Chapter 7
Applying Threshold Concepts to Conservation Management of Dryland Ecosystems: Case Studies on the Colorado Plateau

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Abstract Ecosystems may occupy functionally distinct alternative states, some of which are more or less desirable from a management standpoint. Transitions from state to state are usually associated with a particular trigger or sequence of triggers, such as the addition or subtraction of a disturbance. Transitions are often not linear, rather it is common to see an abrupt transition come about even though the trigger increases only incrementally; these are examples of threshold behaviors. An ideal monitoring program, such as the National Park Service's Inventory and Monitoring Program, would quantify triggers, and be able to inform managers when measurements of a trigger are approaching a threshold so that management action can avoid an unwanted state transition. Unfortunately, both triggers and the threshold points at which state transitions occur are generally only partially known. Using case studies, we advance a general procedure to help identify triggers and estimate where threshold dynamics may occur. Our procedure is as follows: (1) Operationally define the ecosystem type being considered; we suggest that the ecological site concept of the Natural Resource Conservation Service is a useful system, (2) Using all available a priori knowledge to develop a state-and-transition model (STM), which defines possible ecosystem states, plausible transitions among them and likely triggers, (3) Validate the STM by verifying the existence of its states to the greatest degree possible, (4) Use the STM model to identify transitions and triggers likely to be detectable by a monitoring program, and estimate to the greatest degree possible the value of a measurable indicator of a trigger at the point that a state transition is imminent

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(tipping point), and values that may indicate when management intervention should be considered (assessment points). We illustrate two different methods for attaining these goals using a data-rich case study in Canyonlands National Park, and a data-poor case study in Wupatki National Monument. In the data-rich case, STMs are validated and revised, and tipping and assessment points are estimated using statistical analysis of data. In the data-poor case, we develop an iterative expert opinion survey approach to validate the degree of confidence in an STM, revise the model, identify lack of confidence in specific model components, and create reasonable first approximations of tipping and assessment points, which can later be refined when more data are available. Our goal should be to develop the best set of models possible given the level of information available to support decisions, which is often not much. The approach presented here offers a flexible means of achieving this goal, and determining specific research areas in need of study.

Keywords Monitoring · State and transition model · Tipping point · Expert opinion · Alternative stable state · Dryland · Ecosystems · Assessment points · Delphi method

Introduction

Threshold concepts are used in research and management of ecological systems to describe and interpret abrupt and persistent reorganization of ecosystem properties (Walker and Meyers 2004; Groffman et al. 2006). Abrupt change and the progression of reorganization can be triggered by one or more interactive disturbances such as land-use activities and climatic events (Paine et al. 1998). Thresholds occur when feedback mechanisms that typically absorb forces of change are replaced with those that promote development of alternative equilibria or states (Suding et al. 2004; Walker and Meyers 2004; Briske et al. 2008). The alternative states that arise have reduced ecological integrity and value in terms of management goals relative to the original or reference system. Alternative stable states with some limited residual properties of the original system may develop along the progression after passing a threshold; an eventual outcome may be the complete loss of prethreshold properties of the original ecosystem. Reverting to the more desirable reference system becomes increasingly difficult and expensive along the progression gradient and may eventually become impossible. Ecological-threshold concepts have been applied as a heuristic framework and to aid in the management of rangelands (Bestelmeyer 2006; Briske et al. 2006, 2008), aquatic (Scheffer et al. 1993; Rapport and Whitford 1999), riparian (Stringham et al. 2001; Scott et al. 2005), and forested ecosystems (Allen et al. 2002; Digiovinazzo et al. 2010). They have been applied in contexts varying from ecological restoration (Hobbs and Norton 1996; Whisenant 1999; Suding et al. 2004; King and Hobbs 2006) to ecosystem sustainability (Herrick 2000; Chapin et al. 1996; Davenport et al. 1998) to assessment of natural resource impairment (USDI-NPS 2003).

Achieving conservation management goals requires the protection of resources within the range of desired conditions (Cook et al. 2010; Symstad and Jonas (Chap. 8).

The goal of conservation management for natural resources in the US National Park System is to maintain native species and habitat unimpaired for the enjoyment by future generations. Achieving this goal requires, in part, early detection of system change and timely implementation of remediation. The recent National Park Service Inventory and Monitoring program (NPS I&M) was established to provide early warning of declining ecosystem conditions relative to a desired native or reference system (Fancy et al. 2009). To be an effective tool for resource protection, monitoring must be designed to alert managers of impending thresholds so that preventive actions can be taken. This requires an understanding of the ecosystem attributes and processes associated with threshold-type behavior, how these attributes and processes become degraded, and how risks of degradation vary among ecosystems and in relation to environmental factors such as soil properties, climatic conditions, and exposure to stressors. In general, the utility of the threshold concept for long-term monitoring depends on scientists' and managers' ability to detect, predict, and prevent the occurrence of threshold crossings associated with persistent, undesirable shifts among ecosystem states (Briske et al. 2006). Because of the scientific challenges associated with understanding these factors, the application of threshold concepts to monitoring designs has been very limited to date (Groffman et al. 2006). As a case in point, the monitoring efforts across the 32 NPS I&M networks were largely designed with the knowledge that they would not be utilized to their full potential until the development of a systematic method for understanding threshold dynamics and methods for estimating key attributes of state changes.

This chapter describes a generalized approach we implemented to formalize understanding and estimating of threshold dynamics for terrestrial dryland ecosystems in National Parks of the Colorado Plateau. We provide a structured approach to identify and describe degradation processes associated with threshold behavior, and to estimate indicator levels that characterize the point at which a threshold crossing has occurred or is imminent (tipping points), and points where investigative or preventive management action should be triggered (assessment points). We illustrate this method for two case studies in National Parks included in the Northern and Southern Colorado Plateau I&M Networks, where historic livestock grazing, climatic change, and invasive species are key agents of change. The approaches developed in these case studies are intended to enhance the design, effectiveness, and management relevance of monitoring efforts in support of conservation management in dryland systems. They specifically enhance NPS capacity for protecting park resources on the Colorado Plateau, but have applicability to monitoring and conservation management of dryland ecosystems worldwide.

Background: Threshold and State-and-Transition Concepts

Salient features among frameworks of ecological thresholds include concepts of reference conditions, feedback dynamics, threshold triggers, properties of the progression after a threshold crossing, and changes in restoration potential along this progression. Native or reference conditions, typically, are the desired state for conservation management, and consist of community phases and transitions among phases due to natural disturbances and climate variability. Negative feedback of the reference system confer system resilience and maintain the community phases within a characteristic range of variability. For instance, a negative feedback that inhibits shrub dominance in some grasslands is the interaction between amount of grass cover and fire return interval. Given sufficient grass cover, wildfire events are frequent and large enough to maintain grassland structure due to the selective elimination of fire-intolerant woody plants. Phases comprising the natural range of reference conditions differ in their vulnerability to crossing a threshold. Phases with degraded resilience are more vulnerable and may be described as "at risk" of a persistent transition to an alternative state (Briske et al. 2008). Identifying the patterns that increase vulnerability to change and reasons for these patterns can define preventive-management goals (Bestelmeyer 2006).

Both biotic and abiotic mechanisms may trigger state changes (Beisner et al. 2003; Briske et al. 2006). Biotic mechanisms include altered biotic structure and interactions, such as plant–herbivore interactions. Abiotic mechanisms (e.g., extreme soil erosion) can result in threshold behavior through the modification of inherent site characteristics. A single trigger may initiate a state change, or the temporal order or spatial convergence of multiple triggers may be critical. For example, drought or intensive livestock grazing alone may not trigger a state change, but the two factors in combination or in sequence may trigger such a change through adverse effects of one stressor on ecosystem resilience to the other stressor (Scheffer et al. 2001). Triggers result in conditions that exceed the resilience of the reference system, and lead to an increasing dominance of positive, destabilizing feedback. Triggers often initiate changes in the pattern or spatial structure of an ecosystem (e.g., decreased vegetation cover or increased patchiness) with subsequent and often nonlinear changes in processes (e.g., soil erosion, nutrient cycling; Peters et al. 2007).

The progression resulting from a state change is characterized by increasing dominance of positive feedback, and changes in pattern and processes (Briske et al. 2008). Along this progression is the continual loss of properties of the reference condition. Multiple alternative states, each with their own set of varying community phases, can occur along this threshold gradient with some becoming stable as negative feedback of the alternative state confers resilience. Progression can lead to a degraded state where features of the reference condition are effectively no longer present. Degraded states may no longer afford provision of services such as water, livestock forage production, or desirable recreational opportunities, and may no longer support the biodiversity of native systems.

The potential for restoration to prethreshold conditions is determined by the amount of residual properties of the reference condition and the resilience of the new, alternative state (Suding and Hobbs 2009). Where extensive site preparation and reintroduction of native species are required for conversion to prethreshold conditions, the costs may effectively prohibit restoration. In some cases, complete restoration to native conditions may never be possible due to the extinction of native biota (i.e., species, genomes), or the loss of inherent properties (e.g., soil fertility) necessary to support reference conditions.

Focused study and interpretation of threshold processes and consequences benefit from using conceptual models of ecosystem dynamics. State-and-transition models

(STMs) are a type of conceptual model that have become prominent in rangeland management, and are used to illustrate reference conditions of an ecosystem, ecosystem responses to natural and anthropogenic drivers, and the mechanisms of transition among distinctive assemblages or states of an ecosystem (Bestelmeyer et al. 2003, 2009). These models also provide a basis for discerning levels of system properties indicative of both the risk and occurrence of transition among states (Briske et al. 2008).

Identifying indicator levels indicative of an impending state change is a critical component for the design of effective monitoring. Monitoring efforts should result in alerting land managers of indicator levels in advance of a state change to account for lag-time in decision making and uncertainty in the effectiveness of remediation actions. From a statistical perspective, the number and frequency of monitoring observations to provide an early warning is dependent on the difference between the current status of the indicator, the early-warning status level, and the inherent spatial and temporal variability of the indicator. Realistically, given uncertainty in early-warning levels and inherent variability of indicators, monitoring resources are likely insufficient to statistically detect a declining trend within a time period sufficient for decision making (Field et al. 2004). Bennetts et al. (2007) have proposed the use of management-assessment points along a continuum of indicator values to safeguard against uncertainties in estimates of thresholds, in indicator variability, and in the efficacy of a monitoring or sampling design. Ecosystem progression, where monitored attributes reach an assessment point does not necessarily warrant immediate remediation action, but instead motivates close scrutiny. Assessment points ideally are based on management goals and concerns, including understanding risks (Nichols et al. Chap. 2). However, a fundamental component for establishing assessment points is a credible estimate of resource and environmental conditions indicative of impending state changes.

A General Approach to Applying Threshold Theory to Management

We developed a general approach for identifying properties of thresholds to inform estimates of management-assessment points in a long-term monitoring context. Our approach relies on using conceptual models of threshold dynamics, and various sources of information to verify the conceptual model, and to make informed estimates of state changes and associated indicator values:

1. *Identification of target ecosystems:* We adopted the US Department of Agriculture Natural Resource Conservation Service's (USDA-NRCS) ecological site concept as a spatial framework for ecosystem classification and model development. Ecological sites are land units differentiated by (a) physical attributes including inherent soil properties (texture, depth, and horizonation), geomorphic setting, and climate; (b) the potential (rather than current) vegetation associated

with these physical attributes within a specific ecoregion, and (c) characteristic dynamics in response to climate, management, and other driving factors (Herrick et al. 2005; Bestelmeyer et al. 2009).

2. *Conceptual models of system dynamics:* We developed STMs to organize current knowledge or hypotheses about the dynamics and community phases of specific ecological sites, the key alternative states representative of degradation pathways, and the transitions that are possible among these states. Possible triggers of transitions among alternative states, and pattern and process indicators of specific degradation pathways are identified or hypothesized based on published literature, unpublished expert knowledge of an ecological site, or general ecological principles. Identifying triggers is most useful since observations of their occurrence could initiate preventative management actions. This process- and theory-based focus in the construction of the STM, contrasts with pattern-based efforts, which seek to define states based upon classification of multivariate community structure data (e.g., Allen-Diaz and Bartolome 1998). These data-driven approaches offer the credibility of being based upon real data, but assume that a dataset is likely to capture all of the important states that are possible within a given ecological site, and that the identified states are fundamentally and functionally distinct (Bestelmeyer et al. 2003). Rather, we advocate using available data to test specific elements of process-based conceptual STMs, as a means of calibrating and validating the model.
3. *Model calibration:* Model building is an iterative process, and it is important to include a calibration step. Calibration includes testing the concepts presented in the model using available datasets, or subjecting them to the scrutiny of an expert panel. This enables an opportunity to revise the model, identify new transitions and associated triggers, processes and indicators, and allows an estimation of our confidence that the revised model is reasonable.
4. *Identification of key transitions and estimation of tipping points:* The calibrated model is used to identify the most likely transitions that might be detected by a monitoring program, emphasizing those known to be of concern to management, such as the persistent conversion of perennial grasslands to ecosystems dominated by invasive annuals or woody plants. The values of key indicators at the point of a state change—when one state abruptly transitions to another—are estimated. We refer to these as tipping points; they are roughly equivalent to restoration thresholds (*sensu* Bestelmeyer 2006). Because abrupt transitions in progress are seldom observed, statistical methods are used to model the tipping points in indicator values using sample representative of discrete states. In data-sparse situations, these estimates are derived from expert knowledge rather than statistical modeling. The assessment points are another set of indicator values which trigger management action prior to observing a tipping point, so that the undesired transition can be avoided. These values occur chronologically before tipping points and allow managers sufficient response time. They are based upon the range of natural variability in the reference or less-degraded state when data are available, or upon opinions from an expert panel when data are lacking.

Case Studies

We now present two case studies that illustrate different methods for identifying assessment points based on contrasting scenarios of data availability. The case studies specify two ecological sites that occur in NPS units on the Colorado Plateau, where the general monitoring goal is to provide early warning of system decline in sufficient time for management actions to avert impending undesirable changes.

A. Data-Rich Case Study: Semidesert Sandy Loam Ecological Site, Canyonlands National Park

Ecological Site Characteristics The semidesert sandy loam (SDSL) ecological site is widely distributed throughout the Colorado Plateau region of North America and is significant for its past and current use for livestock grazing (USDA-NRCS major land resource area 35, ecological site 035XY215UT). This ecological site occurs on flat to gently sloping landforms at 1,310–2,010 m elevation and receives 20–30 cm mean annual precipitation. Soils are formed in moderately deep to very deep (from 50 to greater than 150 cm) aeolian and alluvial deposits from sandstone and are moderately alkaline with sandy loam or loamy sand texture. In relatively undisturbed settings, the vascular plant community typically has a grassland aspect and is characterized by a mixture of perennial C_3 (*Hesperostipa comata* and *Achnatherum hymenoides*) and C_4 (*Sporobolus* spp.) bunchgrasses, C_4 rhizomatous grasses (*Pleuraphis jamesii* and *Bouteloua gracilis*), shrubs, and annual herbaceous species. In contrast with many dryland ecosystems, most common shrubs (e.g., *Krascheninnikovia lanata* and *Atriplex canescens*) are palatable to livestock and shrub-dominated communities can occur with long-term absence of livestock grazing. Plant nomenclature here and throughout follows USDA-NRCS 2010. Biological soil crust (Belnap 2003) is another biotic functional type that is a characteristic component of relatively undisturbed SDSL sites (Kleiner and Harper 1972; Bowker and Belnap 2008). Biological soil crusts have yet to be widely incorporated in conceptualizations of dryland ecosystem dynamics despite evidence of their functional significance for soil stabilization (Belnap 1995; Warren 2003), nutrient cycling (Evans and Lange 2003), hydrologic processes (Warren 2003), and mediation of plant establishment (Belnap et al. 2003; Escudero et al. 2007). Biological soil crusts are also notable for their lack of resistance to surface disturbances which can result in long-term reductions in spatial continuity, biological diversity, physical structure, and functionality (Belnap and Eldridge 2003; Miller 2008).

Management Goals and Land-Use History General NPS goals for management of natural resources are (1) to preserve and restore the natural abundance, diversity, and dynamics of native plant and animal populations and the communities and ecosystems in which they occur, and (2) to minimize human impacts on native plant and animal populations, communities, ecosystems, and the processes that sustain them (USDI-NPS 2006). Canyonlands National Park preserves regionally significant examples of SDSL ecosystems that remain relatively undisturbed by human activities exclusive of anthropogenic atmospheric changes. Within Canyonlands National Park, however, there also are extensive examples of SDSL ecosystems with persistently degraded composition, structure, and function attributable to impacts of past livestock grazing (e.g., Neff et al. 2005; Belnap et al. 2009). Domestic livestock were introduced to this area in the late 1880s and portions of Canyonlands were grazed by livestock until 1974. Livestock grazing remains an important economic activity on adjacent lands outside Canyonlands. Unlike many semiarid grasslands, neither fire nor frequent grazing by herds of large mammals are characteristic natural disturbances associated with the SDSL site. Thus, grazing and associated surface disturbances by livestock represent novel disturbances in this system.

Data Availability Three general types of data characterize structural and functional attributes of the SDSL ecological site for Canyonlands National Park and surrounding areas: (1) poorly replicated in space and time (Kleiner and Harper 1972; Neff et al. 2005), (2) well replicated in time, poorly replicated in space (Belnap et al. 2009; S.M. Munson unpublished data), and (3) well replicated in space, poorly replicated in time (Miller et al. 2011). Of these options, the first two provide many insights into ecosystem dynamics but only the third type provides the necessary replication for the statistical estimation of tipping or assessment points, or are broad enough to characterize the variability within states. The third type of data is derived from a broad-scale ecosystem inventory project purposefully designed to characterize ranges of variability in key compositional and structural attributes of dryland ecosystems in Canyonlands National Park and on adjacent lands currently used for livestock grazing (Miller et al. 2011). These inventory data were collected over a 3-year time period and thus do not quantify temporal transitions among states. However, through a combination of targeted sampling and extensive spatial replication (substituting space for time) with random sampling, this data set documents current ranges of variability for the SDSL and provides a relatively rich basis for estimating tipping points and associated assessment points. The data set quantified variability among 72 SDSL plots on a single soil type (Begay series) on the basis of live cover of biological crusts and vascular plants, ground cover, and indicators of erosion resistance including soil aggregate stability, spacing between perennial plant canopies, and spacing between perennial-plant bases (Miller et al. 2011; sampling methods followed Herrick et al. 2005). Sampling was conducted both within and outside Canyonlands National Park to ensure that the data set spanned a wide range of ecosystem conditions.

Methods: Building a State-and-Transition Model and Estimating Tipping Points with Rich Inventory Data Field observations, published literature (Kleiner and Harper 1972; Neff et al. 2005; and Belnap et al. 2009) and an existing USDA-NRCS ecological site description (USDA-NRCS ecological site 035XY215UT) provided the basis for developing an STM articulating hypotheses about system dynamics, degradation pathways among alternative states, and associated ecosystem patterns, processes, and feedback.

The conceptual model identifies four ecosystem states based on persistent differences in the relative abundance of biotic functional types (Fig. 7.1; Tables 7.1 and 7.2). Two states are dominated by biological crusts and are distinguished from one another by the absence (S1) or presence (S2) of functionally significant invasive exotic annuals (e.g., *Bromus tectorum* or *Salsola* sp.). An invaded state (S3) is characterized by the replacement of biological crust by bare ground and a vascular plant community dominated by perennial grasses (S3P1) or palatable shrubs (S3P2) with significant levels of invasive annuals. The fourth state (S4) is characterized by persistent dominance by invasive annual grasses or forbs. The first state represents the desired condition relative to NPS management goals, whereas states two through four represent increasing degrees of degradation to be avoided or mitigated.

We used a logical quantitative process to analyze the inventory data set to examine evidence for our STM. It consists of construct validation of the STM, and determination of quantitative classification rules of state membership. To validate the existence of the states proposed in our a priori STM, fuzzy cluster analysis (Equihua 1990) was applied to four state properties including biological crust cover, bare ground cover, combined cover of perennial grasses and palatable shrubs, and relative cover of invasive exotic annuals based upon a Bray–Curtis distance matrix. Fuzzy clustering methods offer more flexibility than hierarchical clustering when attempting to group elements which may overlap or have vague boundaries, such as states. Following cluster identification, classification tree modeling (De'ath and Fabricius 2000) was used to derive quantitative decision rules for differentiating clusters (Fig. 7.3). While these methods may or may not arrive at the same clustering of data, their utility is somewhat different. Starting with the root node (composed of the entire dataset), classification trees iteratively and dichotomously partition the data set into increasingly homogenous groups, producing a dendritic pattern of terminal nodes. Each partition is based upon the values of a single predictor variable. This property of classification trees makes them useful for isolating the single variable(s) most informative in determining node/cluster membership, and provide a decision rule based on that predictor (e.g., $\geq 28\,\%$ relative exotic cover = State 3). These values represent classification thresholds (*sensu* Bestelmeyer 2006) for clusters or nodes rather than actual functional or degradation thresholds for the SDSL ecological site; however, they provide a reasonable first approximation of tipping points in empirical measurements of key functional indicators.

The cluster analysis distinguished three clusters analogous to states S2–S4 in the conceptual model (Miller et al. 2011), and provided no evidence for states not included in the model. The classification analysis splits cluster S4 from clusters S2 and S3 at 28.3 % relative cover of invasive exotic annuals (Fig. 7.2a). Clusters S2 and S3 are split from one another at 30.3 % bare ground (Fig. 7.2a).

Implications for Monitoring This case study applies a conceptual model of ecosystem dynamics, a relatively rich set of inventory data, and multivariate data analysis techniques to derive monitoring-assessment points. Despite the fact that some pristine sites were included in the dataset, examples of S1 (partially defined by a lack of exotics) were not located. Thus, the management goals ought to detect and avoid the

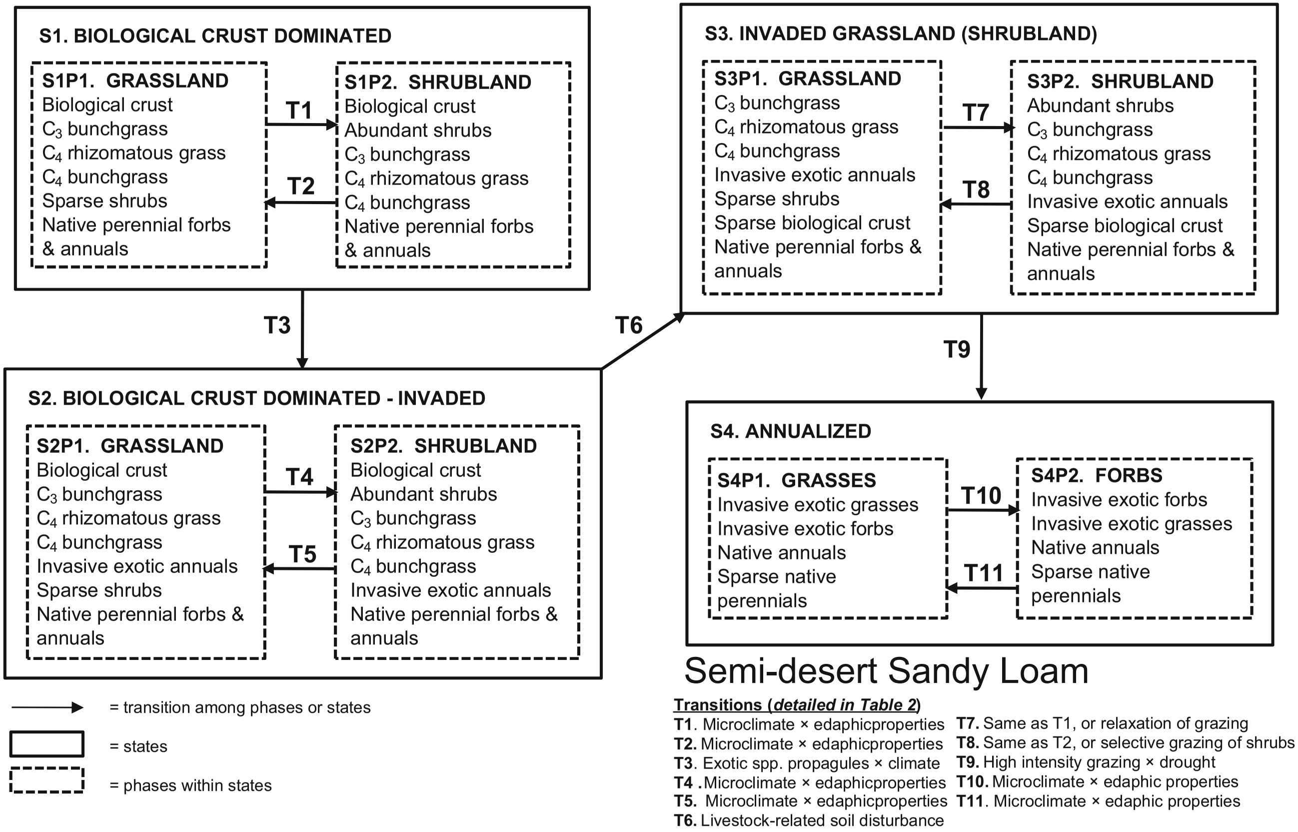

Fig. 7.1 State and transition diagram for semidesert sandy loam

Table 7.1 Catalog of states and phases in semidesert sandy loam

State	Phase	Structural properties	Functional properties	Feedback
S1. Biological crust dominated	P1. Grassland	Biological crust dominant or codominant relative to vascular plants; perennial grasses abundant relative to shrubs, with variable grass composition due to climate fluctuations, soil variability, and site history. High degree of soil-surface roughness	High biological crust cover maintains high capacity for resource capture and retention (including nutrients, water, litter, and seeds) even with fluctuations in plant cover[a,b]	High resource retention promotes plant community resilience to climatic fluctuations and natural disturbance
	P2. Shrubland	Similar to S1P1 but with palatable shrubs abundant relative to perennial grasses	Same as S1P1	Same as S1P1
S2. Biological crust dominated—invaded	P1. Grassland	Similar to S1P1 but invasive exotic annuals present. Cover of invasive annuals fluctuates with climate	Similar to S1P1, but presence of invasive annuals can cause greater climate-driven fluctuations in cover and production relative to S1P1	Same as S1P1
	P2. Shrubland	Similar to S1P2 but invasive exotic annuals present. Cover of invasive annuals fluctuates with climate	Similar to S1P2, but presence of invasive annuals can cause greater climate-driven fluctuations in cover and production relative to S1P2	Same as S1P2
S3. Invaded grassland (or shrubland)	P1. Grassland	Biological crust replaced by bare ground; otherwise similar to S2P1. Major decline in soil-surface roughness relative to S1P1 and S2P1	Loss of stability and roughness associated with biological crust result in major decline in site capacity for resource capture and retention; accelerated losses of soil, nutrients, water, litter, and seeds occur[a,b]	Accelerated losses of soil resources and seeds contribute to declines in plant community resilience to climatic fluctuations and to declines in vegetative cover and production, which result in further declines in site resistance to erosion and resource loss

Table 7.1 (continued)

State	Phase	Structural properties	Functional properties	Feedback
	P2. Shrubland	Biological crust replaced by bare ground; otherwise similar to S2P2. Major decline in soil-surface roughness relative to S1P2 and S2P2	Similar to S3P1, although rates of resource loss may be greater in shrub-land due to relative lack of perennial grass cover	Same as S3P1
S4. Annualized	P1. Grasses	Dominated by invasive exotic annual grasses (e.g., *Bromus*). Native annuals may be present, but perennials sparse	Dominance by annuals results in high fluctuations in cover due to climate, with corresponding high (and potentially extreme) fluctuations in resource loss/erosion[b]	Same as S3P1, but greater. Potential spiraling declines in resource availability and site productivity[a]
	P2. Forbs	Dominated by invasive exotic annual forbs (e.g., *Salsola*). Native annuals may be present, but perennials sparse	Same as S4P1	Same as S4P1

[a] Neff et al. 2005
[b] Belnap et al. 2009

Table 7.2 Key to transitions in Fig. 7.1 (semidesert sandy loam)

Transition	Trigger(s)	Associated process(es)	Relevant indicator(s)
T1	Climate variability, perhaps interacting with landscape position and inherent soil properties (shrubland phase may be more common on older geomorphic surfaces with greater inputs of late-Pleistocene loess, greater silt content, and greater pro- file development relative to grassland phase.	Plant population processes (reproduction, recruitment, mortality)	Absolute and relative cover of perennial grasses and palatable shrubs (or shrub:grass ratio)
T2	Similar to T1, but favoring opposite relative dominance of plant functional types	Same as T1	Same as T1
T3	Establishment of invasive exotic annuals, facilitated by favorable climatic conditions	Seed dispersal and plant population processes	Density, frequency, and/or cover of invasive exotic annuals
T4	Same as T1	Same as T1	Same as T1
T5	Same as T2	Same as T1	Same as T1
T6	Repeated soil disturbance (trampling), typically associated with livestock grazing	Destruction of biological crusts due to trampling; increased connectivity of bare-ground patches; decreased soil-surface roughness and capacity for capturing/retaining litter, seeds, aeolian dust inputs, and runoff; accelerated erosion	Absolute cover of biological crust; cover of biological crust relative to bare ground and vascular plants; soil-surface roughness; percent bare ground; size and connectivity of bare ground patches; soil aggregate stability
T7	Similar to T1, but also may be facilitated by a sustained reduction in grazing pressure on palatable shrubs where previous herbivory by livestock has suppressed shrubs relative to perennial grasses.	Plant population processes; shrub regrowth following reduction in grazing pressure	Same as T1
T8	Similar to T2, but facilitated by heavy grazing pressure and selective herbivory on palatable shrubs	Selective herbivory and competitive suppression of palatable shrubs relative to perennial grasses; plant population processes	Same as T1

Table 7.2 (continued)

Transition	Trigger(s)	Associated process(es)	Relevant indicator(s)
T9	Sustained high-intensity grazing and associated soil-surface disturbance (trampling), perhaps in combination with drought	Selective herbivory and reduction of perennial grasses and palatable shrubs through effects on physiological vigor, resistance/resilience to drought, competitive relations, seed production, and replenishment of the soil seed bank; facilitation of invasive exotic plants through soil-surface disturbance and reduced competitive vigor of grazed perennials	Absolute cover of perennial grasses and palatable shrubs; absolute cover of invasive exotic annuals; relative cover of perennials and invasive annuals; soil aggregate stability
T10	Climate variability that favors exotic annual forbs relative to exotic annual grasses; relative dominance of exotic annual forbs and exotic annual grasses also may vary along elevation and/or topo-edaphic gradients through effects on soil moisture	Plant population processes (reproduction, recruitment, mortality)	Absolute and relative cover of exotic annual grasses and forbs
T11	Similar to T10, but favoring opposite relative dominance of plant functional types	Same as T10	Same as T10

initiation of transitions from S2 to S3. Likewise, for sites already in S3, management should strive to detect and prevent transition to S4. Current monitoring conducted by NPS is well designed to detect changes in key indicators of these transition sequences for the SDSL ecological site, including the relative cover of invasive exotic plants and percent bare ground. Because we are able to provide rough estimates of tipping points based on these data, the necessary prerequisites for establishment of assessment points are established. We reason that an assessment point for a given transition must lie between the estimated tipping point and the mean value of the relevant indicator in the state at risk of transition. Its actual position is determined subjectively based upon management goals and adaptively refined based upon success as a decision support tool. Some reasonable management-assessment points, ordered from most conservative to most liberal, include: the at-risk state node mean $\pm$ SE, the upper or lower bound of 95 % confidence interval of the at-risk node mean, the midpoint between the at-risk mean and the tipping point, the upper or lower bound of 95 % confidence interval of the tipping point, and the tipping point $\pm$ SE (Fig. 7.2b).

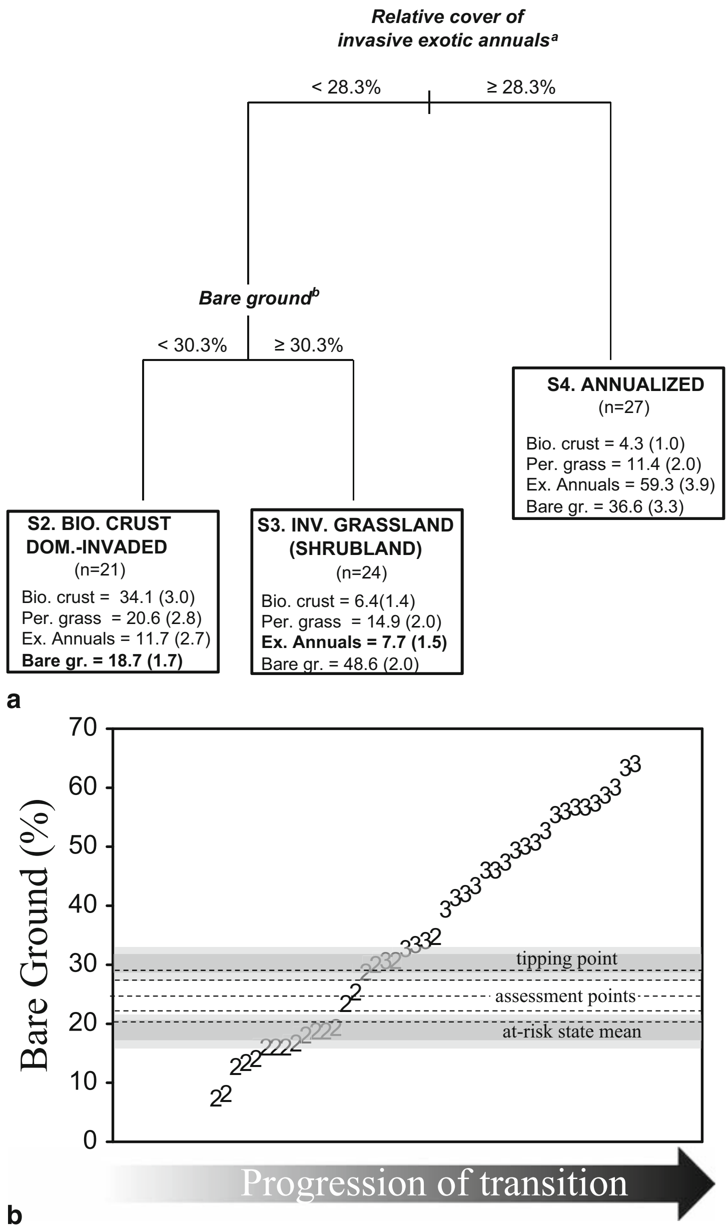

Fig. 7.2 Tipping and assessment points in the semidesert sandy loam case study: **a** Classification tree diagram depicting classification thresholds separating three states of the semidesert sandy loam ecological sites. The figure, from *top* to *bottom*, classifies samples into groups based upon values of indicators using a sequential dichotomous splitting procedure. The indicator used to make a split is in *bold italics*. Its critical values appear below it; these values are initial approximations of tipping points. End nodes are represented as *boxes* which correspond well to hypothesized states. Indicator

Current sampling is not designed to characterize or detect changes in the spatial configuration or connectivity of invasive exotic plants or bare ground. Spatial connectivity (or the length of connected pathways) in dryland ecosystems is increasingly recognized as an important structural indicator of processes such as accelerated soil erosion, overland flow, and wildfire (Okin et al. 2009). Current NPS monitoring of the SDSL ecological site on the Colorado Plateau includes measurements of gaps between perennial plant canopies and bases as indicators of resistance to erosion by wind and water (Herrick et al. 2005; Okin 2008). But no data are collected to characterize the connectivity of bare ground patches (or biological crust patches, alternatively) in the spaces between perennial plant canopies or bases. In circumstances when an assessment point is prompted by increasing levels of bare ground, measures of surface patch (intact biological crust and/or bare ground) connectivity may provide additional insights regarding degradation risks related to erosional processes.

B. Data-Sparse Case Study: Limy Uplands Ecological Site, Wupatki National Monument

Ecological Site Background Limy uplands are an ecological site represented in Wupatki National Monument and surrounding areas, situated atop fairly level basalt flows, receiving 15.2–25.4 cm of rainfall per year (USDA-SCS 1983). The soil is weathered from the underlying basalt, and from later cinder deposits due to regional volcanism. The surface is gravelly due to high-surface cinder coverage. Grassland vegetation is most common, and is dominated by C4 rhizomatous or stoloniferous grasses including *Pleuraphis jamesii* and *Bouteloua eriopoda*; C3 grasses may have been somewhat diminished due to past grazing. Savannah vegetation is less common and is characterized by an overstory of *Juniperus monosperma* of varying density and an understory of perennial grasses (Jameson 1962; Ironside 2006; DeCoster and Swan 2009).

Management Goals The primary management goals of the National Monument are to protect and preserve over 2,000 catalogued archeological sites, including structures, and agricultural fields of the ancient ancestral Hopi cultures, and to provide interpretive and educational experiences for park visitors (USDI-NPS 2002). In addition to these primary goals, NPS management goals for natural resources are the same as those summarized earlier for the semidesert sandy loam ecological site. Cattle grazing was permitted in portions of the Monument until 1989 when livestock were removed and a boundary fence was constructed (USDI-NPS 2002). The

means and standard errors are presented along with each node. *a* approximate tipping point corresponds to T9 in Table 7.2 and Fig. 7.1. *b* approximate tipping point corresponds to T6 in Table 7.2 and Fig. 7.1. **b** Percentage bare ground in rank order as a basis for establishment of assessment points. Point symbols represent state membership. Five alternative definitions of assessment points are derived from tipping points and at-risk state means estimated using a classification tree

Monument highlights the presence of a rare, large, ungrazed grassland as one of its significant resources, and NPS staff are concerned that increasing tree densities in Monument grasslands are attributable to a decrease in fire frequency since the late nineteenth century caused by diminished fine fuels due to grazing (Cinnamon 1988; USDI-NPS 2002; Ironside 2006). Currently, the wildfire management plan calls for suppression of fires, but retains the option of prescribed fire (USDI-NPS 2005).

Data Availability Relevant vegetation data for this ecological site either are well replicated and incomplete, or modestly replicated and reasonably complete. In aggregate, these data may not represent a sufficient range of the possible states, nor the ideal time series data capturing a transition in action to validate an STM, and may lack measurements of some potentially useful indicators. There is no single complete dataset, for validation of an STM or estimation of tipping and assessment points. Hassler (2006) likely conducted some sampling of *Juniperus* density, growth rate, and fire mortality on limy uplands. In a remote sensing-based vegetation mapping project, Hansen et al. (2004) sampled numerous accuracy assessment releves in limy uplands that qualitatively identify community type. Miller et al. (2007) developed and tested monitoring techniques at seven plots. DeCoster and Swan (2009) summarize the first years of the I&M program and contains the most purposefully collected monitoring dataset for limy uplands, but is limited to ten sites. The randomly selected study design may fortuitously capture recovery from fire gradients (1 plot in 1995 "North fire", 3–4 plots in the 2002 "Antelope fire"; USDI-NPS 2005). The data include detailed information on vegetation structure and ground cover, including some metrics of juniper density, but lacks direct indices of connectivity of fine fuels.

Methods: Building a State-and-Transition Model and Estimating Assessment Points with Sparse Data Due to the incomplete nature of the available data, we pursued an alternative strategy for the validation of the states and dynamics delineated in STM. Our approach has much in common with the Delphi technique of engaging expert opinion panels, in that, it is a multiphase, iterative approach, employs a "straw-document" as a starting point, and engages participants individually so that outputs are not disproportionately affected by dominant personalities (Linstone and Turoff 1975; Oliver 2002). This approach has proven to be useful when "the problem does not lend itself to precise analytical techniques but can benefit from subjective judgements on a collective basis (Linstone and Turoff 1975)." We constructed email-based questionnaires in two stages: (1) model calibration, (2) estimation of tipping and assessment points in indicators which enable detection of proximity to threshold crossings. Based on literature findings and past experience, we drafted an STM including a catalog of states, phases, and transitions. We identified a list of potential expert consultants from the authors of relevant literature, and from professional interactions. We initially contacted selected experts by email to gauge interest. Of eight people contacted, five were willing to participate. The format of the model calibration survey included: (1) a paragraph-length overview of STM concepts, (2) a brief description of the target ecological site, (3) a draft STM including a diagram and verbal catalog, and (4) a questionnaire. The questionnaire consisted of four required questions and six optional ones. The required questions asked respondents to identify any states, phases, or transitions which should be removed from or added to the model. For additions, respondents were prompted to identify: structural and functional properties and stabilizing negative feedback of states and phases, and triggers (including their characteristic scale) and appropriate indicators of transitions. Our questionnaires specifically employed estimates of confidence in responses, an important measure of uncertainty. In the Phase 1 questionnaire, respondents were asked to estimate their confidence in a revised model, which took into account their proposed changes (a subjective scale taking any value from 0 to 100 %, where 0 % = "It's anyone's guess, this model is no better than any other model," 50 % = "Because this model is reasonable, I would tend to believe it until evidence to the contrary is presented," 100 % = "The model is so well supported by evidence and accumulated knowledge, that I am certain it is correct."). The same information was requested for each individual model component (states, phases, and transitions). These confidence estimates are hereafter known as "C-own." As a complementary question, respondents were also asked to estimate their confidence in the model generated by a theoretical "best qualified" person, to help gauge their confidence in a survey-based procedure for developing STM (hereafter known as "C-best"). We received four surveys with an average response time of 9 days (we had requested return within a week). We revised the model, according to all respondents' comments. We also calculated an aggregate confidence value. First, the C-best values were used to correct optimistic or pessimistic tendencies in respondents' estimation of C-own. For example, if a respondent's C-best value was 20 % less than the mean C-best value, their C-own value was adjusted up by 20 % to account for their greater than average pessimism. The adjusted C-own values were averaged across all respondents, and calculated for the entire model and for each model component.

The second phase of the survey was more focused on thresholds associated with a key transition (T6 from reference grasslands to savannized ecosystems, see later). This survey consisted of the following parts: (1) a revised STM with aggregate confidence values, (2) a background section regarding resiliency concepts, tipping and assessment points, and (3) a questionnaire. In the questionnaire portion, respondents were presented with a set of indicators and their characteristic units, and were asked to estimate tipping and assessment points for each. As in the previous survey, we required C-own and C-best values for all indicators overall, and respondents were invited to provide them for each individual indicator. We emailed the Phase 2 surveys to the four respondents who had previously returned Phase 1, in addition to one new respondent and several previous candidates who had not been able to respond. We received six of them back with an average response time of 20 days. To calculate aggregate estimates of assessment and tipping points, we adjusted each respondent's C-own values using their C-best values, using the same procedure described earlier. The adjusted C-own values were then used to compute a weight for a given estimate of a given respondent by dividing the adjusted C-own of the respondent, divided by the sum of all respondents' adjusted C-own values, yielding a proportion. This proportion was used in a weighted averaging procedure to calculate the group's estimates.

Final Resilience-Based State-and-Transition Model We acknowledge a prehistory of volcanism and occupation by agricultural societies, and subsequent depopulation (Sullivan and Downum 1991), but omit detail on these states for brevity. We emphasize states, phases and transitions within the current management sphere (Fig. 7.3; Tables 7.3 and 7.4).

The survey-based approach proved to be quite useful, but perhaps not fully satisfactory. On one hand, they proved to be an excellent tool for calibration of STMs, as new states, phases, and transitions were identified, and differing levels of confidence emerged in different portions of the model, identifying the greatest research needs (e.g., potential for transition to woodland, and recovery of grass dominance after shrub dominance; Fig. 7.3). The overall aggregate confidence in the model was quite high on a subjective scale (71 %), indicating that despite the lack of data, survey respondents tended to believe that this model was the correct model of ecosystem dynamics. We were able to provide quantitative approximations of tipping and assessment points based upon subjective rather than empirical data for only three of ten indicators (based on estimates given by a minimum of three respondents; Table 7.5). This was because respondents were reticent to offer estimates about subjects for which they did not feel knowledgeable (less than about 20 % confidence), thus for indicators related to livestock or native grazer activity and connectivity of fuels we obtained little information. However, each respondent did suggest at least one additional indicator resulting in a total of seven additional indicators that could be folded into a monitoring program. Data gaps could probably be ameliorated with a larger sample size of surveys when possible; however, our approach has the inherent limitation that there are a small pool of respondents with knowledge of the target site, and even fewer available to respond to surveys.

Implications for Monitoring Expert opinion surveys resulted in a highly useful model of ecosystem dynamics and seven suggestions of indicators which should be investigated further for their potential to indicate change, several of which could be derived from the data currently being collected. Most respondents tended to believe that the transition to savannahs is fire regulated. As a result, we were able to establish rough first approximations of tipping points in some related indicators to aid in the establishment of assessment points (Table 7.5). These estimates should be confirmed based upon data when possible, but illustrate that even when data are lacking, an operational tipping point can be established. Compared to the data-rich case, there is less available information to establish assessment points; for example we do not know the distributions of indicator values within the at-risk state. However, the weighted average of survey respondents' assessment points provides a reasonable starting point.

Survey products suggested several ways to learn about this ecosystem. For example, the two leading hypotheses regarding savannization, that the process is fire-limited, and that the process is favored by wet climate periods, could be tested using monitoring data. Currently, the NPS I&M sampling strategy within Wupatki's limy uplands is well designed for detecting changes in vegetation structure such as increasing relative abundance of woody plants. However, the design could be

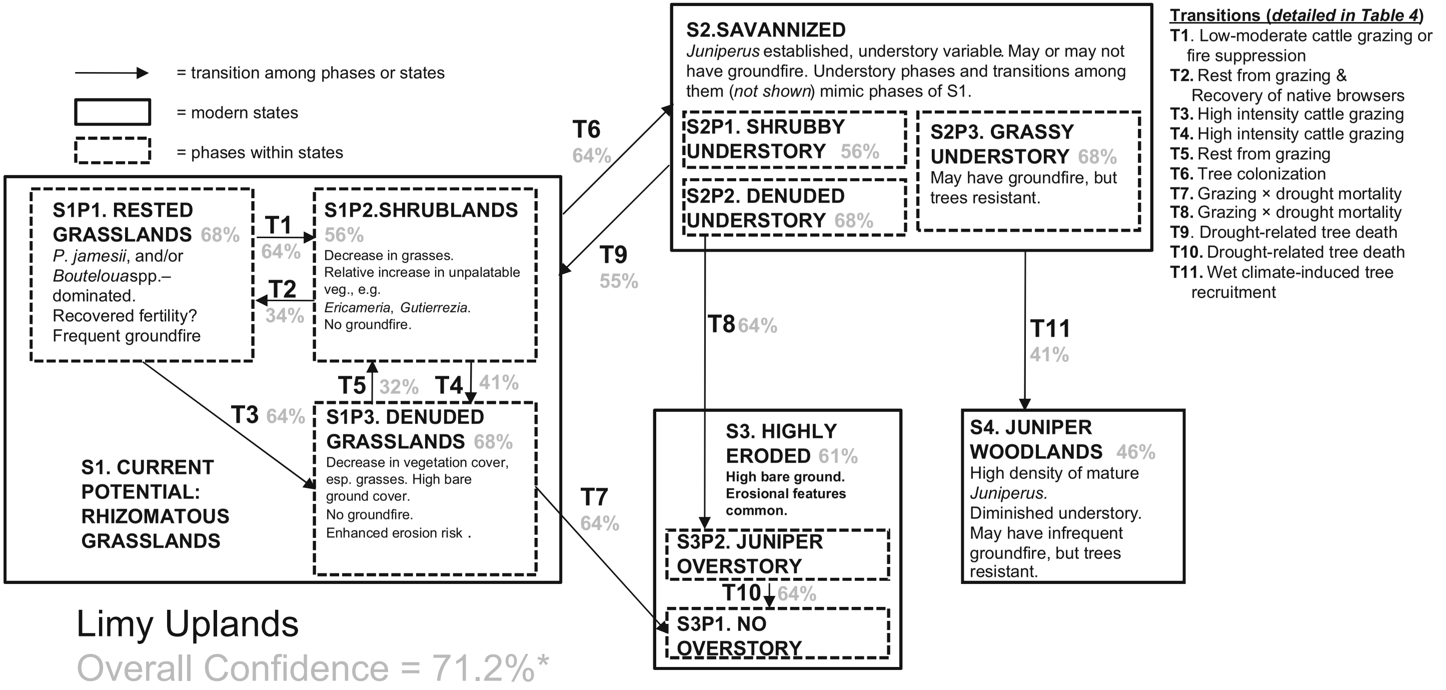

Fig. 7.3 State-and-transition diagram for limy uplands. Overall confidence values also apply to any model component (state, phase, and transition) for which no confidence estimate is provided (*gray text*)

Table 7.3 Catalog of states and phases in limy uplands

Phase	Structural properties	Functional properties	Feedback
P1. Rested grassland	Grassland: *P. jamesii*, and/or *Bouteloua* Bouteloua spp., *H. comata* well represented[a,b]	Presumed recovered productivity equal or greater than Pre1; possibly recovered soil fertility; otherwise similar tp Pre2	Frequent ground fires (15–20 year return),[a,c] resprout of rhizomatous grasses, and browsing by *Antilocapra americana* constrain woody plant abundance
P2. Shrubland	Relative increase in unpalatable shrubs (*Ericameria, Gutierrezia, Artemisia*) or cattle-grazing tolerant grasses (e.g., *Bouteloua gracilis*)[d,e]	Frequent fire cycle of SIP1 interrupted due to loss of connectivity or amount of fine fuels[c,e]; at-risk of state transition; otherwise similar to S1P1	Resprout of rhizomatous grasses, after cattle grazing confers resilience[b,f] improved forage for *A. americana* promotes transition back to grass dominance
P3. Denuded grassland	Relative increase in unpalatable shrubs (*Ericameria, Gutierrezia*), or cattle grazing tolerant grasses (e.g., *Bouteloua gracilis*)[d]; increased bare ground (may be extreme)[e] *Juniperus* may begin colonizing[e]	Frequent fire cycle of SIP1 interrupted due to extreme loss of connectivity and amount of fine fuels[c,e]; at-risk of state transition; otherwise similar to S1P1	Resprout of rhizomatous grasses, rapid colonization of shrubs, after cattle grazing confers resilience[b,f]
P1. Shrubby understory	Understory similar to S1P2; *Juniperus* established in site[a,b,c]	Frequent fire cycle of SIP1 interrupted due to loss of connectivity and amount of fine fuels[c,e]	Same as S1P2 in understory
P1. Denuded understory	Understory similar to S1P3; *Juniperus* established in site[a,c]	Frequent fire cycle of SIP1 interrupted due to extreme loss of connectivity and amount of fine fuels[c,j]	Same as S1P3 in understory
P2. Grassy understory	Understory similar to S1P1, *Juniperus* established in site[a,c]	Recovered connectivity and amount of fine fuel in understory; Except for overstory functionally similar to S1P1	Frequent ground fires (15–20 year return)[a,c] and browsing by *Antilocapra americana* prevent new woody plant colonization, but does not cull extant *Juniperus*[c]
P1. Highly eroded—no overstory	Low vegetation and high bare ground cover	Productivity too low to temper erosivity, declining soil fertility, erosional features apparent	Lack of vegetation allows erosion, erosion prevents recolonization

Table 7.3 (continued)

Phase	Structural properties	Functional properties	Feedback
P2. Highly eroded—Juniper overstory	Same as S3P2, except *Juniperus* established in site	Same as S3P2 in understory	Lack of vegetation allows erosion, erosion prevents recolonization
n.a.	Increased frequency, cover of *Juniperus*,[c] decreased understory due to shading and litter deposition	May have less frequent ground fire, but mature trees not culled	*Juniperus* reduces fire susceptibility, which favors *Juniperus*

[a] Cinnamon 1988, [b] USDA-SCS 1971, [c] Hassler 2006, [d] Jameson 1962, [e] Sullivan and Downum 1991, [f] Stone and Downum 1999

improved in terms of its ability to detect changes in fire susceptibility, since fire occurrence is a resilience mechanism. We recommend refinement and implementation of indicators focused directly on fine fuels connectivity (e.g., combustible patch length, interspace length (devoid of combustible materials)). While the total amount of fuels is important, fuel arrangement in space may be equally informative. A site-specific fire susceptibility model, using these same indicators, would be a highly useful tool to predict the effects of monitorable variables upon site resiliency, which is based upon the fire return cycle. Fire susceptibility may function as a more anticipatory indicator than vegetation structure alone. Such a model could provide a simulation-based confirmation of transition dynamics, and assessment/tipping point estimates, and some degree of forecasting ability, such as the most probable location of the next fire. The role of periods of above-average precipitation in the savannization phenomenon should also be investigated both retrospectively, and using simulation modeling of future climate.

This case study is an example of a situation where monitoring can be applied for scientific or learning processes (Nichols and Williams 2006). As understanding of this ecosystem advances, the monitoring program could move towards a focused tool for decision making.

Discussion

Our operational approach to evaluating threshold dynamics for upland ecological sites in dryland systems offers a variety of advantages:

1. State-and-transition models for individual ecological sites specifically articulate hypotheses regarding reference conditions and ecosystem dynamics in the context of goals for management and monitoring. Attributes of alternative states help to identify biophysical features that may be indicators of an impending transition (threshold crossing). Listing known or hypothesized mechanisms and processes underlying transitions among alternative states and phases also aids in identifying indicators to be monitored. This helps guide quantitative and qualitative estimation of tipping points,

Table 7.4 Key to transitions in Fig. 7.3

Transition	Trigger(s)	Associated process(es)	Relevant indicator(s)
T1	Introduction of persistent light to moderate cattle grazing, associated reduction of native browsers; fire suppression	Reduced amount/connectivity of fine fuels (e.g., grass) leading to interrupted fire cycle	Stocking rate, cowpie density, *A. americana* pellet density, total or basal cover (incl. litter), shrub: grass cover, bare and combustible patch size, time since fire
T2[a]	Cessation/reduction of cattle grazing fire—wild or controlled *Antilocapra americana* browsing	Recovered amount/connectivity of fine fuels (grasses) leading to restored fire cycle	Rest period length, total or basal cover (incl. litter), *A. americana* pellet density, shrub: grass cover, bare and combustible patch size, time since fire
T3[a]	High intensity cattle grazing with little rest (similar to pre-Taylor Grazing Act), associated reduction of native browsers	Strong reduction in amount/connectivity of fine fuels leading to interrupted fire cycle	Stocking rate, cowpie density, *A. americana* pellet density, total or basal cover (incl. litter), bare and combustible patch size
T4[a]	Same as T3	Same as T3	Same as T3
T5[a]	Cessation of cattle grazing	Recolonization of vegetation, including resprouting shrubs and grasses or persistent wet conditions	Rest period length, pellet density, total or basal cover (incl. litter), bare and combustible patch size
T6[b]	Tree colonization (linked to T1, T3, T4)	If seed source exists, *Juniperus* may establish due to lack of fire	Frequency/density of trees, tree height
T7[a]	Sustained high-intensity grazing possibly in concert with drought	Vegetation loss allows erosion, high erosion rates prevent recolonization	Rills, gullies, terracettes total plant cover
T8	Same as T7	Same as T7	Same as T7
T9[c,d]	Interaction of extreme drought, high temperatures, edaphic/physiographic stressors	Hydraulic failure of trees, loss of overstory	Percent of tree mortality
T10	Same as T9	Same as T9	Same as T9
T11[d]	Climate change-linked prolonged wet period	Major recruitment and establishment of *Juniperus*	Same as T6

[a] Cinnamon 1988
[b] USDA-SCS 1971
[c] Hassler 2006
[d] Jameson 1962

Table 7.5 Estimates of tipping and assessment points based upon expert opinion surveys in the limy uplands case study

Indicators	Assessment point[a]	Mean adj. confidence	Maximal adj. confidence	Number of respondents	Tipping point	Mean adj. confidence	Maximal adj. confidence	Number of respondents
Time since fire (y)	22.4	37 %	48 %	4	28.0	40 %	48 %	3
Total plant cover (%)	18.8	40 %	51 %	6	7.7	39 %	51 %	5
Basal cover incl. litter (%)	17	38 %	51 %	5	9.7	37 %	51 %	4
Interspace length (cm)	66.7	42 %	42 %	4	–	–	–	–
Average tree height (m)	0.86	42 %	42 %	3	1.62	27 %	42 %	3

– indicates that fewer than three respondents supplied an estimate, thus these estimates are omitted
[a]Value represents the mean assessment point supplied by survey respondents, and functions only as a guide for where managers would place such a subjective value or values

and establishment of assessment points for monitoring purposes. In dryland systems, resource managers use ecological sites to stratify sampling in monitoring programs due to the likelihood that dynamics will vary among ecological site types (e.g., Herrick et al. 2005, 2006; O'Dell et al. 2005; Thomas et al. 2006). Applying STMs and associated threshold-related assessments to individual ecological sites provides results specific to individual ecosystems and their unique management challenges.

2. This approach enables monitoring for focused management decision making, by narrowing the breadth of information to monitor. Theoretically, the number of possible threshold triggers affecting an ecological site and resulting pathways can be unlimited. In developing an STM, there is a natural rendering of this unlimited number to those known to occur from past observation, or perceived to be highly plausible based on logic and inductive reasoning (i.e., experience with other dryland systems or ecological site types). This more limited and practical domain is more understandable by managers, and preventative and remediation actions can be prescriptive for specific conditions and alternative states. Furthermore, explicit consideration of key-change agents and associated management actions in STMs promotes monitoring for management decision making (qv. Nichols and Williams 2006). A major barrier to monitoring for active conservation is a lack of explicit representations of hypotheses about ecosystem responses to management actions, climate, and other drivers of ecosystem dynamics. Formalizing current system knowledge in STMs is an initial and critical step for focused discussion and understanding of useful indicators for monitoring, and for designing responsible and efficient monitoring efforts to inform management actions.

3. We provide a quantitative approach to estimate tipping and assessment points using data. An ideal dataset for the estimation of assessment and tipping points would consist of a well-replicated experimental manipulation of stressors where quantitative sampling of multiple key indicators in a time series would capture the progression of a transition. Such data resources are the minority, whereas data employing space-for-time replacement tend to be much more available. Within one or a few points in time, samples are obtained that represent spatially discrete examples of different states and phases. Since the transitions are not actually documented in the data, it is assumed that the hypothesized states and transitions articulated in the STM are the correct model of ecosystem dynamics; observed degraded states are assumed to have transitioned in the past from other states due to the model-specified mechanisms. Statistical assessments relying on cluster analysis and the quantification of differences among clusters defines state membership, and indicator values, most useful for distinguishing among states, represent operational tipping points. Assessment points for the identified indicators can be specified on the basis of the natural variation in the less-degraded state. Identifying key indicators and status associated with vulnerable phases or threshold crossings enables managers and scientists to ascribe meaningful and useful assessment points to ensure detection of a changing resource, and to provide sufficient response time to prevent resource degradation or loss. This approach can be applied to the majority of cases for which there are available data; the basic requirements are hypothesized ecosystem dynamics and datasets which are able to capture multiple ecosystem states.

4. We provide a nonempirical, partially quantitative approach to modeling ecosystem dynamics and estimating tipping and assessment points in the absence of data. We developed a practical, qualitative approach to developing STMs and describing system dynamics where empirical data are sparse or lacking. This may be the dominant data-availability scenario in dryland ecological sites of the Colorado Plateau. To accommodate these situations, we developed a Delphi-like protocol to use expert opinion and experience of resource managers and scientists to develop an STM, and to begin to identify system attributes of impending thresholds and of alternative states after a threshold crossing. The Delphi method is based on the principle that group judgment is more accurate than individual judgment. Delphi methods attempt to estimate an unknown quantity (e.g., probability of an event occurring) by asking an anonymous expert panel their opinions in isolation (Linstone and Turoff 1975; Oliver 2002). Multiple iterations allow respondents to change their answer, based upon the anonymous responses of other members, until convergence is achieved on a single value or a narrower range of values. We used some of the principles of this approach, but did not seek convergence. We used the respondents' confidence in their own responses as weights in a procedure analogous to model averaging. In this way, we arrived at quantitative estimates of both assessment points and tipping points in a few indicators along a transition sequence in only one iteration. We found this method to be reasonably efficient, requiring only 2 months and two surveys; however, it was difficult to obtain sufficient information on most indicators. Further, rather than seeking consensus, confidence estimation provides an additional product measuring respondents' self-assessed level of uncertainty about an issue and identifies the most pressing needs for evidence.

Critics of similar expert-opinion methods suggest that such approaches only serve to boost confidence in respondents' ignorance. However, the dominant practice in resource conservation tends to be based on the experiential knowledge of individuals, rather than high-quality data or organized group judgment (Cook et al. 2010). We present our expert-opinion protocol as an improvement over the experiential knowledge of individuals that can be applied to identify critical indicator levels in monitoring any ecosystem. This approach can be applied more quickly and cheaply than a scientific study, giving it much utility when time or funds are limiting. Weighted averages of group assessment and tipping point estimates provide an intermediate level of quantitative data quality, higher than individual judgment and lower than quantitative field and experimental data. We do not consider a model produced using this procedure to be final, rather it is a first iteration of a useful model which should be refined as more information becomes available. Estimates of model parameters can serve to inform prior information in later Bayesian estimation using data.

Concluding Remarks Monitoring efforts by the NPS I&M networks are unlikely to attain their full potential without a clear understanding of vulnerable conditions and tipping points associated with ecological thresholds; however, the strength of these monitoring efforts is that they anticipate the development of this understanding. Scientific research and synthesis must provide the missing information. The two approaches we used in this chapter have the potential to provide a credible basis for establishing assessment points for these monitoring efforts. Estimates of assessment point values are surprisingly rare in the literature (but see Digiovinazzo et al. 2010), yet they seem crucial to the goal of applying threshold concepts to management problems. This goal is consistent with application of a preventive threshold: Attaining an assessment point of one or more indicators could trigger regulation of "changes to patterns that make systems vulnerable to deterministic or event-driven change" so that the undesired transition never occurs (Bestelmeyer 2006). In conservation and resource management, decisions must often be made regardless of the level of confidence in our knowledge of ecosystems (Soulé 1985; Cook et al. 2010). Our goal should be to develop the best set of models possible given the level of information available to support decisions. The approach presented here offers a flexible means of achieving this goal, and determining specific research areas in need of study.

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