed fire management	• Concerned about removing prescribed fire completely from management options, which are limited by funding and institutional arrangements • The ability to conduct prescribed fire contingent upon moisture conditions in spring and fall	• Baseline – % of park area/enclosure annually • Reduced/no prescribed fire • No prescribed fire under drought (conditional)
Prairie dog and invasive plant management	• Good neighbor agreements stipulate that both be managed so as to not extend outside of park boundaries and on to adjacent jurisdictions	• Boundary threshold – treatment occurs when prairie dogs and invasive plants extend outside park boundaries

(e.g., Archer et al., 1987; Fahnestock and Detling, 2002; Krueger, 1986) would further support model development.

The fire sub-model directly and indirectly impacts biomass availability and distribution for bison. Several studies have simulated fire dynamics and impacts to vegetation using STSMs (Blankenship et al., 2015; Costanza et al., 2015; Daniel et al., 2017b; Halofsky et al., 2013; Miller et al., 2015). Fire dynamics were only briefly discussed by local managers, specifically with regard to decreased fire during the most recent persistent drought period, and the ebb and flow of fire with respect to wet and dry cycles, both of which could be useful to evaluate fire behavior within the model (Table 3). However, additional needs are more precise estimates of the area burned in historical fire events during normal and drought years, for example.

5.2.3. *Climate and management scenario development*

Below, we describe five scenarios relevant to the case study results presented in 5.1.3 (Fig. 3). These climate and management scenarios vary and affect sub-models explained in Sections 5.2.1 and 5.2.2. We highlight each of the scenarios, with a focus on the qualitative narrative that framed each scenario, how parameters are varied in light of these insights, and data and modeling examples to address these scenarios (See also Table 4).

5.2.3.1. Climate. Climate scenarios vary and affect vegetation production and composition, fire frequency and intensity, and surface water availability. Local informants reported that differences in the onset of drought frequency, severity, persistence, and recovery all interact to differentially impact forage and water resources on the ground (Table 4). Therefore, relevant drought regime scenarios include a historical (control) run, coupled with combinations of quick onset drought (e.g., 2012) and persistent drought scenarios (e.g., 2002–2007). The observed temporal lag in forage production from accumulated precipitation necessitates exploration of different recovery period intervals following drought. Further, concerns regarding changes to the timing and seasonality of precipitation could be explored by altering the proportion of precipitation that falls during winter versus spring.

Precipitation amount and duration was described as spatially heterogeneous (Table 4). Therefore, local precipitation gauge data or gridded estimates of historical precipitation (e.g., gridMet; Abatzoglou, 2013), and downscaled future conditions (e.g., MACA Abatzoglou and Brown, 2012) could be used to drive modeled vegetation dynamics, for example. Both ABM and STSM models can easily incorporate stochasticity (i.e., randomness, uncertainty) for modeling complex climate processes (Daniel et al., 2016; Railsback and Grimm, 2012). For instance, precipitation effects on forage production can be randomized across patches using probability distribution functions embedded in simulation platforms (e.g., NetLogo or ST-Sim). In this sense, biomass (modeled using the stock and flow module) would be driven by the effects of accumulated precipitation in each cell during key seasons and previous years on net primary production.

5.2.3.2. Bison management. Bison management scenarios vary grazer densities, round-up timing, range size, and grazing timing (Fig. 3; Table 4). In this regard, scenarios explore informant-defined recommended and current grazer densities as optimum/high values, and when and how often bison are round-up. Licht (2017) provided empirical data on the number of bison in WICA before round-up and the number of surplus animals each year from 1914 to 2013, which could frame these scenarios, though similar estimates from other places are needed.

Range expansion was considered to buffer uncertain socio-political and climatic environments, yet questions emerged regarding grazing timing in expanded ranges (e.g., year-around, seasonal, reserve), and whether additional bison should be supported (Table 4). Simple rules in an ABM modeling environment that determine when and under what circumstances expanded range areas are open to grazing could provide a test of these scenarios.

5.2.3.3. Water management. Managers were concerned about the need to develop additional stock dams for bison and to better distribute bison within enclosures (Table 4). Simple rules can be used to vary the number and location of stock dams within enclosures at initialization, which would then dictate bison movement (Boone and Galvin, 2014). While local managers provided estimates of the number and location of developed and natural springs and streams within park office boundaries, there is a need for water layers of the study region.

Further, the impact of drought and rural water extraction upstream on water resources was a major concern (Table 4). Examining water use by households upstream of bison enclosures (through the USGS National Water Use data for example) could help isolate effects of upstream water use on downstream availability from drought impacts. The Rainfall-Response Aquifer and Watershed Flow Model (RRAWFLOW; Long, 2015) was used to simulate the impacts of climate change on streamflow at WICA (Symstad et al., 2014, 2017). Yet, the model did not incorporate rural population growth and was limited to one perennial stream in the park. Future work is needed to modify RRAWFLOW to accommodate different inputs (e.g., water use), and to define pre-, post-, and projected rural population development levels with local managers to parameterize the model.

5.2.3.4. Prescribed fire management. Prescribed fire management was limited by drought conditions, and managers were concerned about the impacts to vegetation composition and forage resources should prescribed fire management be removed as a management option (Table 4). Therefore, useful parameters to vary in prescribed fire management include a baseline scenario where a percentage of park area is burned each year (Miller et al., 2017), a scenario in which prescribed fire is removed from management options completely (Symstad et al., 2017), and one in which prescribed fire is contingent upon fall and spring precipitation. The latter scenario could be explored in an STSM by calculating the area of prescribed fire applied each year based on a timeseries of precipitation, and then specifying the area burned as time-varying transition targets within ST-Sim.

5.2.3.5. Prairie dog colony and invasive species management. Good neighbor agreements between WICA and BADL and adjacent jurisdictions stipulate that prairie dog colonies and invasive plants be controlled from streaming out of park boundaries, which can impact the spatial distribution and amount of forage for bison (Table 4). STSMs and ABMs are spatially-explicit, in that they use gridded landscapes made up of patches with unique coordinates, which make it possible to set a boundary threshold for dictating when and where to treat invasive plants and prairie dog colonies within the modeled landscape (e.g., Miller et al., 2010).

6. Discussion and conclusions

In this study, we explored how a determinants and analogue vulnerability assessment (DAVA) approach, which was informed by key informant interviews with local DOI and tribal resource managers and grounded theory can inform simulation models that account for multiple, complex, and uncertain drivers of change. Our results illustrated several unique advantages of this approach regarding the case study we presented, and to the co-production of knowledge more broadly. First, these results helped to determine the appropriate modeling framework to address the complex and uncertain factors that impact bison management. Local seasonal and inter-annual climate and drought; fire and prescribed fire management; and prairie dogs, invasive plants, and their management combined to impact the spatial distribution and availability of forage for bison. Increased water use from rural population development and drought combined to impact surface water availability for bison. Cross-scale institutional arrangements were a climate risk multiplier. For instance, arrangements between NPS, tribal organizations, and NGOs can, at times, affect bison round-up timing and amount, which affects timely response and the ability to manage bison under recommended grazer densities, and therefore can increase drought vulnerability.

Therefore, we developed a conceptual joint ABM/STSM modeling framework that combined a spatially-explicit STSM of vegetation dynamics with an ABM that more realistically modeled bison grazing and population dynamics under different and locally-defined management and climate scenarios. The modular approach we presented provides a level of flexibility for exploring scenarios and trade-offs in isolation; for example, the impacts of grazing on biomass. ABMs and STSMs can isolate the relative importance of each of these processes through sensitivity analysis. At the same time, simulating the combined effects of multiple interacting variables can identify surprising, emergent outcomes, and important thresholds (Epstein, 2008), which may not be evident from sensitivity analysis or qualitative scenario exploration (Symstad et al., 2017). In many cases, and as we have shown here, the exploration of scenarios can be done by small modifications to rules or parameters in ABM or STSM modeling frameworks (Daniel et al., 2016; Railsback and Grimm, 2012).

Second, the DAVA relied on local knowledges and observations to: determine the social-ecological factors that underpin vulnerability to drought and climate change; document local drought indicators, triggers, and monitoring efforts; and identify past drought impacts and experiences, which were used as analogues to gauge future sensitivities and response capacities. Together, this helped to prioritize the components included in the model framework, frame scenario development, and parameterize and evaluate model performance. The key issues identified through our empirical analysis align with other studies in the region that employed qualitative scenario planning methods (Miller et al., 2017; Symstad et al., 2014, 2017). Our results also documented additional management issues and important sources of information to consider in this context. This included information on: bison grazing, movement, and population dynamics; appropriate spatial and temporal scales of analysis for drought management; locally relevant monitoring and assessment activities; and the social-ecological contexts surrounding prescribed fire and prairie dog management, for example.

Third, our results highlighted relevant data inputs, design concepts, and models that would be useful options to consider in modelling efforts. In the same vein, our results helped highlight data gaps, uncertainties, and needs moving forward. Indeed, these results advance the development of usable models that can help local managers better prepare for, and respond to, drought and climate change. We suggest the need for frameworks that focus on the co-production of science with local resource managers and scientists, with the goal of better linking institution and actor relationships within and across scales, and further refining "top-down" data and modeling efforts through "bottom-up" place-based understanding and elicitation of local knowledges and observations (McNeeley et al., 2017).

Fourth, the approach we used is a systematic, rigorous, and repeatable method to document and analyze local knowledge. For instance, the coding, memoing, and constant comparison across informants in a grounded theory analysis provides a database of information. The database can be interrogated in new and concerted ways, and expanded upon, as new questions emerge from researchers and managers. In turn, this provides the context to diagnose barriers to adaptation, determine shared understandings and goals, and start to move towards holistic and "clumsy solutions" to climate change (McNeeley and Lazrus, 2014; Verweij et al., 2006).

Yet, there are several limitations to both qualitative case studies and simulation modeling that need to be considered. First, qualitative case studies and co-production efforts more broadly require large investments in time from both researchers and local managers, which can lead to stakeholder fatigue, for example (Fischer et al., 2013). Second, simulation modeling and qualitative analyses require substantial expertise that many managers do not have, and may not have time to acquire. Therefore, there is a need for experienced individuals capable of translating complex modeling and analysis to local managers. Finally, data availability for parametrization and evaluation remains a major challenge to simulation modeling (Bennett et al., 2013; Verburg et al., 2016). Not all processes, entities, and interactions can be modeled, which can lead to some managers finding them unusable in a tangible sense. It is critical, then, to explain the assumptions and caveats inherent in any model-building process at the start to ensure the science that is produced is useful.

In sum, simulation models provide flexible platforms to represent complex social-ecological system dynamics and explore the effects of different scenarios. We explored how a social-science driven DAVA approach can help characterize what should be included in a modeling effort, what managers hope to learn from modeling, and ensure that relevant data is used at scales that are meaningful to managers' decision-contexts. In this sense, qualitative case studies can ground simulation models in local places and help make them more structurally realistic and useful, which is a step toward development of actionable climate change adaptation strategies. While this was illustrated through a case study of bison management in southwest South Dakota, we feel that the principles of this approach can be applied to other complex natural resource management contexts.

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Appendix A:. Interview questions protocol

1) How do you define or think about drought in the context of your landscape?
2) Do you view drought as a significant risk to your management activities?
3) [if yes to #2] At what time of year is drought most problematic (how/why) [this is getting at seasonality/timing issues)?
4) What year (or years) was the worst drought in this area? What happened?
5) What management decisions do you have to make that are affected by drought?
6) a. What, if any, indicators do you use to know if/when/how drought is going to cause negative impacts on your landscape?
   b. What do you consider to be the best source or sources of information on drought?
7) Are there fish, wildlife, and/or plant species you haven't mentioned impacted by drought in your landscape?
8) a. Are there human livelihoods or other activities impacted by drought in your landscape?
   b. Does this cause any conflicts?
   c. Do you collaborate with other stakeholders or jurisdictions on drought-related issues? If so, with whom and how?
9) Do you have the capacity to either respond to or prepare for drought?

10) Are there barriers that inhibit your ability to respond to or prepare for drought?
11) Anything else we haven't discussed?

Appendix B. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.crm.2018.09.002.

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