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SYNTHESIS & INTEGRATION

National-scale assessment of ecological content in the world's largest land management framework

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Abstract. Meeting the diverse sustainability targets of modern society has led to the development of national-level management frameworks meant to guide resource management actions and conservation funding decisions. In U.S. rangelands, state-and-transition models have been developed within the Ecological Site Description (ESD) Database as an application of alternative state theory and to move the discipline toward a more dynamic platform for resource management. After 15 years of development, and with government-mandated collaboration among federal agencies, these models are set to become one of the world's largest guiding frameworks for terrestrial ecosystem management. Yet, ESD state-and-transition models are being marketed for broad-scale application without a national-level critique evaluating their strengths and limitations. In this article, we conduct a national assessment of ESDs with a central focus on evaluating the specific details of ESD state-and-transition models. Importantly, we are not evaluating the conceptual underpinnings of the state-and-transition management framework, but rather its application. Specifically, we (1) quantify and summarize the information presented in ESD state-and-transition models; (2) determine whether ESDs fully meet U.S. Congress's goal of a nationally consistent system for defining, mapping, and interpreting ecological sites; (3) identify limitations and logical holes in ESD predictions; and (4) evaluate whether conservation funding priorities are consistent with output from ESDs. Our evaluation reveals multiple shortcomings in the application of the state-and-transition model concept within ESDs, primarily that they are highly subjective, inconsistent in design and application, focus on a single historical climax community, and overuse grazing as a driver of both ecological degradation and restoration. Considering that many of these limitations have been a consistent criticism of rangeland assessment procedures throughout the history of the discipline, state-and-transition models within ESDs will require major reconstruction beyond the current plans for revision if they are to meet society's demand for more effective management and utilization of rangeland resources. While ESDs were developed to link science and management in rangeland ecology, our assessment suggests well-intentioned management frameworks built upon expert opinion and qualitative inputs will not effectively shift ecosystem management from long-held practices rooted in community climax theory to modern scientific perspectives based on alternative state theory.

Key words: alternative state; climate change; community climax; conservation funding; ecosystem models; equilibrium; historical plant community; natural resource policy; rangeland assessment; succession; state-and-transition model; threshold.

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Introduction

The global human population has placed greater demands on the natural environment in the latter half of the 20th century than at any other point in human history, resulting in worldwide degradation of the majority of the world's ecosystems (MA 2005). This trend is expected to significantly worsen, necessitating the development of broad-scale management frameworks that improve the conservation and sustainability of the resources afforded to humans by nature. Scientists and government officials expect these management frameworks to be built upon recent scientific assessments of the consequences of broad-scale ecosystem transformations and to make decisions based on the evolving state of scientific knowledge (MA 2005, IPCC 2007, Briske 2011). In ecology, theoretical advancements have led to a major disciplinary shift from a climax-based perspective of tightly coupled, internally regulated ecosystems (e.g., Clement's climatic climax theory, 1916; Tansley's polyclimax theory, 1935; Whittaker's climax pattern theory, 1953) to an alternative stable state perspective where heterogeneity, resilience, external forcing drivers, non-linearity, and transient equilibria are central components of ecological assembly (Holling 1973, Sutherland 1974, May 1977, Scheffer et al. 2001, Scheffer and Carpenter 2003, Peters et al. 2004). This shift in ecological theory has contributed to changes in existing management directives and the emergence of resilience-based management frameworks in terrestrial and aquatic systems (e.g., The Nature Conservancy's Marine Resilience Program; USDA Natural Resource Conservation Service's Ecological Site Descriptions). However, there are concerns that current management frameworks are not sufficiently addressing the plethora of sustainability issues facing the human population, now and in the future. Society in the 21st century can ill-afford management frameworks that fail to meet the broad sustainability targets of the ecological discipline, or worse, to promote practices that contribute further to ecological degradation. Management frameworks are at risk of being more representative of cultural ideologies and socioeconomic pressures than they are of fully representing the actions needed to conserve biodiversity, production, food, water and human health in a dynamic world. Traditional utilitarian approaches of natural resource management have focused on a solitary ecosystem service, such as grazing or harvest, yet modern societal demands have shifted to embrace management for multiple environmental services across complex landscapes (Berkes et al. 1998, Fuhlendorf et al. 2012). This new purview requires national and international management frameworks capable of establishing the trade-offs of implementing management actions while also prioritizing conservation funding to ensure the sustainable provision of goods and services for modern and future societies.

In order to meet the sustainability targets of scientists, policymakers, and the general public (see for example the Millennium Ecosystem Assessment, MA 2005), broad-scale management frameworks are needed that guide decision-making at national and international levels. In the United States, requests from Congress for a standardized method for classifying and understanding vegetation dynamics on rangelands has led to an interagency collaboration among the Bureau of Land Management (BLM), U.S. Forest Service (USFS), and Natural Resource Conservation Service (NRCS). The interagency collaboration is a byproduct of an executive order (EO 2004) meant to ensure cooperative conservation among the U.S. Departments of the Interior, Agriculture, Commerce, Defense, and the Environmental Protection Agency and is described in detail in the Rangeland Interagency Ecological Site Manual (RIESM 2010). According to the RIESM (2010), Ecological Site Descriptions (ESDs), which were originally developed by the NRCS, will serve as the foundation for identifying, monitoring, evaluating, and managing U.S. rangelands for these three agencies (RIESM 2010). When combined, the BLM, USFS, and NRCS have a management footprint of nearly 370 million ha, which is roughly equivalent to the 7th largest country in the world, India (Central Intelligence Agency 2009). USFS manages national forests and grasslands totaling 78,104,300 ha of public land. BLM is a multiple-use agency that manages 99,148,000 ha of public land for ecosystem services ranging from agricultural production to recreation to energy development. NRCS has provided assistance and monetary support on approximately 188,000,000 ha of private land for conservation-related projects (Briske 2011) while the other agencies focus on public land. The result is a government-sponsored interagency collaboration that solidifies state-and-transition models (Westoby et al. 1989, Bestelmeyer et al. 2003, Briske et al. 2005, 2008) in ESDs as one of the world's largest guiding frameworks for the management and restoration of terrestrial ecosystem services.

State-and-transition models originated from the rangeland discipline's long-held debate on the appropriateness of equilibrium and non-equilibrium paradigms of vegetation dynamics and community reorganization (Briske et al. 2003). Prior to state-and-transition models, rangeland classification and assessment relied on methodologies that compared the composition and biomass of current rangeland vegetation to that of a historical benchmark (BLM and USFS used the ecological status model, NRCS used the range condition model; NRC 1994). These assessment models grew from Clements' (1916) theory of succession and vegetation climax and its refinement by Dyksterhuis (1949) to suggest that succession and retrogression of vegetation are well-defined, predictable changes along a single reversible trajectory (Briske et al. 2003, 2005). Such succession-retrogression models led to a grazing-centric rangeland discipline and the widespread belief that adjustments in grazing pressure could maintain rangelands at equilibrium (Fig. 1). This purview is now widely discredited and has resulted in the adoption of Westoby et al.'s (1989) state-and-transition model as the latest rangeland assessment and classification model (Fig. 1). Unlike previous range models, state-and-transition models are meant to integrate both equilibrium and non-equilibrium dynamics (Westoby et al. 1989, Briske et al. 2003). State-and-transition models still include aspects of the classical range model, but they feature multiple drivers in addition to grazing, multiple alternative stable states, and various trajectories of vegetation change (Briske et al. 2003). More recently, state-and-transition models have incorporated concepts of thresholds, resilience, and heterogeneity into the models (Briske et al. 2005, 2006, 2008, Bestelmeyer et al. 2009, 2011).

The use of state-and-transition models in ESDs is meant to provide a more nationally consistent framework for managing rangeland resources than occurred with preceding rangeland management and assessment frameworks (NRC 1994). State-and-transition models have been developed in ESDs with the intent of guiding land management decisions at a national-level and to help decision-makers prioritize conservation expenditures for the management of rangeland services (NRC 1994). State-and-transition models and ESDs are also being advocated as a valuable tool for the national assessment of new threats to rangeland resources and have been explicitly referenced as a potential tool for projecting future changes in rangelands as a result of climate change (Allen et al. 2009). Yet, state-and-transition models in ESDs are entirely conceptual, based on expert opinion, and have not been exposed to a national-level, scientific critique that determines their existing usefulness compared to their perceived potential.

In this paper, we present a national-scale assessment of the Ecological Site Description Database to determine its usefulness as a national management framework for rangeland resources. This assessment is an important critique of the link between science and management in rangeland ecology. The development of state-and-transition models in ESDs began 15 years ago (Brown and Bestelmeyer 2008), and many ESDs are now being revised to comply with the standards set forth in the interagency agreement (RIESM 2010). To conduct this national-scale assessment, we use approved ESDs that are readily viewable by the general public. Our objectives are to: (1) quantify and summarize the information presented in ESD state-and-transition models; (2) determine whether ESDs fully meet U.S. Congress's goal of a nationally consistent system for defining, mapping, and interpreting ecological sites; (3) identify limitations and logical holes in ESD predictions; and (4) evaluate whether conservation funding priorities are consistent with output from ESDs. Importantly, we are not testing alternative state theory or the conceptual underpinnings of the state-and-transition management framework, but rather evaluating their applications using actual state-and-transition models from one of the world's largest terrestrial management frameworks, the Ecological Site Description (ESD) Database. The findings from this assessment and critique therefore document the extent to

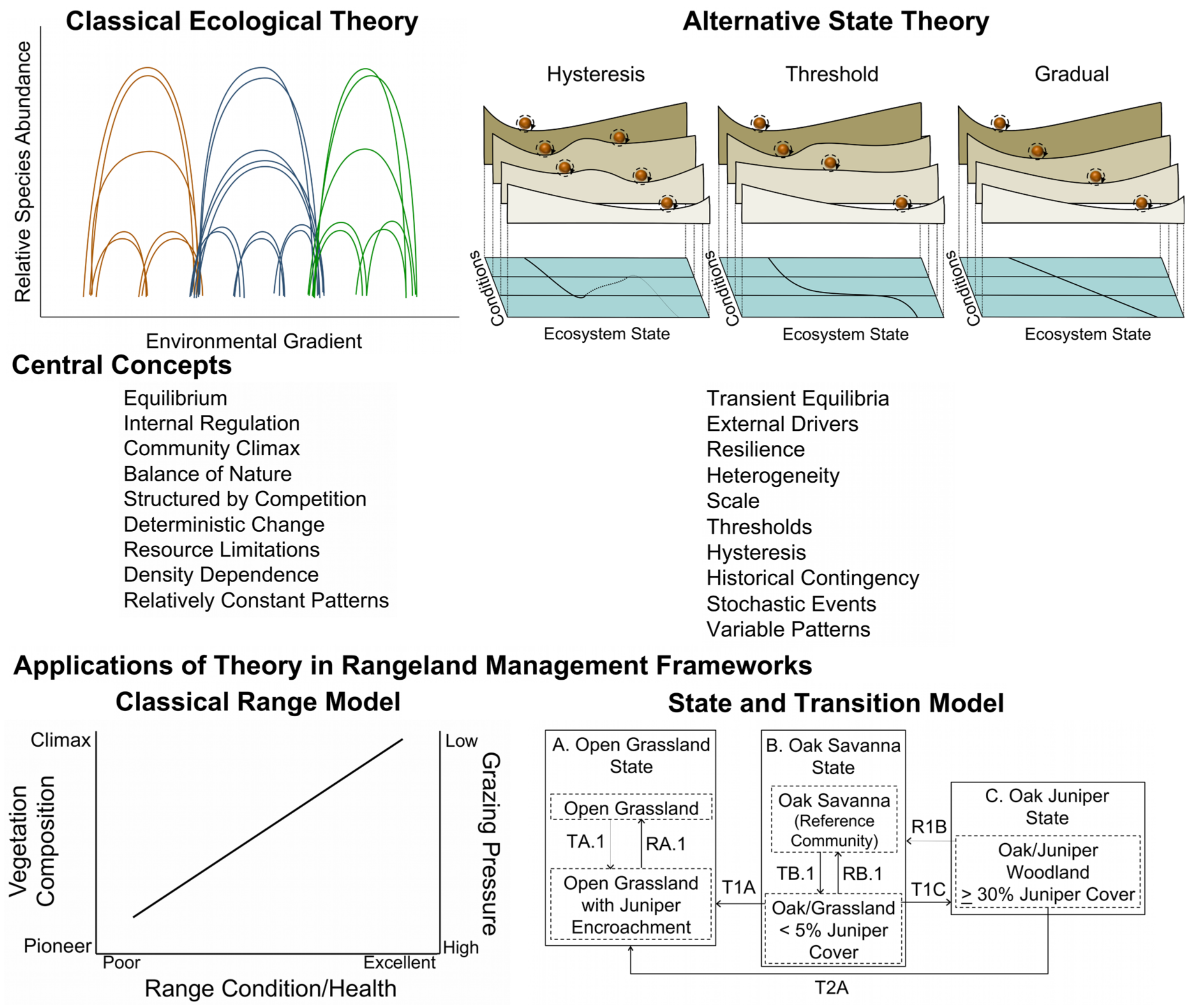

Fig. 1. Applications of theories in ecosystem ecology within rangeland management. Classical ecological theory of community climax (Clements 1916) served as the foundation for the classical range model. The classical range model (based on Dyksterhuis 1949) guided rangeland management actions from approximately 1950–1995 and proposed that vegetation composition could be maintained at a desired equilibrium solely by changing grazing pressure. Rejection of classical ecological theory, its displacement by alternative state theory, and the lack of empirical evidence to support the classical range model led to the emergence of state and transition models as the preferred framework for rangeland management since 1997. Alternative state theory is represented here using landscapes with transient equilibria that change with differing external conditions (modified from Scheffer et al. 2001 and Scheffer and Carpenter 2003); however, state shifts also result from changes in the parameters of ecosystem states (see overview by Beisner et al. 2003). The ball represents the ecosystem state and its position is contingent upon feedback mechanisms (shown as dashed loops) operating within the historical configuration and trajectory of the stability landscape (refer to Scheffer et al. 2001, Scheffer and Carpenter 2003 for further explanation). State changes can be gradual (linear), nonlinear and rapid (threshold), or exhibit multiple states over a range of conditions and follow alternate trajectories of collapse and recovery (hysteresis) (adapted from Scheffer and Carpenter 2003, Suding and Hobbs 2009). The state and transition model given here is a qualitative model based on expert opinion that characterizes alternative ecosystem states as boxes and the feedback mechanisms driving transitions between states as arrows (adapted from Westoby et al. 1989, Briske et al. 2008, ESD User's Guide 2011).

which rangeland management has moved past preceding management paradigms based on concepts of equilibrium, grazing-driven successional retrogression, and community climax.

Methods

Our review required the use of two databases: the Ecological Site Description Database (ESD 2011) and the Soil Survey Geographic Database (SSURGO 2012). The ESD Database contains a series of reports that characterize (1) physiographic, climatic, water, and soil features, (2) potential plant communities and vegetation dynamics, and (3) site interpretations based on a hierarchical land classification system. At the apex of the hierarchy are major land resource areas (MLRAs; defined as "an area of similar climate, physiography, dominant soil taxa, and consequently, land use and vegetation" [Bestelmeyer et al. 2009]). Two hundred twenty-six MLRAs have been classified within the US and mapped into 1805 distinct spatial polygons (note that a MLRA can be mapped more than once; USDA NRCS Geospatial Data Gateway, accessed March 09, 2012). Within each MLRA are multiple ecological sites defined as "a class of land based on recurring soil, landform, geological, and climate characteristics that differs from other such classes in (1) the production and composition of plant species under the disturbance regime of reference conditions, associated dynamic soil property levels, and ecosystem services provided and (2) responses to management and the processes of degradation and restoration" (Bestelmeyer et al. 2009). For each ecological site, Ecological Site Descriptions (ESDs) present information relevant to seven main sections: ecological site characteristics, physiographic features, climatic features, influencing water features, representative soil features, plant communities, and ecological site interpretations. Within the "plant communities" section, ESDs contain a figure of a state-and-transition model, and often include detailed written text explaining the states and transitions featured in the model. Additional background on the development and application of ESDs and state-and-transition models is available in a relatively recent special issue of Rangelands (2010; volume 32, issue 6).

To determine which ESDs to review and critique, we used the SSURGO database to identify the ecological sites that accounted for the greatest amount of land area within each MLRA. At present, the ESD database operates only as a means to view information for a particular ESD, which is referenced by a unique ecological site identifier (e.g., R064XY015NE). Information within the database is not spatially referenced and a user is unable to retrieve an ESD (and information contained therein) for any precise location. Unlike the ESD Database, SSURGO includes geospatially accessible data; for a given location a user can look up ecological information including an ecological site identifier. By using the ecological site identifier within these two databases, we were able to link the geospatial information within SSURGO to the non-spatial information of ecological sites in the ESD Database.

We downloaded SSURGO data for the lower 48 states of the U.S. from the USDA NRCS Geospatial Data Gateway (data accessed March 09, 2012). Data were imported into a spatially enabled database (PostgreSQL/PostGIS). Spatially mapped areas within SSURGO are delineated into soil map units. These map units are composed of smaller unmapped soil types called components. For each component, SSURGO provides detailed soils information (type, taxonomy, etc.), identifies whether it is the dominant component within a map unit, and labels it with the appropriate ecological site identifier. Because soil map units contain smaller unmapped components, it is possible for multiple ecological site identifiers to occur within a soil map unit. To spatially reference ecological sites, we assigned soil map units information from the dominant component. As a result, a soil map unit was associated with a single ecological site. This resulted in 11,413,380 soil map units and approximately 6000 unique ecological sites across the western US ranging in size from 0.0001 to 185,000-ha. Of the 1805 spatially distinct MLRA polygons, 487 contained ecological sites within them. These 487 MLRA polygons became our basis for mapping and summarizing ecological sites. Using these spatial data, we identified and selected the five ecological sites that accounted for the greatest amount of land area within each spatially distinct MLRA polygon. These five ecological sites represented 95%

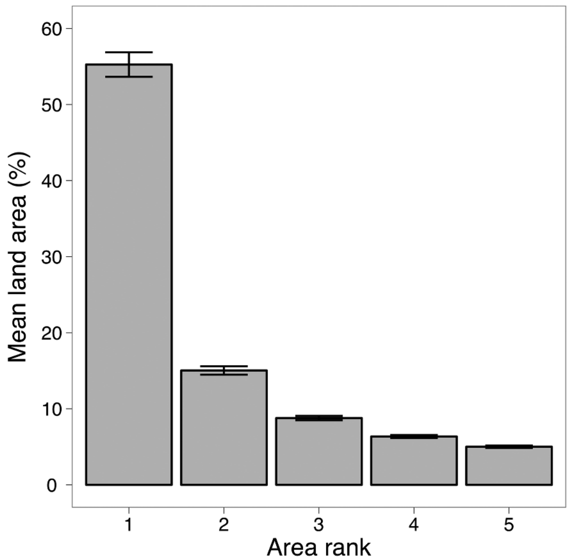

Fig. 2. The mean land area (%) accounted for in spatially distinct major land resource areas (MLRA) by each of the top five ecological sites of greatest area in the SSURGO Database. The top five ecological sites of greatest area in each MLRA formed the basis for this review and cumulatively accounted for 95% of land area, on average.

of the land area within spatially distinct MLRA polygons (Fig. 2) and resulted in a total of 789 ecological sites to be examined.

We used a systematic approach to assess the 789 ESDs (Fig. 3). We first determined the proportion of ESDs that had been completed and whether they met the list of requirements needed for their approval (Table 1). A major section was deemed to be finished if (1) the template or table outlined in the ESD User's Guide (2011) was completed for the section, or (2) if, in the absence of a template or table, detailed information was written to thoroughly explain the vast majority of subheadings within major sections. Of the 789 ESDs examined, 378 (47.9%) completed at least one major section. For those 378 ESDs, the ecological site characteristics section was completed 100% of the time, physiographic 99%, climatic 85%, water 74%, soil 87%, plant communities and state-and-transition models 90%, and site interpretations 82%. The highest rate of completion for state-and-transition models occurred in the central and southern plains (Fig. 4). Many ESDs were being revised to comply with the memorandum of agreement established in 2010 (RIESM 2010), which may explain why over 50% of the ESDs we evaluated had yet to be approved and released to the general public. The development of new ESDs may also explain the surprisingly low rate of completion in regions where their development has been a priority (e.g., Utah; Fig. 4).

Addressing Objective 1

To quantify and summarize the information presented in ESD state-and-transition models, we manually reviewed each of the 789 state-and-transition models based on the stepwise approach shown in Fig. 3. We classified the type and number of ecological states (labeled as boxes in ESDs), the state identified as the historical climax plant community (HCPC; a term used by the USDA-NRCS to identify the plant community or ecological state that occurred at the time of European settlement of North America; ESD User's Guide 2011:43), the drivers of state transitions (labeled on arrows in ESDs), the type of change that occurred as a result of a transition between two alternative stable states, the driver(s) listed in the state-and-transition model that was associated with the ecological degradation of the HCPC, and the driver(s) listed in the state-and-transition model that was associated with the ecological restoration of the HCPC. Defining ecological states is not standardized in ESDs, so the manner in which states are named or characterized has a strong influence over the types of changes that are possible between states. To ensure consistent evaluation across all ESD state-and-transition models, we developed standardized classification schemes to quantify ecological states (Table 2), the drivers of state transitions (Table 2), and the type of change(s) that accounted for a transition between two alternative states (Table 3). While this approach provided a consistent foundation for our assessment, it prevented us from distinguishing between more specialized groupings of ecological states (e.g., mixed prairie versus tallgrass prairie) or drivers of transitions between states (e.g., continuous grazing versus rotational grazing; no fire versus prescribed fire). To evaluate pathways of degradation and restoration (Fig. 3), the state labeled as the historical climax plant community (HCPC) in each ESD was used as the reference

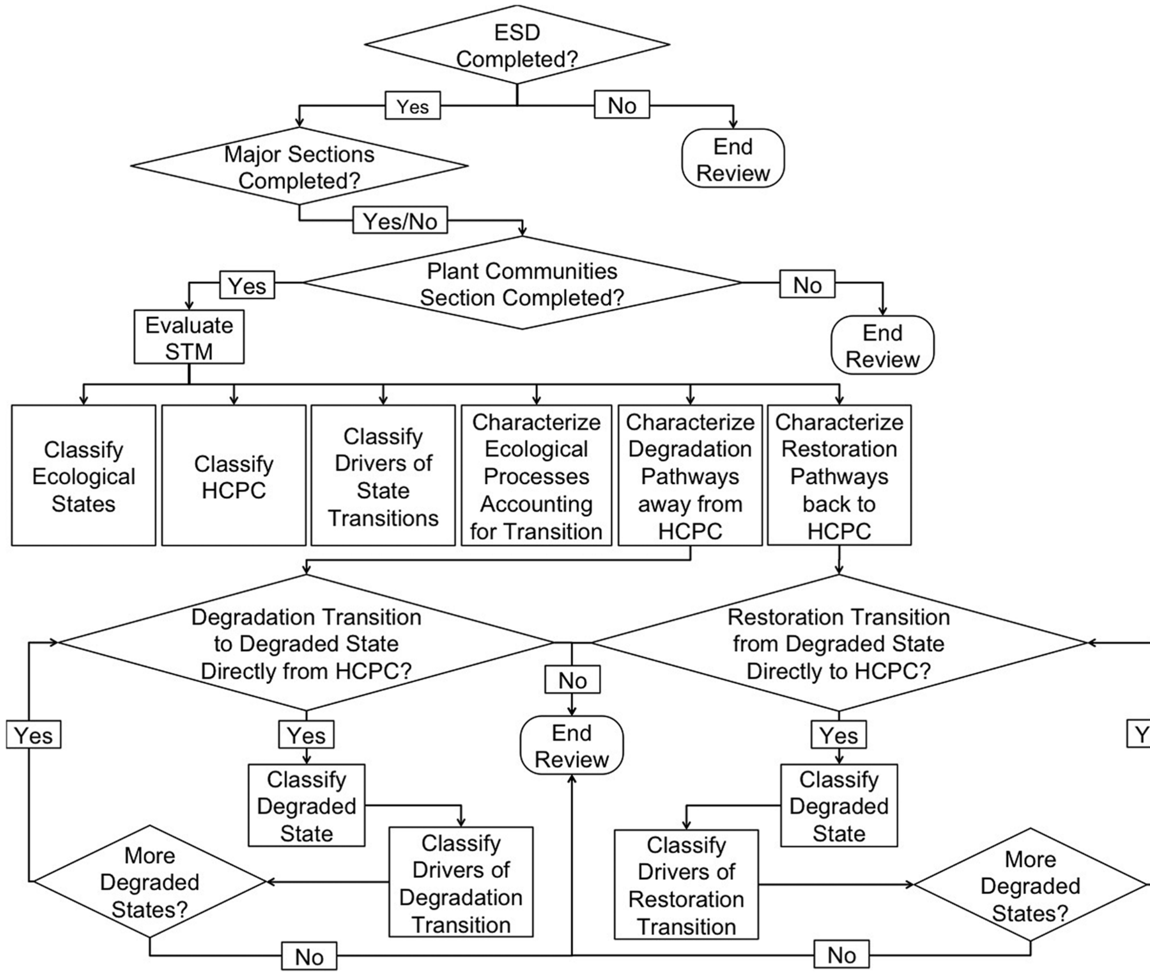

Fig. 3. Flow diagram showing the systematic process we used to review and evaluate state-and-transition models featured in the Ecological Site Description (ESD) System Database.

state for degradation and the ultimate target for restoration (this view of degradation and restoration is consistent with the guidelines established for managers in the ESD User's Guide 2011:48). Only degraded states directly connected to the HCPC were used. Thus, we did not evaluate restoration or degradation pathways that proceed through multiple states and transitions.

The percentage of land area (A) a specific ecological state, driver of state transitions, or type of ecological change was characterized across all spatially distinct MLRA polygons within the ESD Database was calculated as

$$A = \frac{f(\text{area factor characterized})}{f(\text{area completed ecological sites})} \times 100$$

where f (area factor characterized) is a function depicting the total land area a given factor (an ecological state, driver of state transitions, or type of ecological change) is characterized within the five largest ecological sites occurring within MLRAs based on:

$$f(\text{area factor characterized}) = \sum_{i=1}^{487} \sum_{j=1}^{5} M_i \times Apc_{ij}$$

where M_i is the area of the ith of 487 MLRAs and A_j is the proportion of area the jth of the largest 5 ecological site occurs within the ith MLRA given

Table 1. The specific components we evaluated in our review of the Ecological Site Description (ESD) System Database and the minimum requirements outlined in the (RIESM) policy that are required for ESD approval.

Minimum content requirements to be included in ESD	Evaluated in this review
I. General Information including:	X
ecological site name	X
ecological site number	X
map identifying approximate geographic extent of the ecological site	
II. Physiographic Features including: landform, geology, aspect, elevation, slope, water table, flooding, and ponding	X
III. Climatic Features including: frost-free period (length and dates), freeze-free period (length and dates), mean annual precipitation, monthly moisture and temperature distribution, and name of approved climatic stations	X
IV. Influencing Water Features existing on the site or adjacent wetland/riparian ecological sites that influence vegetation and/or management of the site.	X
V. Representative Soil Features including those that differentiate from other ecological sites, affect plant adaptation, establishment, growth, and response to disturbance.	X
VI. Ecological Dynamics of the Site including:	X
states	X
transitions	X
thresholds	X
restoration pathways	X
community phases	
animal species	
wildlife habitat elements	
hydrology	
changes in soil properties that are expected to occur as a result of disturbances and/or stresses	
i. Include information related to landscape scale processes such as runoff, erosion, fire behavior, wildlife use, etc.	
ii. Discussion of temporal scale associated with transitions, community pathways, restoration pathways, and thresholds. Where information exists about response to disturbance or management actions, probabilities of occurrence can be included (drought occurrence, fire frequency intervals).	
VII. Vegetation	X
i. Describe the most common, predominant, and/or ecologically significant states and community phases. Include description of transitions, restoration pathways, and community pathways. Include a state-and-transition diagram.	X
ii. Describe ecologically significant associations of plant species that indicate important environmental gradients and to differentiate sites, states, or plant community phases.	X
iii. Use standardized plant names from the Integrated Taxonomic Information System as presented in the NRCS PLANTS database.	
iv. For the reference state include a narrative description, detailed listing of plant species (includes scientific and common name, normal annual production in pounds air dry weight (ADW) per acre, and either canopy, foliar, or basal cover (depending on life form), total annual production by growth form (median ADW pounds per acre per year in favorable, normal, and unfavorable years), and growth curve (monthly growth by plant species or communities).	X
v. For all other states/community phases include, at a minimum, a narrative description	X
vi. Productivity of Major Tree Species: annual productivity and site index for forested plant communities occurring on rangeland ecological sites, if applicable.	
VIII. Supporting Information	X
i. Record information about the relationship of the ecological site to other ecological sites and the documentation and references used to develop the rangeland ecological site description.	
ii. Identify relationships to other classification systems such as National Vegetation Classification System (NVCS).	

that the factor was present (p, a binary value of 1 or 0) and the state-and-transition model was completed (c, a binary value of 1 or 0) (note that if p or c is 0, the area for the ith ecological site occurring within the jth MLRA is 0). f (area completed ecological sites) is a function that depicts the total land area of all ecological sites that were the among five largest within all MLRAs, as shown below:

$$f(\text{area completed ecological sites}) = \sum_{i=1}^{487} \sum_{j=1}^{5} M_i \times Ac_{ij}$$

An example where we use this equation to calculate the percentage of land area a given

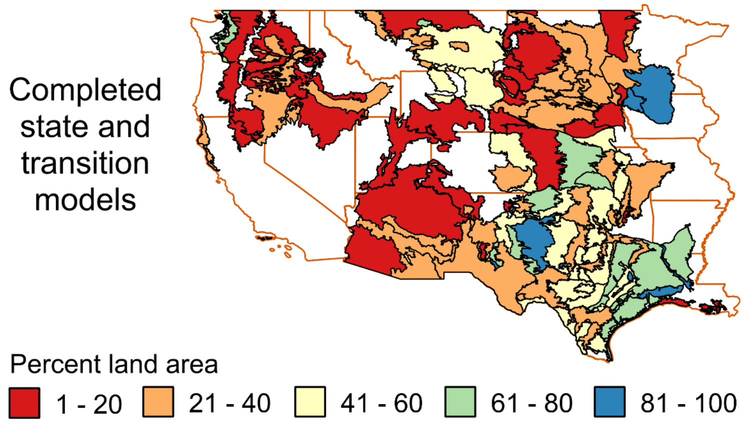

Fig. 4. The percentage of land area where state-and-transition models in ESDs have been completed and approved for the five ecological sites of greatest area within each major land resource area (MLRA). Blank areas show MLRAs where state-and-transition models for the five largest ecological sites have not been completed or approved.

factor was characterized across MLRAs is shown in the Appendix.

Addressing Objective 2

To determine whether ESDs fully meet U.S. Congress' goal of a nationally consistent system for defining, mapping, and interpreting ecological sites, we assessed whether developers of ESDs used the same approach and framework when filling-out each major section. Specifically, we evaluated whether the eight major sections required for approval had been completed consistently (Table 1). In the plant communities section, we assessed whether developers labeled state-and-transition models in the same manner. We tracked how states (boxes) and drivers of transitions (arrows) were labeled, the presence of a historical reference point, the characterization of thresholds, and the organization and content of text used to describe the state-and-transition model.

Addressing Objective 3

To identify limitations and logical holes in ESD output, we critiqued multiple components of state-and-transition models in ESDs and identified emergent patterns or themes in the models that were nonsensical. We identified management outcomes that were not supported in the scientific literature. We also assessed whether management outcomes in ESD state-and-transition models were outlined as being unrealistic or unlikely in a recent USDA NRCS funded scientific evaluation of rangeland management practices (Briske 2011). Lastly, because ESDs are most useful as a tool that can be modified or incorporated into national plans and assessments for dealing with new threats to the management of ecological resources (e.g., climate change), we identified major weaknesses in ESDs that limits their utility to prioritize land management decisions at a national-level.

Addressing Objective 4

To evaluate whether NRCS conservation-funding priorities are consistent with modeled output from NRCS ESDs, we compared NRCS conservation expenditures on management practices to the relative importance of those management practices within ESD state-and-transition models. A recent summary of NRCS conservation expenditures was presented in Tanaka et al. (2011). Data on NRCS conservation expenditures were classified for management actions (prescribed grazing, prescribed fire, brush management, and reseeding), riparian cover, and upland wildlife habitat. Upland wildlife habitat management comprised the second highest conservation expenditure of the NRCS (Tanaka et al. 2011). However, this classification did not delineate between specific management actions used for the purposes of creating or improving wildlife habitat (Tanaka et al. 2011). As such, we focused our attention on grazing, fire, brush management

Table 2. Classification scheme used to review and evaluate the ecological states and drivers of transitions between states featured in the Ecological Site Description (ESD) System Database, including the number of state-and-transition models (No. STMs) and the percent land area occupied by each ecological state and driver across all major land resource areas (MLRAs).

State/transition classification	Description	No. STMs†	Land area (%)‡
Ecological states			
Herbaceous	State dominated by herbaceous plants with <10% woody cover; no exotic species	323	73.0
Herbaceous-shrub mix	State co-dominated by herbaceous plants and woody species typically <2.5 m tall; woody cover ≥10%; no exotic species	193	33.5
Herbaceous-tree mix	State co-dominated by herbaceous plants and woody species typically ≥2.5 m tall; woody cover ≥10%; no exotic species	139	28.8
Shrubland	State dominated by woody species typically <2.5 m tall with bare ground or little herbaceous cover; bare ground not the result of erosion; no exotic species	34	05.0
Forest/woodland	State dominated by woody species typically ≥2.5 m tall with bare ground or little herbaceous cover; bare ground not the result of erosion; no exotic species	70	22.1
Exotic herbaceous	State dominated by herbaceous plants with <10% woody cover; exotic species present	106	31.2
Exotic herbaceous-shrub mix	State co-dominated by herbaceous plants and woody species typically <2.5 m tall; woody cover ≥10%; exotic species present	58	09.9
Exotic herbaceous-tree mix	State co-dominated by herbaceous plants and woody species typically ≥2.5 m tall; woody cover ≥10%; exotic species present	23	05.9
Converted	State described as cropland, pastureland, "go-back" (successional result of agricultural abandonment), or similar	87	30.0
Eroded	State dominated by bare ground resulting from erosion; woody cover <10%	41	14.8
Eroded shrubland/woodland	State dominated by bare ground resulting from erosion; woody cover ≥10%	30	04.4
Any plant community	State designated in ESDs as "any plant community" or "new site"	39	20.8
Undefined	State was not defined within state-and-transition model or given a general label (e.g., disturbance state 1)	70	11.1
Unable to determine	Information provided in major section, "plant communities", but diagram or text were insufficient to classify the state	8	NA
Drivers of state transitions			
Grazing	State change caused by the presence, absence, or modification of various grazing practices	268	88.3
Fire	State change caused by prescribed fire, wildfire, or lack of fire	235	81.1
Brush management	State change caused by brush management (or lack thereof), tree harvesting, or logging through chemical or mechanical treatment (not fire)	209	68.9
Woody encroachment	State change caused by encroachment of woody plant species	81	26.0
Climate	State changed caused by climatic drivers (e.g., drought, timing of precipitation)	53	19.2
Exotic herbaceous invasion	State change caused by the introduction and invasion of an exotic herbaceous species	43	19.8
Exotic woody invasion	State change caused by the introduction and invasion of an exotic woody species	7	2.9
Reestablishing native species	State change caused by reseeding or replanting of natives	131	52.3
Erosion	State change caused by soil loss	33	08.8
Conversion	State change caused by conversion to agricultural land or pastureland or cultivation	94	33.4
None	No state changes in state-and-transition model; only a single state was given	21	02.1
Undefined	Transition (arrow) linking two states was not labeled or characterized	76	07.1
Other	Any transition causing a state change that occurred infrequently and did not fit the above categories (e.g., soil disturbance by pigs; soil addition)	77	22.4

† A total of 340 ESDs included a state-and-transition model.
‡ Land area values were calculated using the equation in the text (see also the example in the Appendix).

Table 3. Summary of the categories used to explain changes between stable states in the Ecological Site Description (ESD) System Database, the number of state-and-transition models (No. STMs), the percent land area occupied by each type of change across all major land resource areas (MLRAs) and examples of state changes from the ESD database.

Ecological change	Description	No. STMs†	Land area (%)‡	ESD example of state change
Woody encroachment	Increase in the density and abundance of native woody plant species	239	68.3	Bluestem-sandreed prairie to eastern redcedar woodland in Nebraska-Dakota Eroded Tableland
Shift in composition of native herbaceous species	Change in dominance between native herbaceous functional groups	163	57.0	Prairie sandreed-needle and thread-bluestem prairie to sedge-blue grama-needle and thread prairie in Northern Rolling High Plains of South and North Dakota
Shift in composition of native woody species	Change in dominance from one woody species or assemblage to a new woody species or assemblage	62	15.3	Oak savanna to juniper-oak woodland in Edwards Plateau of Texas
Exotic invasion	Increase in the density, cover, or abundance of an exotic herbaceous or woody species	130	41.2	Coastal sagebrush to non-native annual grassland in Southern California Mountains
Shift in composition of exotic herbaceous species	Change in dominance from one exotic herbaceous species to a previously subdominant or non-existent exotic herbaceous species	6	0.5	Mesquite-non-native annual co-dominated state to mesquite-Lehman lovegrass co-dominated state in southeastern Arizona Basin
Erosion	Increase in bare ground resulting from soil loss; may or may not be the result of loss of surface vegetation	82	24.8	Juniper woodland to eroded juniper woodland in Blue Mountain Foothills of Oregon
Woody reduction	Decrease in the density, cover, or abundance of woody plants	223	70.7	Shrubland to mixed shrub-grassland in Southern Desertic Basins, Plains, and Mountains of New Mexico
Reseeding native species	Reseeding, replanting, or reestablishing native species through human intervention	133	52.5	Go-back land to seeded rangeland in Upper Arkansas Valley Rolling Plains of Colorado
Eradication of exotic invader	The wholesale eradication and removal of an exotic invasive species after establishing dominance	60	20.7	Rabbitbrush-cheatgrass state to wheatgrass-needle and thread state in Central Desertic Basins and Plateaus of Wyoming
Permanent inundation	Permanent increase in water depth and inundation	5	< 0.1	Coastal marsh to open water in gulf coast of Louisiana
Conversion	Conversion or cultivation of ecosystems to agricultural land or pastureland	103	40.5	Upper tidal meadow to farmland in Willamette and Puget Sound Valleys of Washington

† A total of 340 ESDs included a state-and-transition model.
‡ Land area values were calculated using the equation in the text (see also the example in the Appendix).

and reseeding to relate funding priorities to ESD state-and-transition models.

Results of a National-Level Critique of ESDS

Objective 1: Quantify and synthesize information in ESD state-and-transition models

Ecological states.—Of the 789 ESDs we reviewed, 340 state-and-transition models had been completed and approved (Fig. 4). These state-and-transition models contained a relatively low number of alternative ecological states (mean = 3.40 ± 0.05; Fig. 5). Most ecological states were classified as native herbaceous, followed by native herbaceous-shrub mix and native herbaceous-tree mix (Table 2). Ecological states were classified into these three categories more frequently than the total number of states classified as shrubland, forest/woodland, converted, or associated with exotic species or

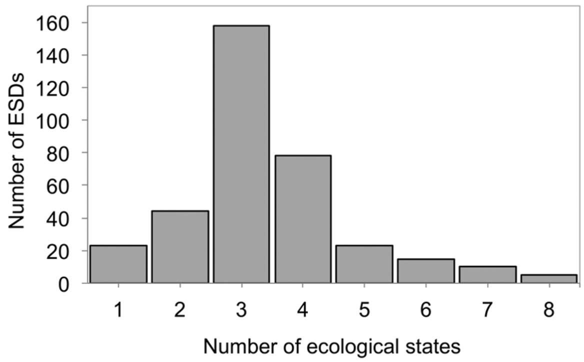

Fig. 5. Frequency distribution of the number of Ecological Site Descriptions (ESDs) that contained a given number of ecological states.

erosion. States classified as native herbaceous were also listed as the HCPC in 59.5% of ESDs and covered the most land area (Table 2; Fig. 6). The next most frequent classifications of HCPC states were native herbaceous-shrub mix (23.4% of ESDs) and native herbaceous-tree mix (10.5% of ESDs). All other classifications of ecological states represented the HCPC in less than 4% of ESDs suggesting that regional experts (i.e., ESD authors) perceive their historical landscapes to largely be dominated by grasses and forbs.

Drivers of state transitions.—The dominant management actions promoting transitions between states in ESDs were grazing, fire, brush management, reseeding, and cultivation, respectively (Table 2; Fig. 7). Grazing was associated with state transitions more frequently and on more land area than any other natural process or management action (Table 2; Fig. 7). Overall, 79% (268 of 340) of state-and-transition models in ESDs listed grazing as a driver of state transitions indicating that it is considered the dominant driver of state changes by professionals. Grazing was linked to 31% (303 of 994) of degradation transitions from the HCPC and 27% (305 of 1120) of restoration transitions back to the HCPC. This was more than any other driver of degradation or restoration (Fig. 8).

In contrast, state-and-transition models in ESDs indicated that ecological processes independent of traditional management actions accounted for a small proportion of state transitions (Table 2). Climate change, whether described as normal climatic fluctuations (e.g., drought) or long-term deviations from the historical norm (e.g., wetter climatic regime), was featured as a possible driver of state transitions in only 16% (53 of 340) of ESDs. Exotic invasion of herbaceous or woody plants were featured in 13% (43 of 340). Erosion was listed as a driver in 10% (33 of 340).

Changes associated with state transitions.—Woody reduction and woody encroachment were the clear dominant ecological changes in ESDs (Table 3; Fig. 9). Changes associated with woody reduction and woody encroachment were identified on approximately 15% more land area than shifts in native herbaceous species composition, 20% more area than reestablishing native species, 30% more area than exotic invasion or cultivation, and on more than double the land

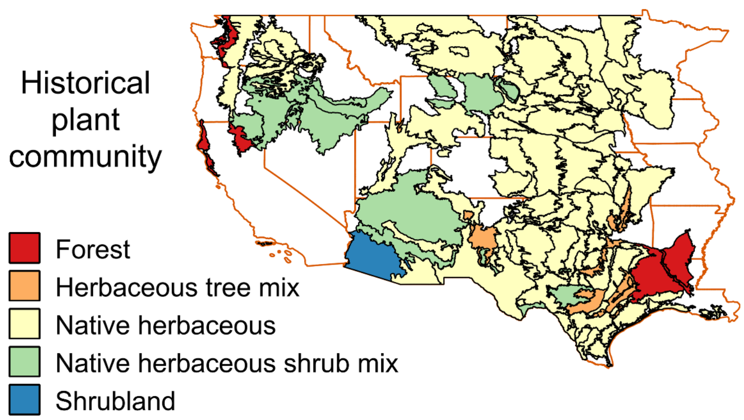

Fig. 6. Map of the dominant historical climax plant community (HCPC) for each Major Land Resource Area. HCPCs designate the historical point of reference in ESD state-and-transition models. Blank areas show MLRAs where state-and-transition models for the five largest ecological sites have not been completed or approved.

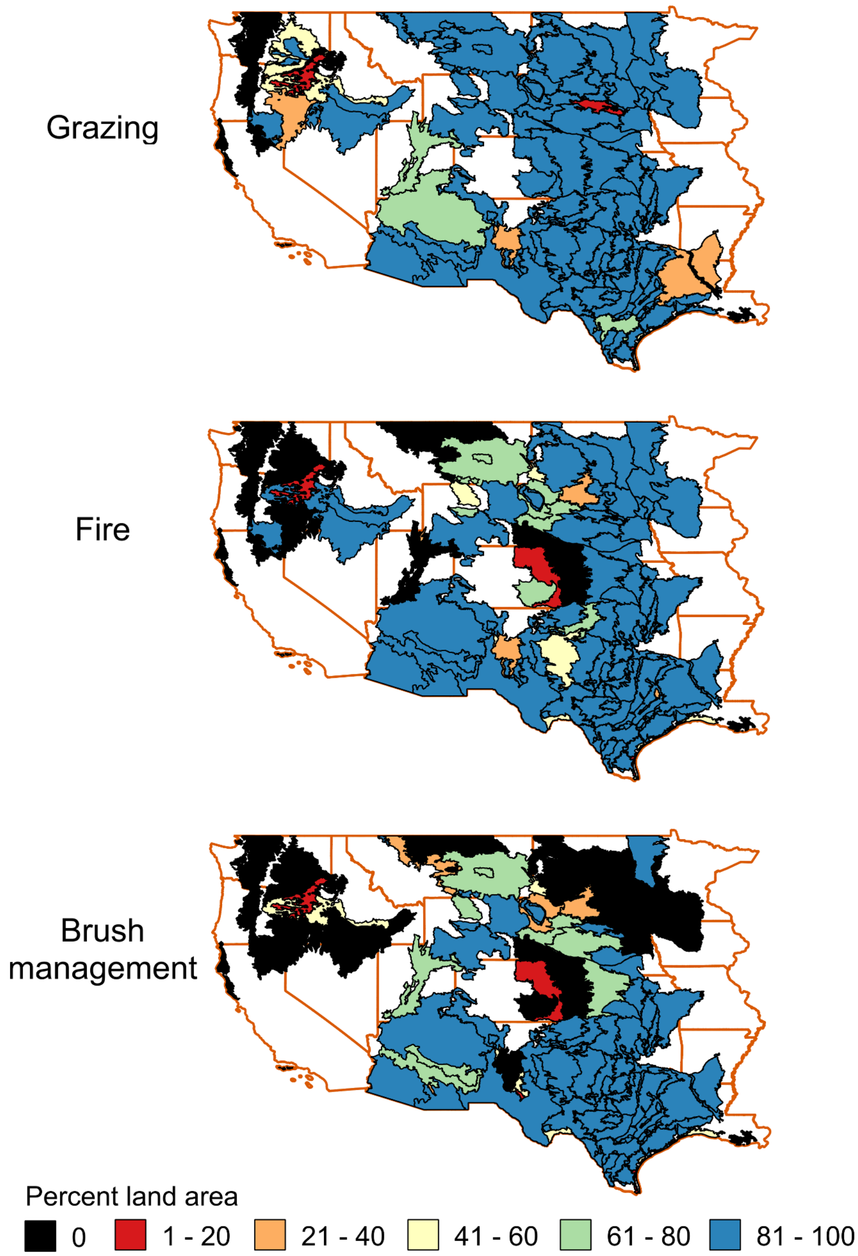

Fig. 7. The three dominant drivers of state transitions in Ecological Site Descriptions (ESDs) and the weighted percentage of land area within major land resource areas (MLRAs) that each driver was featured in state-and-transition models.

area of all other state changes (Table 3; Fig. 9). Based on the emphasis on woody reduction and woody encroachment in defining ecological states and their transitions in ESDs, our assessment suggests that rangeland professionals consider changes in woody plant abundance to be the dominant ecological change occurring on rangelands.

Objective 2:
Determine whether ESDs fully meet U.S. Congress's goal of a nationally consistent system for defining, mapping, and interpreting ecological sites

The ESD User's Guide (2011) provided a

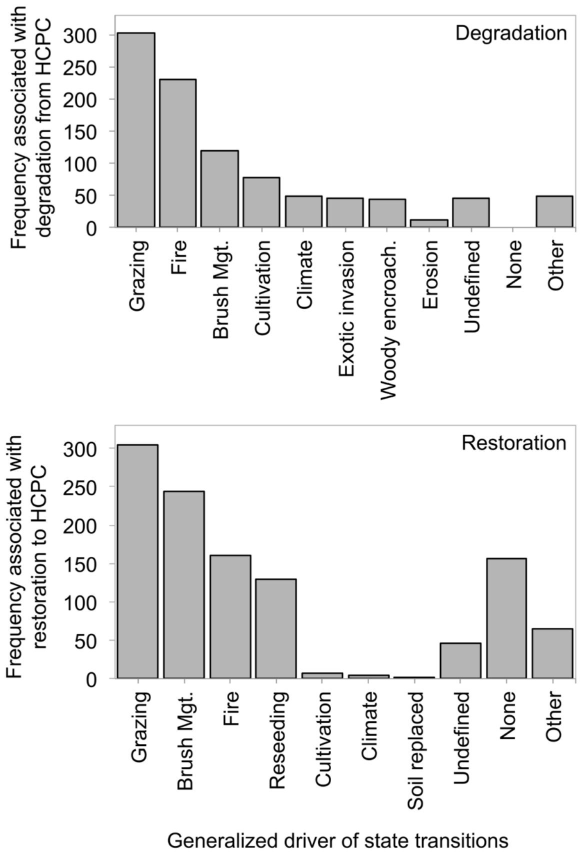

Fig. 8. The frequency each driver of state transitions was associated with degradation pathways away from the HCPC (top panel) and restoration pathways back to the HCPC (bottom panel) in ESD state-and-transition models. Note that directionality is not implicit for each generalized driver (i.e., grazing = no grazing or prescribed grazing; fire = no fire, prescribed fire, wildfire).

consistent foundation for the creation of state-and-transition models in ESDs, but regional experts tasked with developing ESDs varied in their interpretation of how to complete certain sections. Overall, ESDs were formatted consistently, and each contained the same list of major sections that are required for USDA-NRCS approval. Within major sections, a consistent template was observed and followed for 7 of 8 major sections. However, the most important section, the plant communities section with the state-and-transition model and community description, was the main exception. In this critical section, all ESD state-and-transition models represented ecological states as boxes, transitions between ecological states were represented as arrows, and a single state was selected to represent the historical climax plant community (HCPC) and to serve as a reference point for identifying pathways of ecological degradation and restoration. In contrast, the presentation of state-and-transition models, how they were developed, how information was organized in the text, and how components were defined within each state-and-transition model differed among individual creators, with large discrepancies observed among regions (Fig. 10).

The most notable inconsistency was how state transitions (arrows) were characterized in ESDs. Arrows connecting states were labeled differently from one ESD to the next. Some ESDs labeled transitions with (1) management actions (e.g., brush management, prescribed grazing; Shallow Savanna in KS, ESD ID: R112XY031KS) whereas others used (2) triggers (e.g., episodic inundation; Lakebed in TX, ESD ID: R078BY078TX), (3) feedbacks (e.g., soil erosion; SR Mountain in OR, ESD ID: R010XC032OR), and, in extremely rare instances, (4) quantitative thresholds (e.g., salinity levels above 13 ppt; Saline Mineral Marsh in LA, ESD ID: R151XY002LA). These inconsistencies reflect the uncertainty that exists among individual developers on how to characterize and rank the processes driving transitions, how to define thresholds, and how to interpret the terminology used to direct state-and-transition model development (Knapp et al. 2011). Based on the ESD User's Guide (2011), developers are required to operationalize the terms, feedback mechanism and resilience, when labeling transitions between alternative stable states. In ESDs, feedback mechanisms are defined as "ecological processes that enhance (negative) or decrease (positive) ecosystem resilience." Resilience is defined in ESDs as "the amount of change or disruption required to transform a system from being maintained by one set of mutually reinforcing processes and structures to a different set of processes and structures." While a focus on resilience and feedback mechanisms is an attempt to give state-and-transition models a theoretical foundation (Briske et al. 2008), such definitions are consistent with increasing tendencies to characterize resilience as a more vague and malleable concept, which is considered to

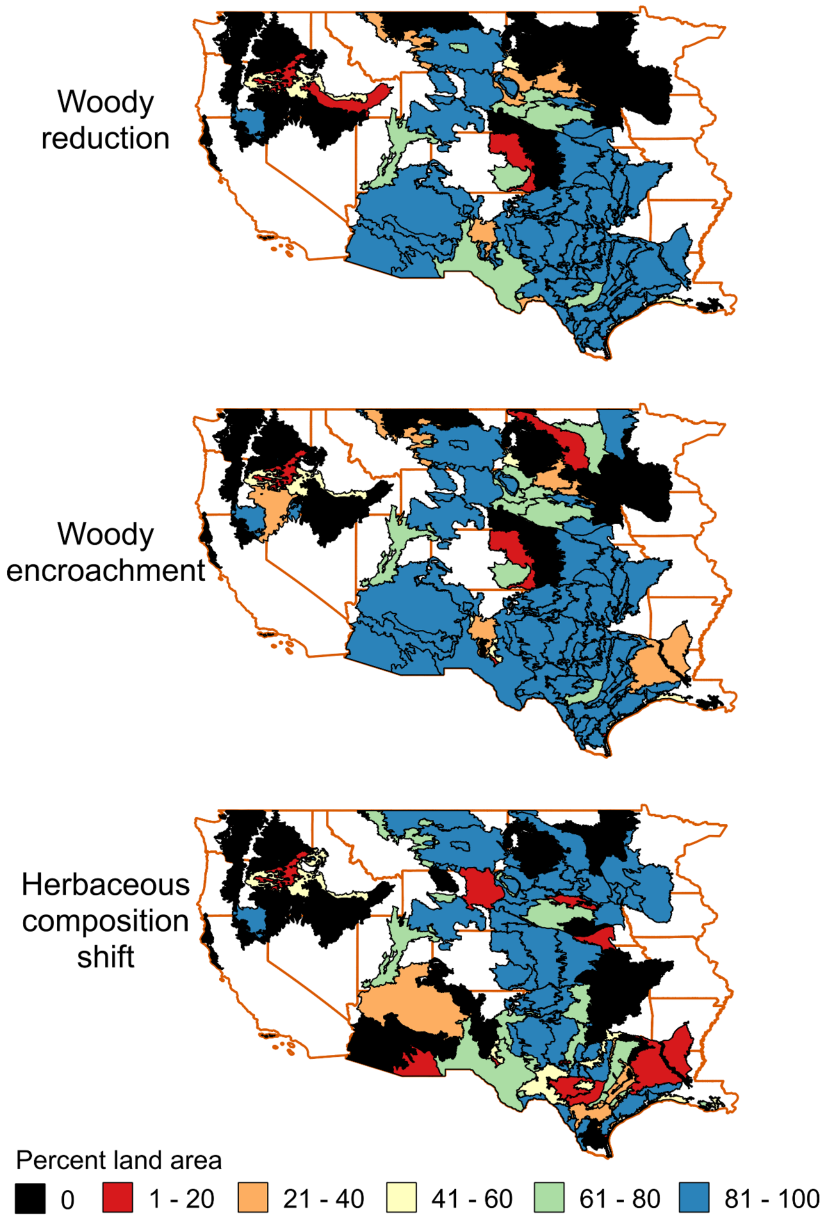

Fig. 9. The three dominant types of changes in Ecological Site Descriptions (ESDs) and the weighted percentage of land area within major land resource areas (MLRAs) each type of state change was featured in ESD state-and-transition models.

have reduced the conceptual and practical utility of the resilience concept (Cumming et al. 2005, Brand and Jax 2007). Based on the inconsistencies that emerged in our review of ESDs, a more clearly specified working definition of transitions in state-and-transition models is needed to operationalize the resilience concept and make ESDs more consistent among developing individuals.

Objective 3: Identify limitations and logical holes in ESD output

The grazing-woody plant fallacy.—The issue that

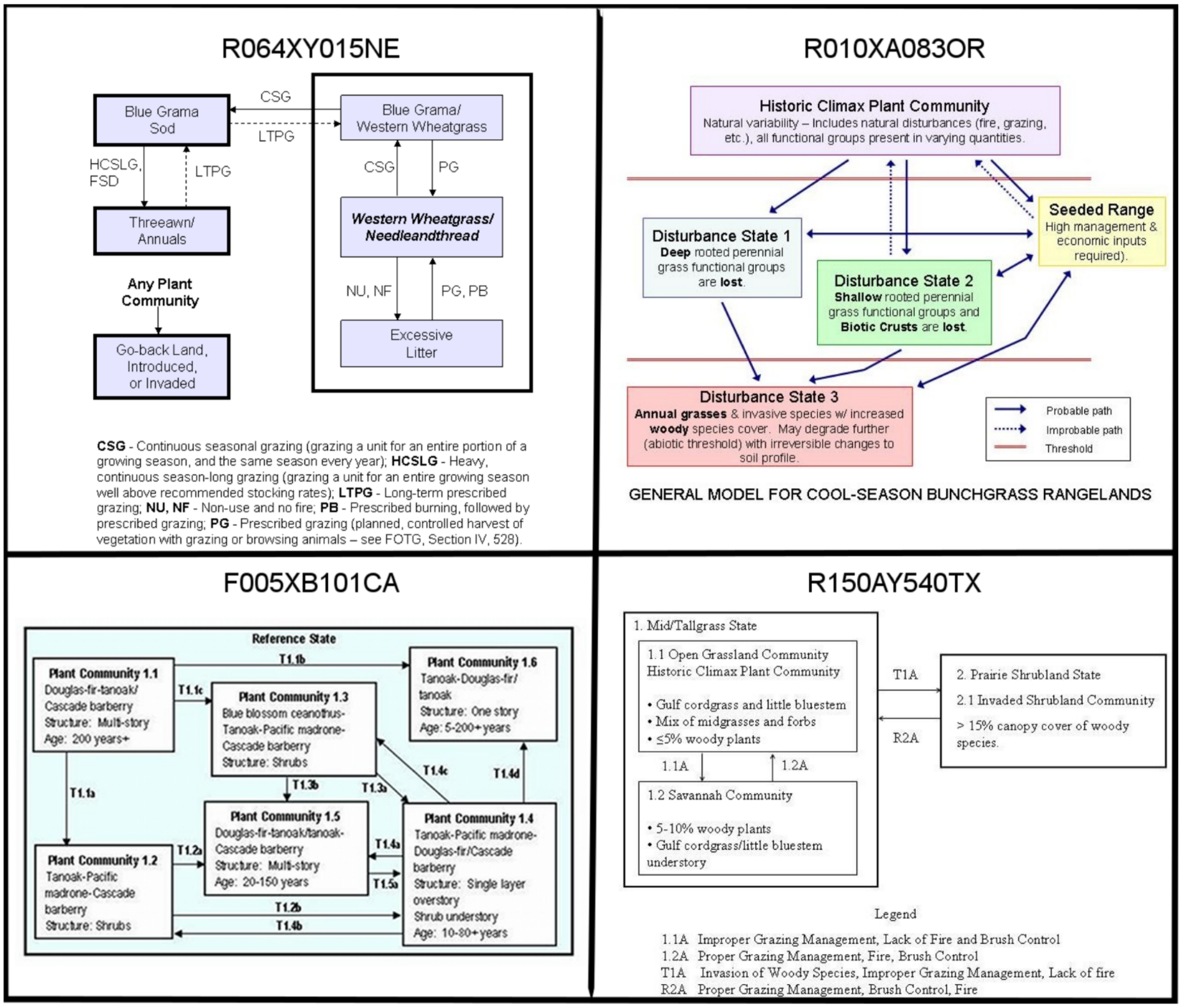

Fig. 10. Examples of regional differences in how ESD state-and-transition models are organized, labeled, and defined. Examples are from Nebraska (R064XY015NE), Oregon (R010XA083OR), California (F005XB101CA), and Texas (R150AY540TX).

immediately emerges from our review is why grazing is listed as the number one driver of both degradation and restoration when woody plant encroachment and reduction characterize the two dominant state changes in ESD state-and-transition models. Decades of scientific research suggest grazing management does little to prevent the conversion of grass-dominated ecosystems to woody-dominated ecosystems upon the onset of woody plant encroachment (see overview by Archer et al. 2011). Grazing management has the potential to prevent undesirable transitions in herbaceous vegetation (Bestelmeyer et al. 2013), and consumption of grasses and dispersal of seeds by grazers have accelerated rates of encroachment of some woody species (Van Auken and Bush 1997; but see Brown and Archer 1989). However, unless managers switch to browsing ungulates, conversion from grass-dominated to woody-dominated ecosystems still typically occurs even if managers are using "proper" grazing systems, reducing stocking rates, or removing grazers entirely (Walker et al. 1981, West et al. 1984, Smeins and Merrill 1988, Brown and Archer 1989, McClaran 2003, Browning and Archer 2011, Allred et al. 2012, Taylor et al. 2012). Similarly, herbaceous-eating herbivores have little direct relevance to the reduction of woody plants and the restoration of grass-dominated ecosystems from woody-dominated ecosystems. In contrast, fire has been shown to consistently limit the recruitment of woody plants (see review in Fuhlendorf et al. 2011), and brush management treatments involving chemical or mechanical techniques remove shrubs and facilitate the restoration of grass-dominated ecosystems on a regular basis (see review in Archer et al. 2011).

Our discovery that grazing is the primary driver of state transitions brings into question just how far the rangeland discipline and ESDs have distanced themselves from the old range assessment model. The succession-retrogression model was derived from Dyksterhuis' (1949) refinement of Clements' (1916) climatic climax plant community concept (NRC 1994). Interpretation of the Dyksterhuis (1949) model led to the notion that rangeland managers could adjust the extent, pressure, or season of use of grazing animals to stabilize any successional stage (West et al. 1984, NRC 1994). One of the primary arguments for switching from the Dyksterhuis model to the new state-and-transition modeling framework was the recognition that the succession-retrogression model prioritized grazing over all other ecological processes and, as a result, was incapable of accounting for many transitions observed in nature (Briske et al. 2003). Yet, the grazing-centric bias of the succession-retrogression model still appears in practice. Our analysis of ESDs shows that rangeland assessment and monitoring continue to focus on grazing as the primary driver of both ecological degradation and restoration. Considering that state changes associated with woody plants define the top two state transitions in ESD state-and-transition models, grazing-induced successional expectations are still being overused to characterize rangeland dynamics in practice.

Some might counter that "proper" grazing supports other management actions and is therefore justified as the most dominant driver of state transitions in ESD state-and-transition models. We use one of the better state-and-transition models in ESDs to showcase an example that was repeatedly observed to address this point. In the Low Stony Hill ESD of Texas (ESD ID: R081BY336TX), continuous heavy grazing and a decrease in the frequency and intensity of fire were referenced as the two major forces driving the increase of woody plants and degradation of the HCPC. Two justifications may be given here to explain the inclusion of grazing as a major driving force. The first is based on an old rangeland management philosophy of - if you have a woody plant problem, you must have a grazing problem. The implication is that "prescribed" or "proper" grazing can be applied to maintain the desired composition of plant species whereas "improper" or "heavy continuous" grazing facilitates woody encroachment. The fallacy with this justification is that it applies the ideology of the rejected succession-retrogression model (shown in Fig. 1). Indeed, long-term increases in woody plants are observed in Low Stony Hill irrespective of grazing pressure by cattle (Smeins and Merrill 1988, Allred et al. 2012, Taylor et al. 2012). The second justification is that grazing decreases herbaceous biomass, which if reduced sufficiently can decrease fire intensity below the threshold required to kill encroaching woody plants (assuming all other factors important to fire intensity are constant). The fallacy with the second justification is that it simplifies fire as a physical and ecological process to the point where only a grazing-mediated pathway influences fire intensity and fire effects on woody plants (Twidwell et al. 2013). Grazing-induced reduction of fine fuels is one of many pathways that influence fire intensity and its effects (Twidwell et al. 2009, 2013). Rather than focusing on fine fuel load, managers can instead target weather-induced reductions in fine fuel moisture to meet restoration objectives (Twidwell et al. 2009, 2013). Yet, only grazing is listed along with fire as a driver of restoration in the above example (as well as in many other ESDs), not precipitation or other factors that also influence fuel properties important to fire intensity.

Unrealistic outcomes of restoration.—One issue that occurred frequently in our review was a nonsensical link from a severely degraded state back to the HCPC. For example, 49% (73 of 148) of ESDs with an exotic-dominated state featured direct transitions from the exotic state back to the HCPC, which did not contain the exotic species. For those ESDs that showed an exotic species could be eradicated, grazing was credited with eradication more than any other driver (Table 4). Such a state change, when coupled with the

Table 4. The number of ESDs showing state changes that result from specific restoration outcomes, the proportion of those ESDs that link the outcome to a given management action, and the proportion of ESDs showing restoration actions are not able to restore the historical reference state.

Description of restoration outcome	No. ESDs	For ESDs showing restoration outcome, % associated with given management action: Grazing	Brush mgt.	Fire	Reestablish native spp.	ESDs showing restoration outcome not feasible (%)
Woody reduction/mortality	208	92.3	98.1	60.1	44.7	0.0
Reverse undesirable shift in native herbaceous composition	142	87.3	60.6	41.5	26.8	8.5
Eradicate exotic herbaceous invader	73	58.9	28.8	17.8	31.5	33.8
Reverse effects of erosion	30	73.3	20.0	0.0	30.0	43.9

management actions or ecological feedbacks associated with the transition, imputes restoration with the wholesale eradication of an exotic invasive species. In reality, no management approach has been consistently credited with the eradication of an exotic species at a spatial scale that corresponds with the ESDs reviewed herein (Sheley et al. 2011). Yet, only one-third (50 of 148) of ESDs with an exotic invaded state showed restoration to the HCPC was not possible (Table 4).

Our evaluation reveals multiple other restoration outcomes in ESD state-and-transition models that are not supported in a recent USDA NRCS funded scientific evaluation of rangeland restoration practices (Briske 2011). ESDs show grazing can reverse the effects of erosion in three-fourths of state-and-transition models and reduce woody plants more than fire and nearly as effectively as brush management techniques (Table 4). In fact, fewer ESDs list restoration as infeasible (by showing no pathway back to the historical plant community) than the proportion of ESDs that show grazing could eradicate exotic invaders, reverse the effects of erosion, reduce woody plants, or reverse undesirable shifts in native herbaceous species composition (Table 4). No other management action can make this claim (Table 4), and it remains unclear how grazing can produce these outcomes. Rarely are any details given in ESDs that describe how different approaches to grazing may be responsible for desired shifts between alterative states. In the absence of such details, these findings further support our premise that the grazing-centric succession-retrogression model that has been discounted over the past several decades remains a dominant part of modern rangeland assessment in ESDs.

It is important to note the inherent difficulty in evaluating the restoration outcomes in ESDs. For example, brush management is linked to the reversal of undesirable shifts in native herbaceous species composition in a large proportion of ESD state-and-transition models (Table 4), but this is probably because undesirable herbaceous shifts occurred concomitantly with increasing woody plants in 52% (107 of 209) of ESDs that featured brush management as a driver of state transitions. Like any other model, state-and-transition models serve to reduce ecological complexity to a manageable level. However, state-and-transition models in ESDs characterize a single type of state change (e.g., grassland to woodland) as a function of multiple simultaneously occurring ecological drivers (e.g., grazing, brush management) and multiple simultaneously occurring ecological changes (e.g., woody reduction, erosion). Such an approach fails to qualify the relative importance of each driver or the impact of their interactions, causing the effects of multiple ecological processes associated with a single type of state change to be indiscernible (e.g., state-and-transition models in ESDs model simultaneous changes in woody abundance and shifts in native herbaceous composition, typically without qualifying how brush management causes this change relative to other drivers). Evaluation of state-and-transition models in ESDs is therefore more difficult since little to no information is given on the rates of change between states as a result of restoration actions, the magnitude of restoration intervention needed to induce a state change, or the context in which restoration actions are to be applied. We acknowledge these

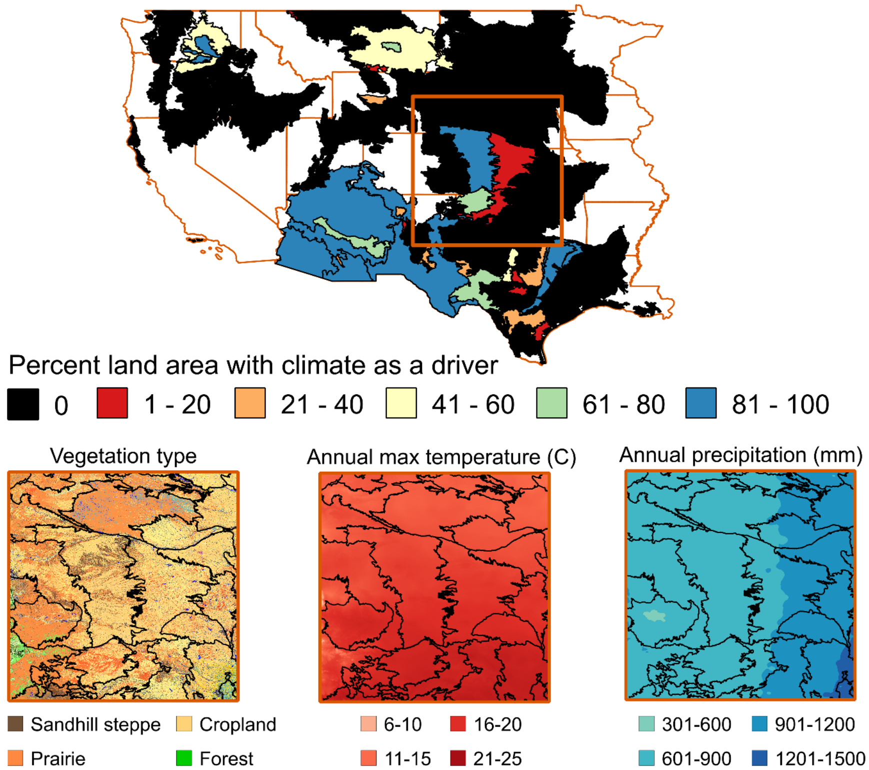

Fig. 11. The percentage of land area within major land resource areas (MLRAs) where climate is featured as a driver of state transitions in the Ecological Site Description Database. The boxed area illustrates the inconsistent use of climate among MLRAs with similar vegetation types and climate. Vegetation type retrieved from LANDFIRE (2012); annual maximum temperature and annual precipitation retrieved from PRISM (PRISM Climate Group, Oregon State University, http://prism.oregonstate.edu, created 4 Feb 2004).

limitations to illustrate how restoration pathways need further consideration and revision when the state-and-transition model concept is applied in the future.

The climate conundrum.—At multiple spatial scales, ESD state-and-transition models produce odd and confusing patterns on the role of climate in vegetation change. Across the western half of the U.S., the inclusion of climate as a driver of state transitions varies widely (Fig. 11). Overall, few ESDs include climate as a driver of state transitions (Table 2; Fig. 8). Instead, most ESDs only include climate data in the climate major section. While this approach helps to define the spatial extent of ecological sites, it also causes the vast majority of ESDs to assume climate will not change sufficiently over time to create vegetation change. Even in the near future, this assumption does not reflect the scientific consensus (Oreskes 2004, IPCC 2007, Brysse et al. 2012). More regionally, at the scale of MLRAs, adjacent land areas with similar ecological states vary widely in their use of climate in ESDs. For example, climate is listed as an important driver of state transitions in western Oklahoma and the Texas panhandle but is given minimal or no attention within ESDs in adjacent MLRAs (Fig. 11).

The climate conundrum in ESDs reveals the underlying subjective nature of the models and the pitfalls of depending on the expert opinions from professionals scattered in field offices throughout the country. The high degree of subjectivity may be an inadvertent consequence of the inconsistent application of the state-and-transition model framework (see *Objective 2*). However, state-and-transition models are also susceptible to being representative of the biases and knowledge limitations of an individual developer or organizing agency rather than the spectrum of drivers that have transformed rangeland ecosystems in the past, present, and future. Here, the flagship agency for rangeland management, the USDA NRCS, has produced a state-and-transition model-driven database that lists grazing as the dominant driver of ecological degradation and restoration in rangelands. Yet, livestock production has dominated the history of the rangeland discipline and the USDA-NRCS is considered to be a grazing-centric organization (Fuhlendorf et al. 2012). Similarly, one might expect a fire-based organization to list fire as the dominant driver, a herbicide-based organization to list the spraying of herbicide as the most important driver, a climate-focused organization to list climate as most important, and so forth. Under the current expert-opinion based framework, state-and-transition models can be developed and used to justify the mission, funding needs, and actions of any developing entity. A major challenge for the advancement of systems ecology is to objectively characterize how ecological processes interact in space and time to drive transitions in state-and-transition models rather than arbitrarily selecting ecological processes perceived to be important by an individual or organizing agency. If ESDs are to move toward a more objective foundation for characterizing ecological states and transitions, the expectation should be to insist that transitions between alternative states are based on scientific evidence when such information exists (and it should be explicitly stated when developers suspect no scientific evidence exists). Developers should cite specific outcomes from ecological field experiments when identifying transitions between alternative states and assess the degree of similarity between the field experiment and a given ecological site. An additional consideration is to include stakeholders from many different backgrounds throughout the entire model development. This approach is rare. Except in isolated cases, stakeholders with backgrounds that differ from the organizing agency are typically only used as reviewers in post hoc consultations.

Emphasis on the recent past.—A primary demand for ESDs and the initiation of an interagency collaboration is to develop a nationally consistent hierarchical database that could be used for projecting future changes to ecosystems as a result of climate change (Allen et al. 2009, US Forest Service 2010). However, state-and-transition models in ESDs mostly characterize ecological states and transitions that have occurred in recent memory. Developers of ESDs are asked to "describe the most common, predominant, and/or ecologically significant states..." (RIESM 2010:3). Over the last several decades, woody encroachment has emerged as one of the leading drivers of the degradation of modern utilitarian preferences in rangelands throughout the world (Scholes and Archer 1997, Asner et al. 2004, Van Auken 2009). ESD state-and-transition models are consistent with this recent change. Over two-thirds of ESDs show transitions associated with changes in woody plant abundances (Table 2). Similarly, ESD state-and-transition models readily incorporate new states to explain changes resulting from the establishment and invasion of exotic species. Yet, one can envision how cultivation or conversion to cropland or pastureland would have been included in more state-and-transition models had they been created fifty years earlier (30% of current ESDs include ecological states associated with agricultural or pastoral conversion). From 1880 to 1969, agricultural and pastoral conversion was the leading reason for the loss of two-thirds of rangelands (US Bureau of the Census 1975), but the amount of area in rangelands stabilized over the last 50 years (Nickerson et al. 2011). Moreover, based on the considerable amount of research devoted to the ecological effects of climate change, one would expect climate change to be featured as a more dominant driver of state transitions if greater emphasis was placed on changes that are expected to occur in the near future (81% of the area covered in current ESDs do not explicitly include climate change as a driver of state transitions; Fig. 11). Even though climate-driven transitions cannot be inferred from any existing ecosystem, state-and-transition models can incorporate such transitions using empirical manipulations or simulation models that provide reasonable first-order forecasts of the response of vegetation to global warming, increased precipitation variability, or CO_2 enrichment.

A single historical equilibrium.—The presence of a single historical point of reference and the focus on managing for that reference state shows theories of community climax continue to be at

Table 5. USDA NRCS characterization of management actions in ESD state-and-transition models, mean annual USDA NRCS conservation program expenditures for each management action, and mean annual land area treated by the USDA NRCS using each management action.

Management action	ESDs featuring management action as driver of state transition		Annual conservation expenditures		Annual land area treated	
	No.	Percentage	$	Percentage†	ha	Percentage†
Grazing	268	79	9,364,843	24	6,536,543	67
Brush management	209	61	25,450,791	66	1,590,489	3
Fire	235	69	417,781	1	123,957	1
Reseeding natives	131	39	2,752,753	7	103,976	1

† Percentages are based on total mean annual conservation expenditures from 2005–2009 and total mean annual land area treated from 2004–2008 (total mean annual USDA NRCS program expenditures were $38,446,155 to treat 9,657,813-ha). In addition to the management actions listed in the table, USDA NRCS also tracks expenditures and land area treated on riparian cover and upland wildlife habitat; however, specific management actions for riparian and wildlife habitat management are not known. Riparian cover cost $4,693 to treat 3,156-ha; upland wildlife habitat management cost $455,294 to treat 2,572,082-ha. Data are from Tanaka et al. (2011).

the core of rangeland management, even though it has long-been long challenged in the ecological literature. For all ESDs, a single ecological state is denoted as the historical point of reference (labeled as the historical climax plant community, HCPC). The single HCPC is considered to be the plant community that occurred at the time of European settlement of North America (ESD User's Guide 2011:43). According to the ESD User's Guide (2011:48), land managers are to restore the ecological state that occurred prior to crossing a threshold, making the state designated as the HCPC the ultimate target for ecological restoration and for management against ecological degradation. As a result, land managers are using disturbance agents (e.g., grazing, fire, brush management) to manage for a single, desired reference community and often attempt to minimize spatial and temporal heterogeneity (Fuhlendorf et al. 2012). The designation of a single historical reference point ignores many other ecological states that were important components of the pre-European landscape and may be critical to sustainability and biodiversity. Consequently, managing for a single HCPC has the potential to guide management toward one plant community within land areas that can be greater than millions of hectares in size (e.g., Fig. 6). While such an approach has the potential to be successful under a utilitarian paradigm focused on livestock production, it fails to simultaneously manage for multiple ecosystem services and is a leading reason for the loss of species endemic to rangeland ecosystems (Fuhlendorf et al. 2009, 2012).

Objective 4: Evaluate whether conservation-funding priorities are consistent with ESD output

NRCS conservation expenditures and the amount of private land treated with specific management actions do not consistently match the proportion of ESDs that feature those management actions as drivers of state transitions. ESDs are developed with the justification that they are to guide land management decisions (RIESM 2010). One should therefore expect NRCS conservation program expenditures and total land area treated to be congruent with the relative importance of management actions in ESDs. Grazing and brush management are the two management actions that most closely matched conservation applications to their relative involvement in ESDs (Table 5). Grazing management practices, which occur as drivers in ESD state-and-transition models more than any other management action, have been prescribed on greater than two-thirds of the land and appear to be the most cost effective of all management actions (Table 5). While brush management is a focal point of the NRCS, its use as a conservation practice is economically impractical and has limited impact at a national level. Brush management accounts for two-thirds of all NRCS expenditures but has been applied on only 3% of the total land area (Table 5). Based on the relative importance of fire and reseeding in ESDs, both management actions are underused as conservation practices. Fire, the second most frequent driver of state transitions in ESDs, accounts for only 1% of conservation expenditures and land area treated by the NRCS (Table 5). Similarly, reseeding accounts for a small proportion of total expenditures and has been applied on less than 1% of all land area (Table 5).

Conclusions

Our assessment is the first broad-scale evaluation of the application of the state-and-transition concept in the Ecological Site Description (ESD) Database, which is set to become one of the world's largest terrestrial management frameworks. While the development of ESDs began 15 years ago and continues today, our evaluation shows ESDs are subject to many of the same criticisms used to justify the move from the old range model (West 1982, West et al. 1984, West and Hassan 1985, Wilson 1989, Laycock 1991, NRC 1994). ESDs are highly subjective, fail to meet the goal of a nationally consistent assessment procedure, focus on a single historical climax community, overuse grazing as a driver of ecological degradation and restoration, are an extension of preceding rangeland assessment procedures and ideologies (Fuhlendorf et al. 2012), and have been established as a national framework (US Forest Service 2010, RIESM 2010) without a national-scale critique of their application and limitations. In addition, new criticisms of ESDs have emerged that are associated with the lack of spatiotemporal considerations (Bestelmeyer et al. 2011), the presence of multiple impractical restoration outcomes, the inability to use ESDs to project state transitions that will be important in the future, and the mismatch between conservation funding of management actions relative to the importance of those management actions in ESDs. While some criticisms of ESDs can be resolved in a short time frame, many will require major reconstruction beyond the current revisions outlined in latest development manual (RIESM 2010).

The disconnection between ecological science and management is a problem that goes well-beyond rangelands and is likely at the very core of our inability to meet sustainability targets throughout all ecological specializations. Management for food, fiber, water, disaster avoidance, and biodiversity in the fields of conservation, agriculture, and environmental engineering are more closely linked to concepts of equilibrium and a naïve sense of super-control over nature than management is linked to modern scientific perspectives of hysteresis, resilience, panarchy, and complex adaptive systems in systems ecology. Management based on modern scientific perspectives would strive to capture the natural variation of ecosystems and promote ecological processes that drive system variance rather than relying on human technologies to control ecosystems at an idealized equilibrium (Holling and Meffe 1996). This requires a fundamental shift in society wherein humans allow or even encourage ecological processes to create spatially and temporally variable patterns across broad landscapes (Turner 2010). As our assessment demonstrates, qualitative, expert opinion-based management frameworks meant to embrace the dynamic behavior of nature will not force a paradigm shift toward a more dynamic platform for natural resource management. Such an approach is not a substitute for management frameworks with a scientific foundation that emphasizes clearly testable predictions and quantitative reasoning (Twidwell et al. 2013). The challenge for scientists is to characterize the dynamic nature and resilience of ecological systems in a manner that can be readily incorporated into management-oriented frameworks. The challenge for managers is to develop management-oriented frameworks that are adaptable to scientific advances, include readily testable predictions, and can be improved with scientific experimentation—yet can guide management decisions in the absence of detailed knowledge of how an ecosystem operates. Our ability to meet these challenges and bring the scientific and management communities together is critical if natural resource management is to more effectively address current and future sustainability issues facing society.

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Supplemental Material

Appendix

Here we provide an example calculation of the percentage of land area (A) a given factor (shown here for grazing) was characterized for the five largest ecological sites occurring within all major land resource areas (MLRAs).

First, recall the equation used to calculate A:

$$A = \frac{f(\text{area factor characterized})}{f(\text{area completed ecological sites})} \times 100$$

where

$$f(\text{area factor characterized}) = \sum_{i=1}^{487} \sum_{j=1}^{5} M_i \times Apc_{ij}$$

$$f(\text{area completed ecological sites}) = \sum_{i=1}^{487} \sum_{j=1}^{5} M_i \times Ac_{ij}$$

and then let us consider actual data from the five largest ecological sites occurring within three MLRAs (see Table A1 and Fig. A1). Let us assign each of these MLRAs a unique identifier, $i = 1$, 2, or 3, that corresponds with MLRA 133B, 83A, and 83D, respectively. To calculate the percentage of land area across MLRAs that included grazing as a driver of state transitions (A_g):

$$A_g = \frac{M_1 \times Apc_{1,1} + M_1 \times Apc_{1,2} + M_1 \times Apc_{1,3} + M_1 \times Apc_{1,4} + M_1 \times Apc_{1,5} + \ldots + M_3 \times Apc_{3,5}}{M_1 \times Ac_{1,1} + M_1 \times Ac_{1,2} + M_1 \times Ac_{1,3} + M_1 \times Ac_{1,4} + M_1 \times Ac_{1,5} + \ldots + M_3 \times Ac_{3,5}} \times 100$$

which for purposes of this example we can rearrange as:

$$A_g = \frac{M_1(Apc_{1,1} + Apc_{1,2} + Apc_{1,3} + Apc_{1,4} + Apc_{1,5}) + \ldots + M_3(Apc_{3,1} + \ldots + Apc_{3,5})}{M_1(Ac_{1,1} + Ac_{1,2} + Ac_{1,3} + Ac_{1,4} + Ac_{1,5}) + \ldots + M_3(Ac_{3,1} + \ldots + Ac_{3,5})} \times 100$$

$$= \frac{\begin{array}{c} 157,249[(.1436)(1)(1) + (.1206)(1)(1) + (.5503)(1)(1) + (.0335)(1)(1) + (0874)(1)(1)]+ \\ 2,982,154[(.0432)(1)(1) + (.0526)(1)(1) + (.0475)(1)(1) + (.0780)(0)(1) + +(.0931)(1)(1)]+ \\ 729,368[(.0994)(0)(0) + (.0930)(0)(0) + (.2606)(0)(0) + (.0758)(0)(0) + (.0795)(0)(0)] \end{array}}{\begin{array}{c} 157,249[(.1436)(1) + (.1206)(1) + (.5503)(1) + (.0874)(1) + (.1436)(1)]+ \\ 2,982,154[(.0432)(1) + (.0526)(1) + (.0475)(1) + (.0780)(1) + +(.0931)(1)]+ \\ 729,368[(.0994)(0) + (.0930)(0) + (.2606)(0) + (.0758)(0) + (.0795)(0)] \end{array}} \times 100$$

$$= \frac{157,249[.9354] + 2,982,154[.2346] + 729,368[0]}{157,249[.9354] + 2,982,154[.3144] + 729,368[0]} \times 100$$

$$= \frac{147,090.7 + 704,981.2 + 0}{147,090.7 + 937,589.2 + 0} \times 100$$

$$= \frac{852,008.9}{1,084,679.9} \times 100$$

$$= 78.5\% \text{ across these three MLRAs.}$$

Again, note that this example is for three MLRAs, whereas our actual assessment is based on 487 according to:

$$A_g = \frac{M_1(Apc_{1,1} + \ldots + Apc_{1,5}) + \ldots + M_i(Apc_{i,1} + \ldots + Apc_{i,5}) + M_{487}(Apc_{487,1} + \ldots + Apc_{487,5})}{M_1(Ac_{1,1} + \ldots + Ac_{1,5}) + \ldots + M_i(Ac_{i,1} + \ldots + Ac_{i,5}) + M_{487}(Ac_{487,1} + \ldots + Ac_{487,5})} \times 100$$

$$= 88.3\% \text{ across all MLRAs.}$$

Table A1. Data used in the example calculation in the Appendix.

MLRA	MLRA area (ha)	Top 5 ecological sites	Area of ecological sites within MLRA (%)	Grazing	Complete
133B	157249	R086BY211TX	14.36	1	1
		R087AY221TX	12.06	1	1
		R087AY231TX	55.33	1	1
		R087AY234TX	3.35	1	1
		R087AY237TX	8.74	1	1
83A	2982154	R083AY382TX	4.32	1	1
		R083AY396TX	5.26	1	1
		R083AY407TX	4.75	1	1
		R083AY412TX	7.80	0	1
		R083AY629TX	9.31	1	1
83D	729368	R083DY494TX	9.94	0	0
		R083DY495TX	9.30	0	0
		R083DY501TX	26.06	0	0
		R083DY505TX	7.58	0	0
		R083EY702TX	7.95	0	0

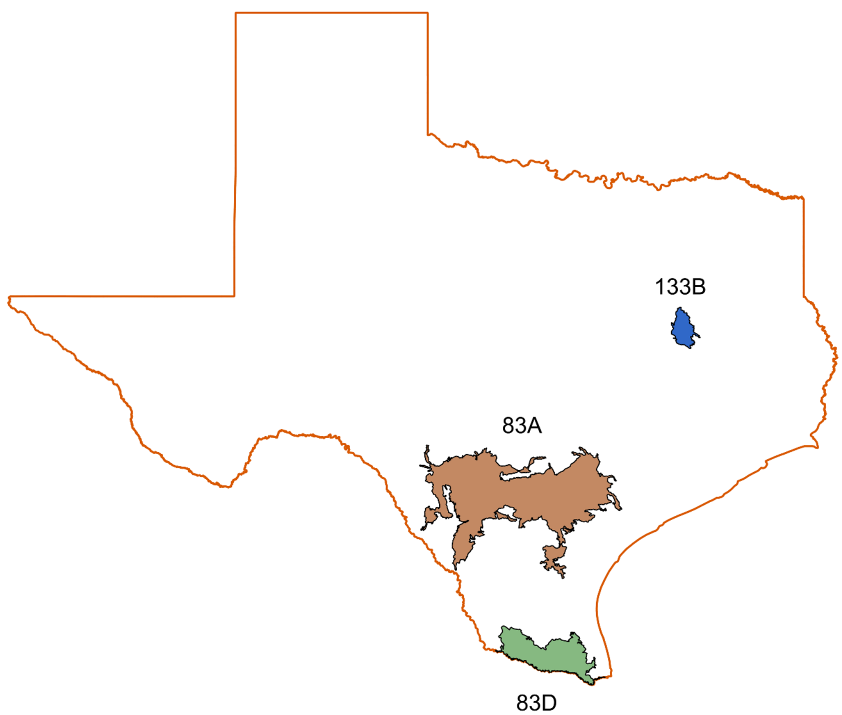

Fig. A1. Locations of the three major land resource areas (MLRAs) selected for the example calculation shown in the Appendix.