# Chapter 9
# State and Transition Models: Theory, Applications, and Challenges

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**Abstract** State and transition models (STMs) are used to organize and communicate information regarding ecosystem change, especially the implications for management. The fundamental premise that rangelands can exhibit multiple states is now widely accepted and has deeply pervaded management thinking, even in the absence of formal STM development. The current application of STMs for management, however, has been limited by both the science and the ability of institutions to develop and use STMs. In this chapter, we provide a comprehensive and contemporary overview of STM concepts and applications at a global level. We first review the ecological concepts underlying STMs with the goal of bridging STMs to recent theoretical developments in ecology. We then provide a synthesis of the history of

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D.D. Briske (ed.), *Rangeland Systems*, Springer Series on Environmental Management, DOI 10.1007/978-3-319-46709-2_9

STM development and current applications in rangelands of Australia, Argentina, the United States, and Mongolia, exploring why STMs have been limited in their application for management. Challenges in expanding the use of STMs for management are addressed and recent advances that may improve STMs, including participatory approaches in model development, the use of STMs within a structured decision-making process, and mapping of ecological states, are described. We conclude with a summary of actions that could increase the utility of STMs for collaborative adaptive management in the face of global change.

**Keywords** Digital soil mapping • Ecological site description • Resilience • State transition • Structured decision-making • Transient dynamics

## 9.1 Introduction

State and transition models (STMs) were conceived as a means to organize and communicate information about ecosystem change as a basis for management. While some authors regard "the state and transition model" as a specific theory about how ecosystems respond to disturbance (see review in Pulsford et al. 2014), we take the view that STMs are not a theory per se, but are a flexible way of organizing information about ecosystem change that may draw on a wide range of concepts about ecosystem dynamics (Westoby et al. 1989). The value of STMs for rangeland managers is in fostering a general understanding of how rangelands function and respond to management actions, thereby leading to more efficient and effective allocation of management efforts.

The fundamental idea is simple (see the Caldenal STM at http://jornada.nmsu.edu/esd/international/argentina). Vegetation, a commonly used indicator of ecosystem conditions, is described according to discrete plant communities (such as an open *Prosopis caldenia* forest with grassy understory). In doing so, we develop a logic for distinguishing different communities so that stakeholders can communicate effectively about them. Next, we describe the multiple plant communities that can occur on a particular site. The key problem in this step is to define the characteristics of the "site"—its climate, soils, and topographic position. Otherwise we might conclude erroneously that a set of plant communities are alternative states of a specific site when in fact they exist on different sites. Finally, we identify the

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causes of transitions between communities and the constraints to recovery of particular communities, including succession, event-driven change, and persistent transitions to alternative stable states (Briske et al. 2003). The causes and constraints to change are often incompletely understood, but they can be tested by monitoring the effects of management and restoration actions.

These steps allow managers to link information about plant community composition collected during inventory with concepts of ecosystem dynamics to develop management plans aimed at long-term stewardship. For instance, management actions may seek to maintain a desired plant community with high forage quality, to restore native plants and animals that formerly occupied the site, or to create a mosaic of different plant communities favoring wildlife. In this way, STMs can help specify management objectives for a site, and serve as guides to maintain and restore ecosystem services.

The diagrammatic and narrative portions of STMs synthesize various sources of knowledge about ecosystem change, including scientific results, historical anecdotes, and local knowledge. The synthesis is used to develop predictions for how ecosystems respond to natural events and management actions (Bestelmeyer et al. 2009b). Conceptual STMs can be expanded into quantitative models by including estimates of the likelihood of change.

Well-developed STMs can serve as a basis for collaborative adaptive management (i.e., management by iterative hypothesis testing, involving multiple stakeholders; Susskind et al. 2012) (Chap. 1, this volume). These guidelines can be updated based on monitoring and new knowledge. In this way, STMs can facilitate a shift from rigid prescriptions based on a one-way relationship between science and management toward a constantly evolving set of recommendations based on collaborative learning and adaptation. Collaborative adaptive management is likely to be more effective than rigid rules of thumb as a basis for environmental stewardship, especially as global climate continues to change.

Because of the potential for STMs to link science to management, they are being developed with increasing frequency in rangelands and other ecosystems on several continents (Hobbs and Suding 2009). While some STMs were never intended to be used for management, others were developed as a basis for outreach and decision support. The linkage of STMs to on-the-ground decision-making, however, remains limited for a number of reasons, including a lack of adequate detail and specificity in STMs and the inability of institutions to develop and use STMs. Moreover, it is inherently difficult to determine the likelihood of transitions, especially given time lags and long timeframes needed to observe some transitions. Nonetheless, there is continued optimism that STMs can provide useful tools for bridging the science-management divide (Knapp et al. 2011b).

Our approach in this chapter is to (1) review the ecological basis for STMs, (2) outline the fundamental components of STMs, (3) review the experiences in several countries with the development and use of STMs (Australia, Argentina, the United States, and Mongolia), (4) identify and address challenges to the use of STMs for management, and (5) describe recent technical advances that may improve STMs, including participatory approaches in model construction, the use of STMs within a structured decision-making process, and mapping of ecological states. We conclude with a summary of strategies to improve the utility of STMs for collaborative adaptive management.

## 9.2 Conceptual Advances in the Ecology of State Transitions[1]

The publications of seminal papers on ecosystem resilience and event-driven vegetation dynamics in rangelands catalyzed a significant shift in thought among scientists and managers beginning in the 1970s (Westoby 1980; Walker and Westoby 2011) (Chap. 6, this volume). Prior to this time, the notions of climax vegetation and orderly succession following disturbance, stemming from early American plant ecology, were used to interpret vegetation dynamics, even in systems where vegetation change is now known to be discontinuous and irreversible (e.g., Campbell 1929). It is now widely acknowledged that (1) vegetation change in response to grazing or weather variations may not occur along a single continuum but rather may produce multiple stable plant communities; (2) vegetation change is not necessarily reversible; and (3) vegetation change can be discontinuous and sudden. While recognition of these patterns occurred prior to the development of STMs, the formalization of "state-and-transition" thinking via the models promoted a broadened view of how vegetation can change (Westoby et al. 1989).

In spite of the impact of STMs on general thought, the continuing challenge is to represent accurately the patterns, timescales, and drivers of change among states in particular settings. To this end, it is important to distinguish transient dynamics from persistent transitions between alternative states (Bestelmeyer et al. 2003; Stringham et al. 2003). *Transient dynamics*, driven by disturbance or weather events, produce significant but temporary changes in vegetation composition or production that can be reversed in a few years to several decades (e.g., via moderation of disturbance, succession, or weather events). *State transitions*, on the other hand, involve persistent changes in vegetation such that recovery of the former state is dependent on unacceptably long recovery times, active restoration, extreme events, or a reversal of climatic change that occurs over several decades or never occurs (Suding and Hobbs 2009). Below, we review the conceptual distinction between these types of dynamics, acknowledging that it may be difficult to distinguish them in practice.

### *9.2.1 Transient Dynamics*

Whether a system undergoes transient dynamics or a state transition following a disturbance is influenced by a variety of factors, including plant traits that evolved in response to disturbance, the ability of alternative plant species to colonize a site, and

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the resistance of soils to degradation (Seybold et al. 1999; Cingolani et al. 2005). For example, in the Chihuahuan Desert where most historical grasslands have converted to eroding shrublands, grasslands dominated by the perennial grass tobosa (*Pleuraphis mutica*) have been comparatively resilient to drought and overgrazing episodes owing to its low palatability, vegetative reproduction via rhizomes that are protected below ground, and its dominance on landforms that receive water runoff and sediment from upslope positions (Herbel and Gibbens 1989; Yao et al. 2006). While disturbances such as continuous heavy grazing can cause significant change in vegetation cover and composition in many rangelands, recovery can be rapid, taking only a few growing seasons in productive settings (Fig. 9.1a), or occur slowly, taking decades in resource-limited environments (Miriti et al. 2007; Lewis et al. 2010). Species having slow recruitment and growth rates may exhibit significant time lags in recovery. Nonetheless, adjustments to the management strategy or disturbance

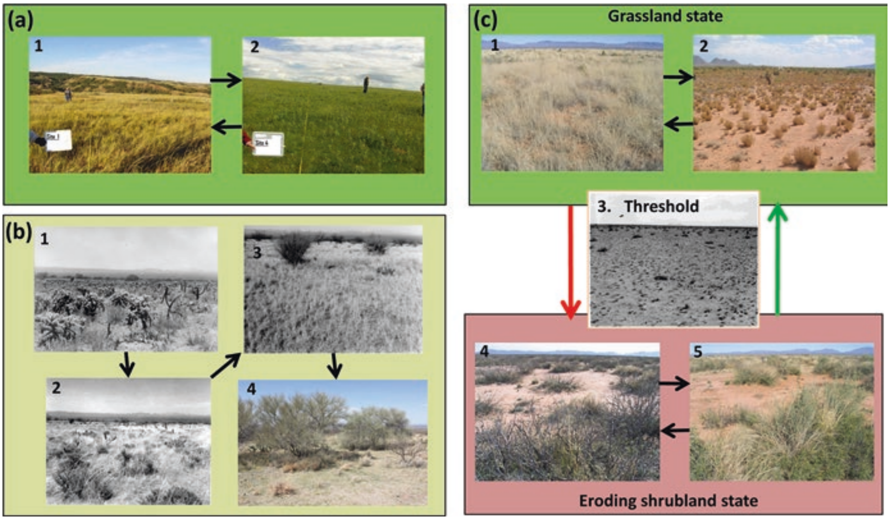


**Fig. 9.1** Examples of transient dynamics and state transitions in rangelands. (**a**) Transient dynamics featuring a reversible shift between communities dominated by western wheatgrass, *Pascopyrum smithii,* (*1*) and blue grama grass, *Bouteloua gracilis* (*2*), in the northern Great Plains of North Dakota, USA; recovery of the more productive *P. smithi* community can occur in several years with changes to grazing management (courtesy of Jeff Printz). (**b**) Transient dynamics on the Santa Rita Experimental Range in the Sonoran Desert of Arizona, USA, starting with cholla cactus (*Opuntia imbricata*) dominance in 1948 (*1*), then burroweed (*Ambrosia dumosa*) dominance in 1962 (*2*), and increasing dominance of blue palo verde (*Parkinsonia florida*) from 1988–2007, which might represent a state transition (*3* and *4*; courtesy of Mitch McClaran). (**c**) A state transition from grassland to shrubland on the Jornada Experimental Range in the Chihuahuan Desert of New Mexico, USA, starting with high cover of black grama grass, *Bouteloua eriopoda*, (*1*) that may be reduced (*2*) and subsequently recovered, unless a threshold is crossed (*3*) after which *B. eriopoda* goes extinct and mesquite (*Prosopis glandulosa*) dominates (*4*). This change results in an eroding shrubland state that experiences infrequent co-dominance by another perennial grass, *Sporobolus* spp. (*5*), during periods of high rainfall

regime (e.g., via reduced stocking rates or reestablishment of natural fire disturbance regimes to remove woody plants) can be used to initiate recovery.

Weather variations are especially important causes of transient dynamics in rangelands featuring high interannual rainfall variability. For example, high winter precipitation initiates recruitment of burroweed (*Isocoma tenuisecta*) in the Sonoran Desert and burroweed dominance can be sustained for one to two decades until dry periods and senescence cause declines in density (Fig. 9.1b; McClaran et al. 2010; Bagchi et al. 2012). Such vegetation changes can be abrupt, but they do not necessarily represent a transition between alternative stable states. This is because vegetation change is predictably related to recent environmental conditions and it can be reversed via plant senescence or subsequent, common weather events (Jackson and Bartolome 2002; McClaran et al. 2010).

### *9.2.2 State Transitions*

The hallmark of a state transition (sometimes referred to as a "regime shift"; Scheffer and Carpenter (2003)) is long-term persistence of new plant communities, or a new range of variation among plant communities that differs from that of the previous state. The persistence of new states can be caused by mechanisms that are internal to the ecosystem, such as competitive dominance of invaders or plant-environment feedbacks favoring new species under the same soil and climate conditions. In addition, directional changes in external environmental drivers, such as climate change, can cause the persistence of new states.

State transitions in rangelands have been described with the following sequence in some STMs produced in the United States (Fig. 9.1c). Weather variations or disturbances can cause transient dynamics within a historical or "reference" state resulting in two (or more) distinct communities (Bestelmeyer et al. 2003; Stringham et al. 2003). Certain of these communities may have low resilience and be susceptible to a state transition (called an "at-risk community"; Briske et al. 2008). Recovery to communities less likely to undergo a transition can occur with the return of favorable weather or reduced disturbance frequency or intensity. Alternatively, an intensification of adverse weather or disturbance can cause the plant community to cross a threshold (often called a "tipping point") into a new state. The new state may be stable with respect to the dominance of key plant species, but still exhibit transient dynamics among a set of plant communities that did not exist in the previous state (Friedel 1991).

The persistence of alternative states can be caused by invaders that are superior competitors when given a foothold in a community (Seabloom et al. 2003). Alternatively, the cessation of natural disturbances can lead to the dominance of superior competitors. For example, the cessation of fire in prairie grasslands can lead to increases in woody plant density and size. When the density of woody plants limits grass (and fuel) continuity and fire spread, and when woody plants grow to a size that limits their mortality in response to fire, then reintroduction of fire can no longer recover the grassland state (Twidwell et al. 2013b) (Chap. 2, this volume). These two types of state transitions involve changes in dominant plants, but not necessarily a change in plant production or other ecosystem properties. While production and soil carbon levels may be maintained (or even increased) with such transitions (Barger et al. 2011), the provision of other ecosystem services (e.g., forage for livestock production) is often reduced (Eldridge et al. 2011) (Chap. 14, this volume).

Plant production can be reduced when the loss of dominant perennial plants leads to a reduction in soil water infiltration, accelerated erosion that reduces soil fertility, or rising water tables resulting in salinization (D'Odorico et al. 2013). In arid and semiarid rangelands, there may be thresholds in plant patch organization below which positive feedbacks between plant patches, resource acquisition, and plant survival and reproduction break down, resulting in a persistent low-productivity/high bare ground state (Kéfi et al. 2011). In other words, if larger plant patches become fragmented too much, the plants occupying those patches suffer due to increased soil erosion and decreased resource availability (Svejcar et al. 2015). State transitions associated with soil degradation are often called "desertification."

State transitions often have multiple, interacting causes (Fig 9.2; Walker and Salt 2012). *Drivers* that are external to the system can cause a gradual or abrupt change in *controlling* (or "slow") variables. The controlling variables directly determine the *state* variables of interest. An example would be a change in the intensity and duration of grazing periods (the driver) that gradually reduces grass root mass, basal cover, and soil organic matter (controlling variables) to affect plant foliar cover, production, and composition (state variables). *Triggering events* occurring over relatively short time

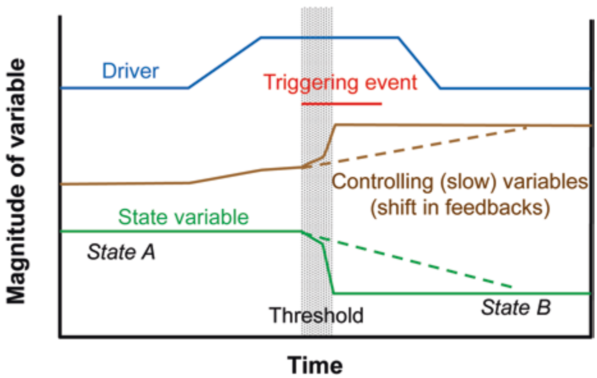


**Fig. 9.2** A schematic illustrating the pattern and interaction of variables over time involved in state transitions. Elements include external drivers that are returned to pre-transition levels, discrete triggering events occurring over short periods that exacerbate the effects of changing drivers, responses of internal controlling (a.k.a slow) variables that may exhibit feedbacks with state variables, and transitions in state variables. The position of different states and the threshold between states are noted. *Dashed lines* indicate that changes in controlling and state variables need not be abrupt

periods, such as an extreme drought, can amplify the rate and magnitude of change in controlling variables and may produce abrupt changes in state variables.

Abrupt changes can occur in both transient dynamics and state transitions. In state transitions, however, *feedbacks* among controlling variables and state variables can lead to persistence of the new state. For example, reduced plant cover leads to increased soil erosion and reduced litter inputs, accelerating the loss of soil organic matter and the ability of the soil to store moisture. Modifications to one or more feedbacks can produce an abrupt change in state variables (i.e., community structure and composition) to create an alternative state, even after the driver has returned to previous levels. The *threshold* between states is the period in time when changes in controlling variables, and possibly feedbacks, lead to persistent changes in state variables.

State transitions need not always be abrupt, however. Abrupt changes in controlling variables can cause strongly lagged, nearly linear responses in state variables. For example, long-lived plants can persist long after the environment required for their establishment has disappeared, leading to a gradual transition after the threshold is crossed. Alternatively, controlling variables may change gradually and be tracked by gradual changes in plant composition, such as with climate change (see the dashed lines in Fig. 9.2; Hughes et al. 2013). Even irreversible state transitions can occur gradually.

### *9.2.3 Distinguishing Transient Dynamics from State Transitions*

The criteria used to distinguish transient dynamics from state transitions depend on the length of time needed for recovery and the implications of these timelines for management. Recovery that does not occur within an acceptable management timeframe without intensive effort is often categorized as a state transition (Watson and Novelly 2012). What is deemed "acceptable" varies among users and contexts, but should ideally be based on measurable recovery criteria. For example, recovery that takes longer than 3 years following a change in grazing management is treated as a state transition in Mongolia by the Mongolian government (National Agency for Meteorology and Environmental Monitoring and Ministry of Environment 2015). For the US government, changes are called state transitions when they are irreversible or take "several decades" for recovery of the former state (Caudle et al. 2013).

It is also important to realize that the type of dynamics recognized might depend on the specific plant functional groups considered. For example, in the Calden (*Prosopis caldenia*) forests of central Argentina, herbaceous plants can exhibit transient dynamics and be managed over multi-year timescales (Llorens 1995), even as gradual shrub and tree encroachment over decades increasingly constrains herbaceous cover and composition, representing a state transition (Dussart et al. 1998). Unlike the simpler models of the past, transient dynamics and state transitions can be represented simultaneously in STMs.

## 9.3 Development of State and Transition Models

STMs should be designed to serve land managers and policymakers by: (1) communicating locally relevant indicators of transient dynamics and state transitions and their consequences for ecosystem services; (2) describing the drivers and environmental conditions affecting susceptibility to transitions; (3) recommending management to avoid undesirable transitions (i.e., resilience management) and to obtain desired ecosystem services; and (4) identifying realistic restoration or adaptation options for alternative states (Bestelmeyer et al. 2009a).

Assembling the evidence to support an STM can be accomplished in most cases using a combination of sources. Monitoring data, historical records, comparisons of plant communities and surface soil characteristics among sites with different management histories, published experiments, and local knowledge can be combined to infer vegetation dynamics (Bestelmeyer et al. 2009b). In any case, it is important to recognize that the dynamics represented in STMs are hypotheses that should be tested through the outcomes of management decisions.

The structure of STMs represented in the literature to date is highly diverse. Different authors have used different conventions to develop model diagrams and narratives. Models can be entirely qualitative/descriptive (Knapp and Fernandez-Gimenez 2009; Kachergis et al. 2013), quantify only properties of states (Bestelmeyer et al. 2010; Miller et al. 2011), or quantify states and/or transitions (Jackson and Bartolome 2002; Czembor and Vesk 2009; Rumpff et al. 2011). Across all model types, however, there are a set of common elements that define an STM.

### *9.3.1 Define the "Site"*

An STM should focus on the alternative states and dynamics of an environmentally uniform area (Peterson 1984). STMs focus on temporal dynamics, so inclusion of significant ecosystem differences due to inherent differences in soil or climate confuses space and time and may lead to flawed interpretations. In rangelands and forests, terrestrial land units such as ecological sites or potential vegetation types approximate areas of environmental uniformity and can define the spatial extent of individual STMs (Bestelmeyer et al. 2003; Yospin et al. 2014). Attempts to define STMs at too fine a spatial scale, however, may result in an unwieldy number of STMs and make comparisons among environmental contexts difficult. For this reason, STMs can be developed at a relatively broad spatial extent, such as a landscape, and the effects of varying soil and climate context within the landscape can be described as a narrative for transitions. Grouping land areas according to "disturbance response groups" in the northwestern USA similarly seeks to produce more general STMs that sacrifice spatial precision for greater efficiency of development and use (T. K. Stringham, pers. comm.).

### 9.3.2 *Define the Alternative States*

Each state that is possible for a site is described. In some instances, plant communities linked via transient dynamics are represented as “states” in the broad sense (Jackson and Bartolome 2002; Bagchi et al. 2012) and in other cases, alternative stable states in the narrow sense are emphasized and transient dynamics within states are described separately or ignored (Miller et al. 2011).

Descriptions of transient dynamics have been based on differences in species composition of plant communities that are relevant to management, such as grazing use or wildlife habitat value. Descriptions of alternative states tend to focus on the relationships of vegetation structure to the processes maintaining that structure, such as erosion, fire frequency, or nitrogen fixation (Petersen et al. 2009; Kachergis et al. 2011). Some STMs depict both alternative states and transient dynamics within states by using boxes for plant communities and separating certain communities using irreversible transitions across a threshold boundary, signifying a state transition (Oliva et al. 1998). US agencies developing STMs for Ecological Site Descriptions (see Sect. 9.4.3; Fig. 9.3) identify

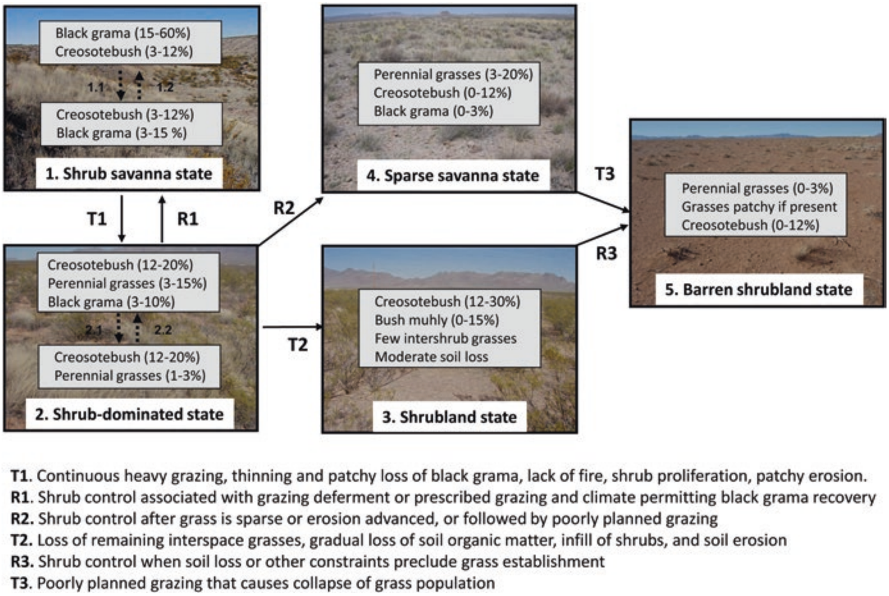


**Fig. 9.3** An example of an STM developed for the Gravelly ecological site, including soils that are loamy-skeletal Haplocalcids and non-carbonatic Petrocalcids in the 200–250 mm precipitation zone of the Southern Desertic Basins, Plains, Mountains Major Land Resource Area (MLRA 42) of New Mexico and west Texas, USA. Following conventions used by US federal land management agencies, rapidly reversible community phases are small boxes whereas states defined by important management and ecological thresholds are defined by large boxes. Each phase is characterized by foliar cover values for dominant or key plant species or functional groups that distinguish it from other phases. In the abbreviated narrative, *T* signifies an unintentional transition whereas *R* signifies a transition caused by restoration action (that can have unintended consequences)

transient dynamics among communities within a state (called "community phases") as smaller boxes connected by reversible arrows, that are nested within larger boxes representing alternative states (USDA Natural Resources Conservation Service 2014).

Each community or state is typically given a narrative to describe its characteristics and, in some cases, the important ecosystem services it provides. Numerical values allow quantitative distinction of states (Fig. 9.3). It is useful to describe the management actions or natural processes that maintain or weaken the resilience of each state and the conditions characterizing low resilience (Standish et al. 2014). Alternative states may exhibit variations in resilience, such that undesirable shifts can be avoided (Briske et al. 2008) and opportunities for restoration toward desirable states can be exploited (Holmgren and Scheffer 2001).

### *9.3.3 Describe Transitions*

Each transition, represented by arrows, is given a narrative. Transient dynamics are typically attributed to perturbations such as grazing or fire, rainy periods or droughts, or to succession. As described in Sect. 9.2.2 and Fig. 9.2, state transitions can be described using four basic elements. First, the mechanisms causing a shift among states are described, including external drivers or triggering events, changes in controlling variables and feedbacks, and indicators of change based on controlling variables (e.g., evidence of soil erosion) or state variables (changes in plant composition). Timelines for transitions can be described, such as whether they are gradual or abrupt relative to management timeframes. Second, the constraints to recovery of the former state can be described (sometimes referred to as a threshold), including how altered feedbacks or environmental conditions preclude the appearance of some plant communities. Third, strategies for the reversal of transitions through restoration actions can be described. Fourth, context dependence in space (such as soils or climate) or time (such as weather conditions) that affects the likelihood of undesirable transitions or restoration success can be described.

## 9.4 Development and Applications of STMs in Rangeland Management

Although many STMs have been created, four countries have produced groups of STMs to support rangeland management. How these efforts originated and progressed (or didn't progress) provide important lessons for future efforts. Below, authors familiar with the history of STM development in Australia, Argentina, the United States, and Mongolia offer accounts representing a variety of global contexts.

## 9.4.1 Australia[2]

### 9.4.1.1 History

Australia was an early adopter of STMs, particularly in their application to rangeland management. This early interest stems from two developments. First, Australian rangeland ecologists were at the forefront of considering how concepts of nonequilibrium dynamics and thresholds were applicable to the management of arid rangelands (Westoby 1980; Friedel 1991). Second, unlike the United States where formal monitoring of rangelands had been instituted based on the range condition and trend concept (Dyksterhuis 1949; Shiflet 1973), Australia had no single or dominant institutionalized model for rangeland monitoring and, consequently, a number of approaches were developed (e.g., Watson et al. 2007).

The absence of a widely accepted framework for describing plant community dynamics in Australia, coupled with the appeal of the state-and-transition format, led to keen interest from the rangeland research community. Adoption was particularly rapid in tropical Australia where the research and management of tropical grazing lands was moving away from a long phase of pasture agronomy associated with the use of introduced species to one based on sustainable utilization of the largely intact, native savannas (Ash et al. 1994; Brown and Ash 1996). STMs provided an effective approach for describing the dynamics of many plant communities in tropical rangelands. This resulted in a special edition of the journal Tropical Grasslands on STMs (Taylor et al. 1994).

In addition to providing qualitative STMs for the major plant communities used for livestock production across northern Australia, the journal issue raised a number of concerns about the broader use of STMs in rangeland management. Major concerns included strategies for communication using models and their role in management; the ability (or inability) to define quantitatively both states and transitions for specific plant communities; and incorporation of spatial processes, such as water flow (Brown 1994; Grice and MacLeod 1994; Scanlan 1994). Shortly afterward, Watson et al. (1996) questioned the strong focus on event-driven processes and abrupt change and suggested that a model of more continuous, cumulative change was just as appropriate to describe vegetation dynamics in many systems. Further, they suggested an emphasis on the management of vegetation within an ecological state to either prime it for a desired transition or protect it from an undesired transition.

Acceptance of STMs was also evident in southern Australia, such as the original bladder saltbush model used by Westoby et al. (1989), as well as in arid rangelands, particularly where piosphere effects can lead to alternative vegetation states within a management unit (Hunt 1992). A strong interest from ecologists in the fragmented and remnant temperate woodlands drove further conceptual development of STMs,

[2] Primary authors are A. Ash and J. Brown.

primarily in the context of restoration (Price and Morgan 2008; Hobbs and Suding 2009; Rumpff et al. 2011).

#### 9.4.1.2 Current Applications

The early interest in developing and applying STMs was not followed by a well-resourced or formal approach to embedding STMs in management of rangelands used for livestock grazing. STM development was carried out via research projects, or by informal approaches in land management or extension agencies, often driven by enterprising individuals, but rarely through systematic institutional initiatives.

One of the limitations in using STMs has been a robust approach to defining states and the thresholds between states. There is a lack of quantitative data for the majority of plant communities and descriptions of dynamics have tended to be qualitative. Moreover, defining and applying threshold concepts in practical management can be problematic because of the potential misinterpretation of management needs (Bestelmeyer 2006). Thus, a quantitative basis for distinguishing state transitions from transient dynamics is immensely important.

There were early efforts in Australia to describe transitions quantitatively using Markov models (Scanlan 1994). The use of Bayesian belief networks to better incorporate uncertainty and expert knowledge has provided an improved conceptual basis for defining states (Bashari et al. 2009) but to date has had limited application. Other approaches included simulation/scenario modeling based on historical rainfall, understory grassland growth, and utilization rates by livestock (Hill et al. 2005). While the simulation modeling approach has proved useful in research for understanding system dynamics, it has not translated well to practical application. Another approach to testing the applicability of the state transition concept is to monitor how frequently transitions are occurring. Watson and Novelly (2012) used an extensive, long-term monitoring dataset from Western Australia to determine how often pre-defined thresholds were crossed. During a 17-year evaluation period 11 % of grassland sites and 1 % of shrubland sites were judged to have undergone a transition. More recently, a study in semiarid wetlands in Australia provided a robust approach for quantifying the causes of state transitions and using logistic models to generate future transition scenarios (Bino et al. 2015). While there has been a range of quantitative approaches tested, a consistent, structured approach to defining and testing for state transitions is still lacking.

Following the initial interest in the STM approach and continued, sporadic development of models for different plant communities (e.g., Phelps and Bosch 2002), there is little evidence that STMs have been formally incorporated into pastoral management in Australia, by either individual producers or by land management agencies (Watson and Novelly 2012). There is, however, anecdotal evidence that STMs have influenced how rangeland professionals communicate with land managers. One argument is that this “mindset change” is sufficient and that institutionalizing a highly proscribed approach to STMs will stifle flexibility. However, this may be outweighed by the risk of not having a consistent, institutionalized approach to vegetation management in an environment where there are declining resources and capacity in management agencies to proactively assist land managers.

Why has the development of STMs slowed in Australia while it has gained momentum in other countries, most notably the USA? Australia lacks the critical mass of research and extension personnel to develop a comprehensive catalogue of STMs for plant communities at a spatial scale relevant to management. In addition, there is a paucity of robust information on the management-scale distribution of soil properties and accompanying plant community dynamics, exacerbated by the absence of a well-supported and consistent national approach to field-based rangeland monitoring. While that deficiency is being overcome to some extent through a more coordinated national approach to synthesizing information on rangeland condition and trend (Bastin et al. 2009), Australia still lacks a widely applied, ecologically based site classification system such as “ecological sites” in the USA (Brown 2010) which underpins the development of spatially specific STMs.

The lack of formalized STMs does not mean that rangeland management is occurring in the absence of general principles and locally explicit guidelines. Many rangeland professionals working in land management agencies across Australia have been exposed to STMs and either implicitly or explicitly use STM concepts when engaging with producers. In addition, considerable effort has been expended on developing grazing land management education courses for producers, with the most visible example being in northern and central Australia (Quirk and McIvor 2003). However, in an effort to simplify concepts of land condition and its interaction with grazing management, STMs within these educational courses have been replaced by a simple four-level (A[best], B, C, D[worst]) land condition class scheme (e.g., Bartley et al. 2014). While this has been effective as a communication tool, it has tended to de-emphasize the importance of processes responsible for long-term vegetation change. For example, Bartley et al. (2014) showed that even with recommended grazing management practices over a 10 year period, the improvement from class “C” to “B” was proceeding very slowly. This might indicate a state transition related to soil degradation and/or the presence of an exotic grass that was limiting native perennial grass re-establishment. The land condition classes cannot distinguish transient from state transition dynamics or capture the mechanisms involved.

Having been the leaders in the initial development of STMs, rangeland ecologists and land administrators in Australia should consider how development of STMs has progressed elsewhere in the world to see what innovations in application might be relevant to Australia. Recent approaches provide useful frameworks for incorporating STMs into practical management (Bestelmeyer et al. 2009b; Suding and Hobbs 2009). These frameworks go well beyond the development of STMs themselves to include aspects of empirical data to support development of STMs, monitoring protocols, and adaptive management. Ultimately, success will be judged by the utility and relevance of STMs to rangeland managers.

## *9.4.2 Argentina*[3]

### 9.4.2.1 History

Interest in STMs began in the early 1990s following the publication of the seminal paper by Westoby et al. (1989). STMs were motivated in large part by the need for a new framework to describe plant community dynamics. A series of STMs developed for the arid Patagonian region were the earliest examples (Paruelo et al. 1993). Models for arid environments usually involved the effects of grazing, initially causing a loss of palatable grass species but eventually causing a reduction in total grass cover associated with increasing bare soil and erosion rates. Following these models, a decline in plant cover results in a reduction of soil water holding capacity and plant production, causing a feedback to water and wind erosion that further inhibits reestablishment of grass species (Cesa and Paruelo 2011) (Chap. 3, this volume). State transitions were regarded as irreversible or difficult to reverse. This sequence corresponds to most STMs developed for the Patagonian steppe in Paruelo et al. (1993).

STMs were developed for more humid environments later in the 1990s, including montane grasslands (Barrera and Frangi 1997; Pucheta et al. 1997), Pampean grasslands (Aguilera et al. 1998; Laterra et al. 1998; León and Burkart 1998), and herbaceous vegetation of the Caldenal/Espinal ecoregion (Llorens 1995). These STMs emphasized changes in species composition rather than large decreases in total plant cover. In these models, grazing did not produce noticeable changes in soil physical properties through erosion as observed in the Patagonian region because total plant cover is usually not greatly reduced by grazing.

A third type of STM described state transitions in "mallines," a local name for meadows with high productivity and biodiversity within the Patagonian steppe, and which are an important source of forage for livestock (Paruelo et al. 1993). Overgrazing and trampling by livestock in mallines produces a transition to an alternative state due to the loss of plant cover that promotes increased runoff and/or soil salinization. Increased runoff and erosion result in gully formation. Consequently, altered hydrology causes a shift in plant communities. Similar hydrologically based state transitions are observed in alluvial floodplains of the Chaco region (Menghi and Herrera 1998).

In contrast to early expectations, these STMs had little impact on science and management in Argentina. Exploring the reasons why interest and activity waned may provide insights for improving the usefulness of the STM framework in Argentina and elsewhere. First, STMs developed in the 1990s did not feature adequate detail. STMs described drivers associated with transitions but provided little description on processes and mechanisms controlled by the drivers. Narratives did not contain information on thresholds and processes controlling the functions of alternative states (i.e., feedbacks). Transitions identified in these models were rarely

[3] Primary authors are D. Lopez and R. Peinetti.

experimentally tested (López 2011). Most models were superficial representations of community dynamics that did not provide useful predictions.

Second, STMs were used to synthesize general regional information on ecosystem dynamics but lacked the site specificity needed for practical applications. They contained few recommendations on management practices or restoration actions to reverse undesirable transitions. Similar to Australia, the lack of a land classification system tied to STMs such as ecological sites (Bestelmeyer et al. 2009b) led to confusion about the spatial domain to which a particular model applied.

Third, important types of state change were simply not addressed by existing models. Tree and shrub encroachment and "thicketization" of woody plants represents one of the most important kinds of state change occurring in several ecosystems in the central and northern parts of the country (Brown et al. 2006). The thicketization of forests and grasslands has received a great deal of attention in basic and applied sciences (e.g., Dussart et al. 1998), including information about management practices, but there have been very few cases in which this understanding was incorporated in STMs.

Finally, there have been few incentives for scientists to expand development of STMs. Modern Argentinean ecological science, as directed by funding and reward systems over the last few decades, has been focused on short-term studies that yield rapid publication and career advancement (Farji-Brener and Ruggiero 2010). In this environment, there was little incentive for integration across different case studies at a regional level or long-term studies to support STM development.

#### 9.4.2.2 Current Applications

There is a substantial demand from society for responsible natural resource management, in part due to the alarming deforestation rates of the last 10–15 years in the semiarid and humid forests of Argentina (Gasparri et al. 2013). Societal demand for rational management forced the establishment of new federal regulations on the use of natural resources. To apply these regulations, policymakers have recognized that a new suite of management decision tools and a basis for assessment and monitoring are required, leading to a renewed interest in STMs manifest in the recent Argentinean Rangeland Congress in 2013 (http://inta.gob.ar/documentos/jornadas-taller-post-congreso-argentino-mercosur-de-pastizales-cap2013). At this meeting, there was general consensus among participants that STMs associated with ecological site concepts should be explored as an option to organize the available information under a common framework for both rangelands and forests. A research network was proposed. It is hoped that this network will serve as a platform for interactions between different research groups and thereby stimulate the production of systematically structured STMs and ecological site classifications across the nation. As yet, funding the network and motivating coordination among researchers via a network remains a significant challenge.

### 9.4.3 *United States*[4]

#### 9.4.3.1 History

The official adoption of STMs in 1997 as a component of land evaluation can be considered a paradigm shift in US rangeland science. Clementsian, or succession-based, concepts of community dynamics originating in the early twentieth century provided acceptable explanations for observed vegetation changes in rangelands, particularly in response to livestock grazing. Succession concepts embodied in the "range succession" or "range condition" model (Westoby 1980; Joyce 1993) worked fairly well in highly resilient prairie ecosystems where much of the grazing livestock and conservation efforts were concentrated. Even leading proponents of the range condition model (Dyksterhuis 1958; Passey and Hugie 1962), however, noted that the scope of this model was limited to forage for domestic livestock and climax plant communities dominated by perennial, herbaceous species.

In spite of these caveats, use of the range condition model spread throughout US rangelands and was linked to evaluation procedures and financial and technical assistance from federal land management agencies. A relatively well-trained and mature workforce able to detect discrepancies between model predictions and actual conditions, and make ad hoc adjustments to management prescriptions (Shiflet 1973), created a sense of complacency among adherents (Joyce 1993). Strong connections among universities, agencies, and managers strengthened the ability of the rangeland profession to adapt to these inconsistencies (Svejcar and Brown 1991). However, as the application of the range condition concepts spread into diverse rangeland settings, such as those experiencing long-term shrub encroachment, significant limitations in model application became apparent.

As the applicability of the range condition model began to be questioned, theoretical ecologists were developing alternatives to the Clementsian model to explain how ecosystems, and specifically rangelands, behave (Holling 1973; May 1977). The multiple stable state model was less deterministic than the range condition model and multiple trajectories were possible, better matching observations of rangeland change. Soon afterward in the 1980s, concern about the appropriateness of range condition as a universal metric of rangeland function surfaced within US land management agencies. The inability to link non-forage values to the range condition model was now recognized as a major limitation of assessment procedures (Society for Range Management 1983). By the end of the decade, there was widespread dissatisfaction with the application of the range condition model to all rangeland ecosystems (Lauenroth and Laycock 1989; Pieper and Beck 1990).

In this context, the impact of the first publication on STMs (Westoby et al. 1989) was rapid and substantial. Following this paper, there was a flurry of experimental and review papers exploring the application of STMs to particular rangeland ecosystems, both within and outside of the USA. Federal land management agencies

[4] Primary authors are J. Brown and P. Shaver.

undertook extensive reviews of the use of the range condition model as a basis for technical and financial assistance versus implementation of an STM-based approach, culminating in publications by the US National Research Council (National Research Council 1994) and the Society for Range Management (Task Group on Unity in Concepts and Terminology Committee Members 1995). The two reviews called for standardization of rangeland evaluation approaches and replacement of the range condition model with a model that could account for multiple stable states. Different plant communities could have distinct values to society and call for different management approaches, but a primary focus was to preserve "site potential"—the option to sustain desired plant communities and services—by avoiding accelerated soil erosion.

These two reviews were catalysts for adoption of STM concepts by natural resource agencies. Beginning in late 1990s, STMs began to be developed and used by rangeland specialists, primarily those associated with USDA Natural Resource Conservation Service (NRCS), for communication with ranchers about management needs and to provide guidance in administering federal financial assistance. The policy implications of the latter led to a systematic approach to STM development within NRCS. Widespread development of STMs, however, was delayed because they had to be linked to "ecological site descriptions" (ESDs; formerly called "range site descriptions"). ESDs are documents that had long served as the site-specific basis for management recommendations by federal land management agencies. ESDs are linked to soil survey databases through the connection of ecological sites to soil maps maintained by the NRCS. Application of the range condition model via ESDs involved the calculation of plant community similarity between an observed and a single, historical climax plant community identified for each ecological site (Dyksterhuis 1949). In order for the rangeland condition model to be replaced, thousands of STMs would have to be developed for ecological sites across the USA, each requiring the description of multiple plant communities.

#### 9.4.3.2 Current Applications

Acceleration of STM development represents a major logistical challenge because of the large number of STMs needed, particularly in the eastern half of the USA. Added to these logistical concerns, there has been a lack of clear institutional guidance on how to structure STMs that were developed in the late 1990s and early 2000s. Few agency employees have been dedicated to ESD and STM development, and in some locations, contracts were awarded to private enterprises to work on STMs. In most locations, however, STM development was an added duty for existing federal agency staff. Much of this work, although creative, lacked coordination. In some cases, transitions featuring overwhelming indicators of persistence were presented as transient dynamics following the range condition model. In other systems that feature transient behavior, community variations were presented as alternative stable states. The resulting inaccuracies in some STMs have elicited criticism of how they are produced (Twidwell et al. 2013a).

In spite of these problems, STMs have gained greater visibility and are increasingly viewed as useful tools for communicating research and management recommendations. New definitions of STM components, scale considerations, and a greater variety of ecosystem attributes linked to STMs (Briske et al. 2008; Bestelmeyer et al. 2010; Holmes and Miller 2010) have emerged. Systematic approaches to the development, evaluation, and refinement of STMs (Bestelmeyer et al. 2009b; Bestelmeyer and Brown 2010), informed by the successes and limitations of early model development efforts, have been incorporated in recent US government guidelines (Caudle et al. 2013; USDA Natural Resources Conservation Service 2014). These comprehensive guidelines address priority setting, resource allocation, and progress reporting. They also incorporate recent scientific literature, diverse agency policies, and user needs. Nonetheless, significant challenges remain, particularly (1) funding and expertise required to accelerate STM development and deliver STMs to the public, (2) inclusion of information pertaining to ecosystem services other than livestock production, such as climate change mitigation and adaptation, hydrology, and species of conservation concern, (3) how to make STM development more participatory and inclusive to support adaptive management, and (4) how to address the impending effects of climate change in models developed with a high degree of spatial specificity (Knapp et al. 2011b; Twidwell et al. 2013a). Current NRCS and interagency efforts are focused on these concerns.

### *9.4.4 Mongolia*[5]

#### 9.4.4.1 History

Mongolia is dominated by rangelands, and livestock production is a critical component of the national economy and cultural traditions. Nonetheless, Mongolia never adopted well-defined or universally accepted rangeland evaluation concepts or procedures. The shift from a nomadic or transhumant, subsistence herding system into a market economy in 1993 led to dramatic increases in livestock numbers and loss of herder mobility (Fernández-Giménez 2002). The perception of widespread rangeland degradation associated with overgrazing (Bruegger et al. 2014; Hilker et al. 2014) motivated interest in rangeland evaluation and monitoring procedures. A systematic approach was needed because assertions about rangeland degradation have been challenged within the Mongolian government and the broader academic community (Addison et al. 2012), creating conflict about the need for interventions to reduce stocking rates versus calls by some officials to encourage larger livestock numbers. Beginning in 2004, Green Gold Mongolia (GG), a project funded by the Swiss Agency for Development and Cooperation, initiated efforts to build a national capacity for reporting on the present state and future trend of Mongolian rangelands. In addition, GG sought to develop tools to facilitate rangeland

[5] Primary authors are B. Densambuu and B. Bestelmeyer.

management at local, regional, and national levels. Following exposure to the concept of ESD-based STMs from US scientists and land managers in the mid-2000s, GG and its government partners undertook an effort to develop ESDs for Mongolia.

#### 9.4.4.2 Current Applications

In many ways, the relatively recent Mongolian experience with STMs takes advantage of what was learned in the early development efforts of Australia, Argentina, and the USA. STM development began in concert in 2008 with the development of a standard methodology for vegetation measurement, based on procedures used by US government agencies (Herrick et al. 2005). These procedures were officially adopted by the Mongolian government in 2011. In addition to providing a sound basis for reporting trends in rangeland vegetation, adoption of a unified measurement method ensured that cover and production values reported in STMs were comparable to monitoring data produced by the National Agency for Meteorology and Environmental Monitoring (NAMEM). Training for a GG research team on methods to develop STMs and database management began in the USA in early 2009, followed by data collection co-occurring with training in Mongolia from 2009 to 2014. Following recommendations adopted by US agencies (Bestelmeyer et al. 2009b; Knapp et al. 2011a; USDA Natural Resources Conservation Service 2014), inventory of vegetation and soils was conducted at over 600 sites across Mongolia, coupled to workshops aimed at eliciting local knowledge about reference conditions, the presumed causes of vegetation change, and to identify informative sites for inventory. These data are the basis for STMs that were included as a report for the Mongolian government in 2015 (https://www.eda.admin.ch/content/dam/countries/countries-content/mongolia/en/Mongolia-Rangeland-health-Report_EN.pdf).

A National Ecological Site Core Group was established in 2011 composed of experienced plant community ecologists representing different ecoregions across Mongolia as well as decision-makers of key institutes able to develop shared interpretations of inventory data. The National Core group (1) reviews published materials to establish reference conditions and causes of state change, (2) works in close collaboration with the GG research team in developing STMs, and (3) performs outreach to encourage adoption of materials by local government and herder cooperatives.

Because of the magnitude of the project, the limited budget, and the need for landscape-level information matched to herding and transhumance patterns, the decision was made to produce broad-level concepts for ecological sites based primarily on landforms and large differences in soil texture or hydrology. These classes, called Ecological Site Groups (ESG), combine finer-level soil classes that are equivalent to ecological sites in the USA (e.g., Moseley et al. 2010). STMs are developed for each ESG, resulting in 3–5 STMs per ecoregion and 25 total STMs for Mongolia (http://jornada.nmsu.edu/files/STM_Mongolian-catalogue-revised_2015.pdf). Because vegetation dynamics do not differ strongly across ecological sites within an ESG, the general models are deemed adequate for evaluation and management recommendations.

The specification of rangeland management strategies to maintain or recover perennial grasses is a primary objective of the STM development effort. In most of the sites sampled, the presence of well-distributed, remnant perennial grasses suggests that plant community recovery could occur in a few years to several decades with changes to grazing management (Khishigbayar et al. 2015). Thus, STMs are being designed to contain detailed information about recommended stocking rates and grazing deferment periods, tailored to the objectives of either maintaining a state or recovering a former state. Recommendations and expectations are linked to specific vegetation cover indicators that can be monitored.

In addition to their use as rangeland management guides by local governments and herder groups, STMs are being embedded in the activities of two government agencies. NAMEM has responsibility for monitoring 1550 plots across Mongolia to report on national rangeland trends. A lack of information about reference conditions and trends in monitoring data has precluded clear statements about rangeland health. Based on STMs drafted for most common rangeland communities in different ecoregions of Mongolia, NAMEM was able to conclude, preliminarily, that Mongolian rangeland communities are in general altered from historical reference states but that relatively rapid recovery was possible in the majority of cases.

STMs can provide a link between monitoring interpretations and management recommendations at the local level. The Agency for Land Affairs, Geodesy and Cartography (ALAGAC) is responsible for land management planning and its implementation nationally. STM concepts are being integrated into participatory rangeland management plans in several pilot areas. These pilot programs will provide a test of the value of the information content of STMs and therefore how they should be refined. As of 2015, expectations are high. Herder groups are using maps based on STMs (including information about recent forage availability and desired community change) to plan grazing and resting periods. It is encouraging that STMs are being used as a basis for such specific management actions.

### *9.4.5 Summary of STM Applications*

The cases described above suggest that major efforts to develop STMs have taken different trajectories following the introduction of the concept in 1989. In Australia and Argentina, initial enthusiasm and progress was not sustained due to limitations in the data available to develop STMs, the dearth of land classification systems as a basis for STMs, and lack of resources and incentives for scientists and managers. In the USA, these limitations were overcome to varying degrees by the linkage of STMs to rangeland evaluation systems and financial assistance programs supported by government agencies. The vast scientific and administrative infrastructure provided by well-funded US government agencies has supported the nationwide development of numerous STMs. While this strategy has dramatically accelerated STM development compared to Australia and Argentina, it also introduced logistical difficulties associated with managing such a large number of STMs.

The Mongolian effort takes advantage of recent advances and lessons learned. STM development there was motivated by national concerns over rangeland degradation that attracted international development support. A dedicated team of scientists worked with government agencies to develop a relatively simple land classification system as a basis for STMs and employed a broadly collaborative approach to develop STMs. Furthermore, the STMs and related educational materials were purpose-built for collaborative rangeland management at broad spatial scales characteristic of transhumant and nomadic grazing systems of the country. The Mongolian experience may provide a useful model for STM development efforts for many parts of the world.

## 9.5 Knowledge Gaps[6]

The limitations to STM use highlighted above and recent evaluations of STMs in the USA (Knapp et al. 2011b; Twidwell et al. 2013a) suggest several overarching challenges that must be addressed in order to develop more useful STMs and better employ them for management. Below, we describe the main challenges and strategies for responding to them.

### *9.5.1 Reference States, History, and Novel Ecosystems*

STMs, such as those used in the USA and Mongolia, often define a reference state that represents historical or a "healthy" set of ecosystem conditions for society, such that a primary goal of management is to maintain the reference state or to restore it (Fulé et al. 1997; Stoddard et al. 2006). Reference states are usually ascertained using historical information or measurements gathered in areas that have not been transformed relative to historical conditions. In many ecosystems, the societal significance and desirability of the reference state is straightforward when that state is well known and when it supports a set of ecosystem services valued by stakeholders.

In other cases, however, there can be difficulties in identifying a meaningful reference state. Historical conditions may be poorly understood, such that there is controversy about the plant communities present and the nature of disturbance regimes (Whipple et al. 2011; Lanner 2012). This may be especially problematic for plant and animal species that rely on a variety of states (Fuhlendorf et al. 2012). For example, a persistent, low plant cover state associated with prairie dog disturbance is necessary to support some native bird species in shortgrass steppe ecosystems (Augustine and Derner 2012). Thus, areas that may appear degraded to some

[6] Primary author is B. Bestelmeyer.

observers, and with respect to some ecosystem functions, may support biodiversity and valued species.

Furthermore, the recent concept of "novel ecosystems" acknowledges that it may not be practical to target a historical state as a management goal if the likelihood for restoration success is low or the costs high (Hobbs et al. 2009) (Chap. 13, this volume). In such cases, the costs of restoration should be evaluated relative to the ecosystem services provided by different states (Belnap et al. 2012). In some cases, it may be preferable to manage for alternative states. For some scientists, however, evaluations based on ecosystem services rather than historical fidelity are controversial (Doak et al. 2014).

The designation of reference conditions should be based on a broadly collaborative process and take into consideration several factors including history (both recent and evolutionary), the physical processes affecting potential plant communities (climate, soils, and topography), a recognition of specific time scales for disturbance and other processes, practicality of use, and the variety of ecosystem services of interest in particular ecosystems. Similarly, management objectives should be defined in a circumspect and collaborative manner. Managing toward reference conditions may be preferred in some locations, while managing for alternative states may be useful in others.

### *9.5.2 Broader Representation of Ecosystem Services*

Given that STMs are principally used for communication with particular sets of managers, grazing managers for example, they often emphasize a relatively narrow set of ecosystem services (Twidwell et al. 2013a) (Chap. 14, this volume). Minimal recognition of other ecosystem services, including biodiversity and the regulation of water supply, will limit the utility of such STMs for other users. Quantitative interpretations about the different ecosystem services provided by ecological states could be added to STMs (Brown and MacLeod 2011; Koniak et al. 2011). Such information could be used to evaluate the financial costs of restoring a historical state against the change in benefits relative to the current state. Similarly, trade-offs among ecosystem services associated with transitions between states can be communicated in terms of specific variables such as forage provision, species losses, and changes to groundwater recharge rates. As noted above, such exercises may reveal that states considered to be degraded by some observers offer important ecosystem services to others (Mascaro et al. 2012). They may also clarify the trade-offs between specific services, such as forage production vs. biodiversity (Fuhlendorf et al. 2012).

Although it is useful to communicate about states in terms of ecosystem services, it is prudent to acknowledge our limited ability to comprehensively measure all of them effectively. Certain attributes of reference states will be overlooked if they are not adequately measured, especially the biodiversity of organisms that are not the focus of management (Bullock et al. 2011; Reyers et al. 2012). Historical states will continue to be valued for this reason.

### 9.5.3 Climate Change

STMs often implicitly assume that long-term climate properties and potential vegetation are stable (i.e., stationarity). This assumption leads to an emphasis on recent history in designating alternative states (Twidwell et al. 2013a) (Chap. 7, this volume). Given that climate change is likely to cause directional changes in environmental conditions, plant community responses to management observed in the recent past may become less informative in the future. At present, however, forecasts of climate change effects on vegetation, especially at the resolution of STMs, are not well developed (Settele et al. 2014). STMs could benefit from linkages to species distribution models (Bradley 2010) and models examining the role of soil profile properties in mediating water availability (Zhang 2005). Narratives highlighting the consequences of recent extreme events, such as the tree die-off during an extreme drought in the southwestern USA (Breshears et al. 2005), could be readily included in STMs. Particularly in arid rangelands, management strategies aimed at promoting resilience to known extreme events (especially water deficits) would be similar to strategies implemented to adapt to climate change, at least over the next decade or two (Ash et al. 2012).

### 9.5.4 Testable Mechanisms

The inclusion of sufficient detail on mechanisms of vegetation change has been a primary limitation of STMs (Knapp et al. 2011b; Svejcar et al. 2014, Sect. 9.4). For example, transitions in some grassland STMs are sometimes ascribed only to the driver (e.g., continuous heavy grazing) without more detailed analysis of the mechanisms by which transitions occur. Information on plant demography (plant death, lack of recruitment), the timeframe for transitions (1 year or several decades), specific indicators of the risk of transition (reduced reproduction rates, indications of erosion), and the management strategies used to prevent transitions given the processes (proper timing of defoliation to permit successful reproduction during favorable years) are often not described in STMs. Richness of detail may be lacking because (1) the information is believed to be too complicated to include and therefore best left to direct interactions between managers and extension specialists; (2) simple lack of effort on the part of model developers; or (3) a lack of detailed knowledge.

These reasons notwithstanding, model developers should strive to include details in a systematic way (e.g., Sect. 9.3; the Caldenal STM at http://jornada.nmsu.edu/esd/international/argentina) in order for STMs to be used and, more importantly, be tested and improved via adaptive management (Briske et al. 2008; Bestelmeyer et al. 2010) (Chap. 9, this volume). Even when the specific mechanisms of state transitions (or resilience of a state) are not well understood, they can be postulated by blending local knowledge with the rich body of work in ecological science (Kachergis et al. 2013). This can be aided by the development of general STMs at the level of broad ecosystem types that can be refined, if needed, to finer-grained land units such as ecological sites. Analysis of historical treatments and new monitoring data can then be used to revisit the hypotheses. For example, shrub-dominated coppice dune states of sandy soils in the Chihuahuan Desert were believed to resist widespread perennial grass recovery based on historical observations and the notion that high erosion rates precluded grass establishment. An unusual sequence of years with high precipitation, and other poorly understood factors, led to a flush of grass recruitment that was unexpected (Peters et al. 2012). The STM for this system has been modified to include this new information. In this way, STMs can be regarded as theoretical constructs that synthesize what is known, use that knowledge to generate management hypotheses, and are updated as new knowledge is acquired.

### *9.5.5 Information Delivery and Use*

If STMs are to be used as tools for long-term environmental stewardship, then the information presented in STMs must be accessible to land managers and/or become integrated in outreach and management activities. Developing and conveying the information in STMs to users such that they can guide management decisions is a multifaceted problem that should be carefully considered by the institutions developing STMs (and see Sect. 9.4). General approaches to information transfer include (1) collaborative development of STMs that include the managers who will use them (see Sect. 9.6.1; Knapp et al. 2011a), (2) initiation of collaborative adaptive management projects at the scale of landscapes that include STM development and use as key components (Bestelmeyer and Briske 2012), (3) the use of web-based technologies and mobile devices to link users to STMs pertaining to specific localities (Herrick et al. 2013), and (4) the distillation of STM information into simple presentation materials (such as pictorial field guides, web-based materials) and the use of field-based workshops to enable understanding of these materials. The use of STMs for management will require concerted efforts by scientists, government agencies, educators, and technical experts and cannot be limited to the production of reports, publications, and associated databases by a handful of managers and ecologists.

## 9.6 Future Perspectives

Three emerging approaches are currently transforming how STMs are developed and used, including participatory development of STMs with stakeholders as part of community-based management approaches, structured decision-making via STMs, and the use of digital mapping approaches to provide spatially explicit information on ecological states. Here we summarize the current status and future goals of these three approaches.

### 9.6.1 Participatory Approaches to Model Development[7]

Participatory and collaborative STM development approaches emerged for two practical reasons. First, available field data rarely cover the landscape adequately at a sufficiently fine resolution, or over timescales sufficient to detect transitions and calculate their probabilities. Key types and combinations of management and environmental drivers often are not represented in the available data. Second, models based solely on the knowledge of individual scientists or land management professionals may rely too heavily on a single person's observations and experiences, which can result in biases similar to using monitoring data from only a few locations on a landscape or points in time. These limitations suggest that a more inclusive and participatory approach that integrates multiple knowledge sources may be a pragmatic solution to the challenges inherent in STM development (Kachergis et al. 2013) (Chap. 11, this volume).

Perhaps even more important, participatory approaches will increase the utility, credibility, and use of STMs by managers. Recent surveys have shown that many ranchers and natural resource professionals have little knowledge or experience with STMs when they are available (Kelley 2010). Engaging these potential "end-users" of STMs in the process of developing the models increases STM awareness and acceptance, and thus the likelihood that the models will be used to guide and refine management. An acknowledged limitation of many existing STMs is a focus on a narrow set of ecological attributes and management practices to characterize states and transitions, and a limited suite of management interpretations emphasizing livestock production (Sect. 9.5.2; Knapp et al. 2011b). If STMs are to represent multiple ecosystem values and services, and not just changes in vegetation composition and production for a single or narrow range of uses (e.g., forage production), then multiple disciplines and perspectives are needed.

Participatory or collaborative STM development has taken a variety of forms. The most familiar in the USA is the "technical team," an interdisciplinary collaboration of specialists (e.g., rangeland ecology, soils, hydrology, fire, wildlife, geographic information systems, and cultural resources), often involving several natural resources agencies and academic experts, convened to develop STMs for a particular area. In some areas, such technical teams have been expanded to include landowners or ranchers (Johanson and Fernandez-Gimenez 2015). Collaborative STM development usually takes place over a period of months to a few years and may involve multiple meetings and field trips. The "model development workshop" is another type of participatory approach in which a multi-stakeholder group with diverse knowledge and interests in a particular ecological site or set of sites is

[7] Primary authors are M. Fernandez-Gimenez and J. Johanson.

brought together for a single workshop or series of workshops to develop or refine STMs (Knapp et al. 2011a). Such workshops often have an explicit aim to include the local knowledge of long-time residents in an area as well as professional and scientific knowledge. Kachergis et al. (2013) proposed a hybrid approach that involves a diverse set of stakeholders and a combination of literature review, workshops, and field sampling. When it is not possible or practical to bring diverse stakeholders together in one location, or when knowledge documentation is an objective, interviews or surveys with stakeholders can provide a means of recording valuable information that can inform model development (Knapp et al. 2010; Runge 2011).

There is no one best way to facilitate a collaborative or participatory STM development process, but several groups with experience using different collaborative approaches have described the processes that have worked for them (Knapp et al. 2011a; Kachergis et al. 2013; T. K. Stringham, pers. comm.). The process outlined by T. K. Stringham (pers. comm.), which follows the expanded technical team model, focuses on assembling a core team of highly experienced and committed disciplinary experts and inviting participation from a broader group of agency specialists. The workshop model (Knapp et al. 2011a) and integrated literature, workshop, and field sampling approach (Kachergis et al. 2013) draw from a wider array of stakeholders and emphasize the value of including long-term residents and those whose knowledge is derived from land-based livelihoods. All three of these processes begin with a draft graphical model that serves as the basis for initial discussions and feedback from the group.

Johanson and Fernandez-Gimenez (2015) drew on these experiences together with those of participants in 16 collaborative ESD and STM development projects in the USA to identify common outcomes, challenges, and keys to success. Most efforts were successful in producing an STM or portion of an ESD. Additional outputs included publications, applications of the models to management, workshops, and databases. Many benefits beyond these tangible outputs were also identified, such as improved working relationships and communication among participants from different organizations, decreased conflict, increased efficiency of STM development, greater use of STMs, and improved data credibility.

Participatory processes are never without challenges. The most frequently cited concerns were related to the quality, diversity, management, and analysis of available data. Reconciling different concepts for classifying ecosystems and their dynamics and agreeing on goals for STM development efforts were common challenges in expanded collaborations. Time and funding constraints and recruitment/retention of participants were additional obstacles. Because many natural resource professionals are unfamiliar with ESDs and STMs, key concepts must be taught to all participants and reinforced with additional teaching throughout the process. Similarly, when working with nontechnical stakeholders, care must be taken to define key terms in a clear and accessible manner and to provide an introduction to STM concepts and applications. Although some professionals express skepticism about the accessibility of STMs to nonprofessionals (Knapp et al. 2011b), we have found that most people readily grasp these concepts, especially once they are engaged in the process of model development.

The keys to successful participatory STM development are similar to those for any participatory natural resource management effort (Wondolleck and Yaffee 2000; Daniels and Walker 2001). First, involve the right people at the right time. Make sure that the needed expertise is present, particularly experienced specialists in soils and rangeland ecology, but also hydrology, fire, wildlife, geographic information systems, and cultural resources. When integrating local knowledge is an important objective, seek diversity and depth of experience in local knowledge holders. Community referrals are often an effective way to identify knowledgeable residents (Knapp and Fernandez-Gimenez 2009; Knapp et al. 2010).

Second, it is important to maintain clear and open communication, a willingness to learn from others, and focus on mutually beneficial outcomes. In multiagency collaborations, conflicts can arise over the differing mandates and procedures of different agencies. When multiple stakeholders are involved, careful facilitation is required to balance power dynamics and ensure that the contributions of all participants are respected. Clear ground rules should be established regarding the criteria for including states and transitions and how potentially conflicting views of ecosystem dynamics will be handled and represented in the model. In multi-stakeholder STM workshops, the level of agreement among participants about each state and transition can be explicitly documented and used to identify uncertainties to test through targeted field sampling or adaptive management experiments (Knapp et al. 2011a; Kachergis et al. 2013). This leads to more efficient use of limited field sampling resources.

Third, support from management within participating agencies is critical. If administrators do not value collaboration and support their staff in participating in such efforts, it is very difficult to sustain the level of participation and commitment needed for success. Fourth, many participants reported that joint field visits were key to successful collaborative STM development. Discussing conditions observed in specific areas can help resolve misunderstandings and elicit new sources of information. Fifth, because many of the challenges identified relate to data collection, management, and analysis, it is important to discuss and agree upon responsibilities and protocols for these activities up front. Often the university or research partners in STM collaborations take the lead on data analysis. However, we strongly encourage groups to invite broad participation in data analysis and especially in data interpretation. We also recommend formal data sharing and use agreements to facilitate information sharing and protect confidentiality where needed.

Reported participant experiences suggest that collaboration is a good investment that increases the efficiency of STM development. It requires significant human, financial, and time resources, but yields both tangible and intangible benefits that participants perceive to increase the quality, credibility, and utility of STMs.

### 9.6.2 Structured Decision-Making via State and Transition Models[8]

In this section, we ask: can STMs be used in a more systematic way to prioritize management objectives and to efficiently allocate management funds? Below we discuss why managers may benefit from integrating STMs into a structured decision-making process, and developing STMs such that they enable quantitative predictions of management outcomes.

Ecosystem management decisions are invariably complex. There may be a lack of understanding about the processes underlying a specific problem. Alternatively, there may be multiple and potentially competing objectives for management, which may not be readily apparent, but which should be determined before developing the model. For instance, when faced with an imperative to both manage for a certain plant community and protect a threatened species, it may be that the habitat for that species does not correspond to the desired vegetation state. In addition, it may be that an objective to minimize costs is at odds with the funds required to restore a community to the desired state. Stakeholders will not value all of these objectives in the same way, but it is the role of the decision-maker to evaluate these trade-offs. Last, there may be multiple potential alternative management strategies, but high uncertainty and disagreement about ecosystem responses to management. For the decision-maker, choosing the best course of action to help achieve the specified objectives can be extremely difficult (Runge 2011; Gregory et al. 2012).

Many of these problems can be addressed by using a systematic approach to the decision-making process. The term "structured decision-making" broadly refers to a framework that incorporates a logical sequence of steps to help decision-makers (1) define their decision context; (2) identify measurable objectives; (3) formulate alternative management strategies; (4) explore the consequences of those alternatives in relation to the specified objectives; and, if necessary (5) make trade-offs among objectives (Gregory et al. 2012). The framework utilizes a broad suite of decision-analysis tools that can aid transparent and logical decision-making (Addison et al. 2013). Despite the multitude of tools and methods that may be applied, the basic premise is a framework that is driven by the objectives, or values, of those involved in the decision-making process (Keeney 1996; Runge 2011).

STMs are typically developed as conceptual models, informed by expert knowledge and existing data. Such models may quantify the characteristics of states but lack a quantification of transition probabilities given particular values of controlling variables and management actions (i.e., they are qualitative or semiquantitative STMs). Within the structured decision-making framework (Fig. 9.4), a qualitative STM can be used to clarify the decision context among stakeholders, the desired direction of change and key attributes of interest (objectives), and the different management interventions that might be employed to achieve this change (alternatives). In addition, qualitative STMs could be used to begin exploring the consequences of

[8] Primary author is L. Rumpff.

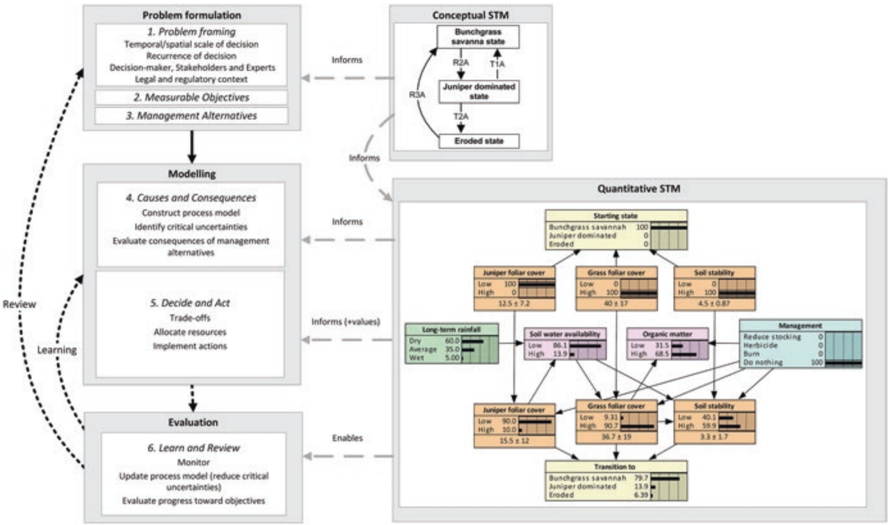


**Fig. 9.4** The structured decision-making framework adapted from Wintle et al. (2011). A conceptual STM (adapted from Bestelmeyer et al. 2010) is commonly used to frame the problem, whereas the quantitative version of the STM (structured here as a Bayesian network) is useful to identify, explore and resolve critical uncertainty

the alternatives in relation to the objectives. As a decision-support tool, a qualitative STM is often all that is required to guide a good management decision within the structured decision-making process. For instance, an STM (based on Bestelmeyer et al. 2010) can be used to identify the interventions required to achieve the ecological conditions for a reference state (Bunchgrass savannah; Fig. 9.4). In this instance there is one objective (the reference state), and clearly defined interventions. However, recognized uncertainty about the effects of climate change may result in different models of cause-and-effect, uncertainty about the most effective interventions, or even uncertainty about whether the goal state is attainable. In cases where there are numerous alternatives to choose from, multiple and competing objectives, conflicting values among stakeholders, differing stories of cause-and-effect, or "critical uncertainty" (i.e., uncertainty that bears on key decisions), decision-making based on quantitative STMs can help select the best decision.

Quantitative (or process-based) models are useful for identifying and exploring the uncertainties that impact management decisions (Duncan and Wintle 2008; Rumpff et al. 2011). A process-based model represents the current state of knowledge and assumptions about the dynamics of the system, and allows predictions to be made about the efficacy of the different management strategies in relation to the objectives of interest. For instance, in Fig. 9.4, the assumptions behind the STM have been quantified and converted into a probabilistic model of cause-and-effect (a Bayesian network). Probabilistic transition estimates now include uncertainty about the efficacy of management interventions under various climatic scenarios.

A management decision will often involve multiple objectives, with no one management strategy that maximizes all objectives. For example, there may be a tradeoff between achieving the reference state and maximizing agricultural productivity. The quantitative model should first be expanded to enable predictions for both objectives. The predictions can then be combined with value judgments (or preferences) that specify which objective should benefit over the other, given the range of possible outcomes (Gregory et al. 2012). The true value of an alternative management strategy is a combination of the consequences (including uncertainty), and the weight or value attributed to the objectives (step 5, Fig. 9.4). At this point, the decision may be obvious, or uncertainty may be obscuring the preferred management strategy.

Uncertainty is inevitable, but decision-makers should pay particular attention to resolving critical uncertainties, as this can result in modified and potentially more effective management decisions. Monitoring is used to resolve this uncertainty, by iteratively updating the knowledge within the process-based model (step 6, Fig. 9.4). This is known as adaptive management, which is a form of structured decision-making, required when decisions are recurrent and hampered by critical uncertainty (Runge 2011). Thus, adaptive management requires extra steps in the structured decision-making framework to provide a plan for motivating, designing, and interpreting the results of monitoring.

Although the development of quantitative state-and-transition models has increased (Bashari et al. 2009; Nicholson and Flores 2011; Rumpff et al. 2011), to date their application in a management context is rare. Thus, it can be concluded that STMs have yet to reach their full potential as decision-support tools for the implementation of natural resource management and the evaluation of its outcomes. Both quantitative and qualitative models can be used to capture our current understanding about system dynamics, and to identify and explore uncertainty surrounding the response to management (Rumpff et al. 2011; Runge 2011). The choice of decision support tool should be dictated by the availability and form of knowledge, whether qualitative or quantitative predictions are required to make a decision, and whether quantitative skills are accessible given the timeframe available for decision-making.

Whether the model is quantitative or qualitative, structured decision-making can help to provide a systematic and transparent framework for identifying objectives, collate existing knowledge, explore the consequences of management alternatives and identify and evaluate uncertainty. The value of qualitative STMs to help frame and guide vegetation management decisions in rangelands is not in question. Rather, managers and researchers should acknowledge the complexities of their particular problem context, and assess whether structured decision-making approaches are useful.

### *9.6.3 Mapping State-and-Transition Model Information*[9]

Managers currently represent ecosystem variations across a wide range of scales for various uses. In rangelands, potential natural vegetation is mapped via land unit classifications such as habitat types (Jensen et al. 2001), range units and range sites (Kunst et al. 2006), and ecological sites (Bestelmeyer et al. 2003). More recently,

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attention has focused on the delineation of the current states of a set of land units based on its STM (Steele et al. 2012). The product is called a “state map” that can make the information within STMs spatially explicit for its use in management.

STMs are typically linked to land units that define the spatial extent to which information in STMs should be extrapolated. Soil survey is often used to map land units bearing distinct STMs (such as ecological sites), particularly in the USA. Hence, STMs can be linked to maps of soil types or landforms. Soil maps thus provide a template for mapping ecological states across multiple STMs. One constraint in linking soil maps to STMs is that any errors in existing soil spatial data are transferred to the state map. Many soil maps in rangelands consist of “soil map units” that represent multiple soil types either due to a limitation of mapping scale or landscape heterogeneity (Duniway et al. 2010). In some cases, soil types are similar and grouped to the same ecological site; however, soil types with contrasting properties combined within the same soil map unit may belong to different ecological sites and STMs. In the USA, it has been a priority to resolve these discrepancies in order to improve the utility of STMs (Steele et al. 2012).

Ecological sites and states can be mapped simultaneously using environmental variables, such as from remote sensing products (Browning and Steele 2013; Hernandez and Ramsey 2013). One benefit of utilizing remotely sensed data to characterize ecological sites and states is the ability to produce scalable information that can be tailored to particular needs (Kunst et al. 2006). For example, West et al. (2005) outlined a strategy for producing a hierarchical map of ecological units for 4.5 million hectares area in western Utah based on a variety of data sources. The finest level was a “vegetation stand” that is similar to ecological states represented in STMs.

Mapping of ecological states can be difficult in rangelands because spectral data from conventional sources, such as MODIS or LANDSAT satellites, is often not of sufficient resolution or quality to distinguish states. Blanco et al. (2014) integrated hyperspectral and multi-spectral remote sensing data to identify ecological sites in rangelands of Argentina. This approach could be extended to map states. Steele et al. (2012) outlined a framework for mapping ecological sites and states in rangelands of southern New Mexico using a combination of soil survey spatial data combined with image interpretation of aerial photography to manually delineate ecological site and state polygons (i.e., line maps).

Digital soil mapping (DSM) is an emerging technique that can improve estimates of soil property and ecological state information at fine spatial scales in rangelands by predicting the properties of pixels of varying resolution (e.g., to 5 m) (Levi and Rasmussen 2014; Nauman et al. 2014). Although DSM has not yet been applied to state mapping, it could fill a much needed gap by increasing automation, using a greater range of data sources, and allowing for rapid updating of state maps when new data become available. Data-driven classification algorithms can greatly reduce the time needed to produce state maps because they provide a means of grouping pixels into similar units, thereby reducing the burden of hand digitizing (Laliberte 2007; MacMillan et al. 2007). DSM approaches can also be scaled up or down to meet desired management objectives, which is currently difficult to do with polygon-

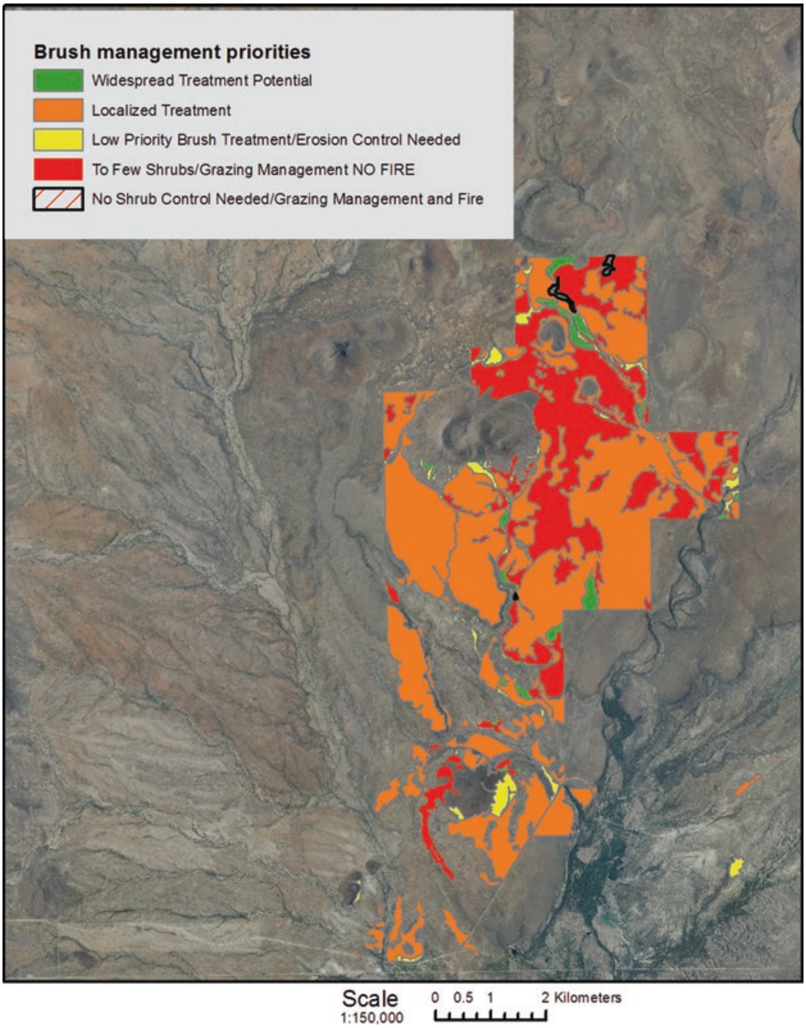


**Fig. 9.5** An example of a product based on an ecological state map for a single ecological site type. The map illustrates interpretations of an STM according to brush management treatment options (courtesy of Eldon Ayers)

based maps. In turn, DSM could be used to identify vegetation responses to soil properties that may improve STMs (Browning and Duniway 2011).

State mapping can extend the utility of STMs for management. In landscapes with a mix of ecological sites and states, state mapping distills information across multiple STMs into a simpler classification scheme that can be used for communication among stakeholders and to develop action plans (Fig. 9.5). For example, a state map was used in the southwestern USA in planning for brush control treatments to identify areas that were (1) near a desired reference condition where no treatment was needed, (2) areas that had experienced soil erosion where treatment would likely not produce increases in perennial grass cover, and (3) areas where treatment would be most likely to produce desired changes. In a similar way, state mapping can be used to plan for land use changes, such as by prioritizing development away from desirable reference states (Stoms et al. 2013). State mapping could also be used to visualize or model spatial interactions in a landscape, such as where increases in grass cover would have the greatest impact on water retention within a watershed.

## 9.7 Summary

STMs evolved from the recognition that vegetation change was more complex than could be accounted for by succession alone, and could occur along numerous pathways, be discontinuous, and result in multiple stable states in the same environment. Conceptualizing vegetation as discrete states also provides a useful platform for tailoring management actions to the properties and possibilities associated with each state. For rangeland managers, the value of STMs resides both in their flexibility for organizing information and in their ability to foster a general understanding about how rangelands function.

Progress toward developing rangeland STMs at a global level has been uneven due to several factors, including limitations of data and fiscal and personnel resources. As strategies to overcome these limitations are developed, the ultimate success of STMs as management tools will require careful attention to several topics. First, there should be a clear understanding of the characteristics of alternative states, including a reference state where such a concept is meaningful. Field sampling, synthesis of experimental results and long-term vegetation records, and participatory approaches are important resources for defining states. State characterization should ideally represent information on a variety of ecosystem services. In most cases, this will require coordinated sampling efforts to link variations in plant community states to empirical or model-based evaluations of habitat quality, soil carbon storage potential, and value for livestock, for example.

Second, STMs should attempt to distinguish transient dynamics from state transitions. Evidence-based approaches necessitate clear statements not only about drivers of transition but also about the controlling variables and processes constraining recovery and timelines for ecosystem change. STMs should feature logical and testable statements about how states will respond to management, such that STMs can support experimentation, quantitative models, and eventual revision. Even where data are scarce, local knowledge can be framed as testable propositions. Predictions regarding the effects of climate change on ecosystems may best be addressed at a regional scale, but information on the impact of past extreme events can be highlighted. Strategies to manage alternative states, such as through novel uses of states invaded by woody plants, may help with climate adaptation over the longer term.

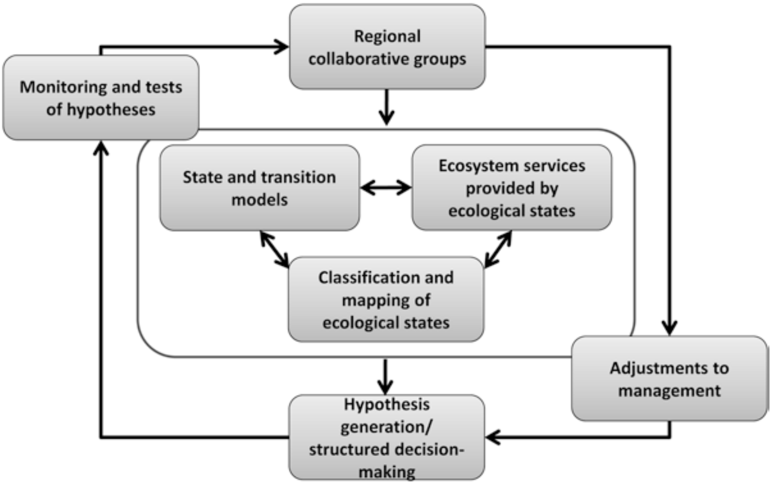


**Fig. 9.6** A schematic of how STMs can be used in collaborative adaptive management, adapted from Bestelmeyer and Briske (2012)

Third, STM development programs should consider how to make information available, useful, and believable to users. Participatory approaches can promote understanding and acceptance of STMs. There should be a clear link between STMs and specific management actions, which can facilitate the inclusion of STMs into collaborative adaptive management programs supported by local communities, non-governmental organizations, or governmental agencies (Fig. 9.6). Regional or landscape collaborative groups can develop STMs and identify ecosystem services of interest from different states. The linkage of STMs to maps of ecological states can facilitate management application and testing. Hypotheses for management responses can be developed for specific land units (Fig. 9.5) and structured decision-making approaches can be used for cases when multiple management options are possible, trade-offs make decisions difficult, and the preferred decision is unclear or controversial. Tests of hypotheses via monitoring can be used to either revise the STM or make minor management adjustments.

In order to facilitate their use in collaborative adaptive management, STMs should be presented and used in a variety of ways, including simple extension materials, formal hypotheses for ecological research and tests of management efficacy, rangeland evaluation criteria, maps, or Bayesian models. Policymakers, technical assistance personnel, regulators, scientists, land managers, and stakeholders should be working from the same general understanding of how a rangeland ecosystem functions, even if those parties differ in their preferred states or ecosystem services. STMs should link understanding across different organizational levels as a basis for collaborative adaptive management. Our hope is that the recommendations presented here will promote development of STMs that are indispensable for the management of global rangelands.

**Acknowledgements** We thank David Briske, Ian Watson, and an anonymous reviewer for comments that substantially improved this chapter. B.T.B. was supported by appropriated funds to the USDA Agricultural Research Service and by the Jornada Basin LTER program (DEB-1235828). M.E.F.-G. was supported by Colorado Agricultural Experiment Station Project COL00698A and NRCS Conservation Innovation Grant Agreement No. 69-3A75-12-213.

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