CHAPTER

8

A Landscape Approach to Rangeland Conservation Practices

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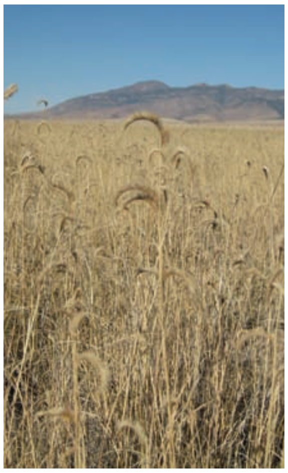
INTRODUCTION

A primary objective of the Conservation Effects Assessment Project (CEAP) has been to evaluate whether or not rangeland conservation practices (hereafter "practices") supported by the US Department of Agriculture Natural Resources Conservation Service (USDA NRCS) yield the environmental benefits that we ascribe to them. In doing so, the authors of most CEAP chapters have focused on specific conservation practices (e.g., grazing management or brush control) and sought generalities about their effectiveness based on a review of the literature. Often, a weight-of-evidence–based interpretation is drawn from the percentage of studies that support a particular assertion. The goal of this approach is to produce general recommendations about the implementation of practices, sometimes tailored to broad ecosystem types. The limitations of this approach, however, are revealed when the weight of evidence for or against the utility of a practice is equivocal. Often, practices both succeed and fail in different situations. Thus, the evidence suggests that we do not have the data to discover *under what circumstances a practice succeeds or fails* (Michener 1997). In other words, *we cannot yet account for spatial heterogeneity*, including variations in soils, climate, vegetation state, within-site patchiness, and landscape position relative to dispersal, water movement, and other processes that strongly influence the success of practices. In the face of this information shortage, we inevitably overgeneralize and fail to recognize important variations in context.

A related issue is that many practices are believed to have benefits that are manifested at spatial scales larger than the treatments themselves. A primary example is the expectation that prescribed grazing or brush management applied to uplands will have measurable effects on riparian or watershed function (Goodwin et al. 1997). It is also expected that practices will have a measurable positive impact on rangeland conditions at a landscape to regional level and that an improvement in one location is not offset by degrading processes at another location. Thus, field-scale (local) evaluations of practices that have been emphasized in this document do not enable evaluation of the *broader-scale, cumulative effects of practices* (e.g., Kondolf et al. 2008). Because of both spatial heterogeneity and differences in how multiple local treatments scale up to affect broad-scale attributes, we cannot simply assume that more is better in a linear way. We will have to measure directly attributes at broad scales and relate them to the locations and consequences of field-scale practices. Such linkages are currently rare because conceptual models of cross-scale interactions are only now being developed and resource managers are usually not certain how to apply them.

Similarly, from a sociological standpoint, there is an expectation that successful practices accelerate their adoption by other landowners in the immediate area (Kreuter et al. 2005). Among the hypothesized benefits of conservation programs, including technical assistance and cost sharing, is that local demonstration of benefits encourages the use of practices among neighboring managers. The spread of practices among managers provides another means for local practices to have effects at broader scales.

In this synthesis, we promote the development of a systematic approach by which the NRCS



Blue grama (Bouteloua gracilis) grass (foreground) and the Animas Mountains (background) in the Malpai Borderlands of southwestern New Mexico. (Photo: Brandon Bestelmeyer)

“…information provided by this “landscape perspective” could enable planners to increase successful application, use federal resources more efficiently, and assess more effectively the consequences of practices.”

and other agencies can evaluate both the local and *landscape context* of practices—that is, where they occur in a landscape and region and the varying processes and constraints associated with those locations. We further emphasize that this approach should include increased attention to *spatial pattern* as an attribute that contains valuable information, in addition to averages or sums of variables (such as plant cover) that are typically emphasized. Collectively, the information provided by this “landscape perspective” could enable planners to increase successful application, use federal resources more efficiently, and assess more effectively the consequences of practices. The empirical basis for these assertions within the rangeland conservation literature is weaker than for other chapters due to limited development of landscape perspectives in rangeland ecology and the consequent paucity of studies. Nonetheless, evidence from the broader literature in landscape ecology (e.g., Liu and Taylor 2002) and some key examples in rangelands supports our contention that it is essential for NRCS and its partners to develop 1) interpretive tools that facilitate a consideration of landscape context and spatial pattern in conservation planning and assessment and 2) database systems that link practice effects to ecological sites, state-and-transition models (STMs), and the mosaic of ecological sites and states in a landscape.

This chapter is organized into seven sections. Following this introduction, we review the current processes used by NRCS in conservation planning at different spatial scales. We then review concepts that can be used to better place practices in a landscape context and introduce a spatial hierarchy to facilitate application of landscape concepts. We then outline a model-based approach that could be used to design and test the effects of practices, taking into account landscape context and linking tests to ecological site descriptions (ESDs) and STMs. We offer some general recommendations for incorporating landscape perspectives in conservation planning, identify knowledge gaps that must be overcome to act on some recommendations, and conclude that landscape perspectives are useful and feasible. Because the language used to describe elements of the landscape perspective is not well-known or standardized, we encourage readers to refer to definitions for terms and phrases used in this chapter (Table 1).

CURRENT STATE OF LANDSCAPE PERSPECTIVES IN CONSERVATION PRACTICES

Conservation Planning at Multiple Scales

Most NRCS staff currently involved in conservation planning and implementation have been trained as “progressive” planners, which means that NRCS planners and clients incrementally implement conservation practices within a specified area. Progressive planning has the advantage of focusing resources on immediate concerns, but lacks the spatial and temporal perspectives necessary to meet landscape-scale goals and, more importantly here, to provide a consistent and transparent basis to assess conservation effects. The end result of the planning process should be a conservation plan that identifies specific actions to be taken by land managers in order to meet objectives for specific land areas. For a variety of reasons, progressive planning often does not result in a comprehensive strategy that addresses the variety of conservation needs at different scales.

Regardless of the planning approach, conservation plans rely on the implementation of individual or combinations of conservation practices to achieve objectives. “Conservation practices” are protocols for actions taken by land managers to improve or maintain the condition of rangelands (USDA NRCS 2003a). Practices are classified as 1) vegetation management practices, 2) facilitating practices, or 3) accelerating practices. To a large extent, this classification also reflects the amount of resources required to implement the practices:

1. Vegetation management practices are intended to influence the use and growth of the vegetation and are specifically evaluated in other chapters. Examples include prescribed grazing and prescribed burning.
2. Facilitating practices are intended to create infrastructure that aids in vegetation management and are only indirectly evaluated in other chapters. Examples

TABLE 1. Glossary of landscape-related terms and phrases used in this chapter. Many definitions adapted from Turner et al. (2001)

Cross-scale interactions	How processes at one spatial or temporal scale interact with processes at other scales (e.g., fine-scale plant growth interacting with flows of surface water in a landscape)
Land unit/plant community	Areas of land that are sufficiently large to be of management interest and are ecologically homogeneous with respect to management issues
Landscape	An area that is spatially heterogeneous in a property of interest, usually with respect to plant communities
Landscape/spatial context	The influence of the location within a landscape or in space, including both underlying heterogeneity and spatial interactions with neighboring locations
Landscape level/scale	A level of biological organization characterized by a mosaic of plant communities that have developed in response to common soil and geomorphic processes and that often exhibit spatial interactions (e.g., a watershed or multiple watersheds)
Multiple scale	Simultaneous consideration of patterns and processes occurring at different spatial scales
Patch	A relatively homogeneous area that differs from its surroundings; used here in reference to distinct fine-scale assemblages of plants or ground cover that make up a plant community
Patchy/Patchiness	Being made up of different patches; the degree to which an area is made up of diverse patches
Scale	Spatial dimension of a measured attribute or process, characterized by its grain (smallest resolved unit) and extent (the area across which measurements are taken)
Scale of heterogeneity/spatial pattern	The idea that heterogeneity and pattern can be defined differently at different spatial scales
Scaling up	Using measurements of properties gathered at finer scales to estimate properties at broader scales
Spatial heterogeneity	Variability or dissimilarity of properties of interest across a defined area
Spatial interaction	The flow of matter, disturbance, or information from one location to another
Spatial or landscape pattern	The arrangement of patches or land units of interest in geographical space and relationships between these units

include water developments, stock trails, and fencing.

3. Accelerating practices are intended to supplement vegetation management by promoting plant community change more rapidly than is possible through vegetation management alone, often at great expense. Examples are brush management, range planting, or channel stabilization.

Conservation planners work with individuals and groups to inventory resources, identify concerns and objectives, and develop a conservation plan. Once all of the planning inputs have been documented, individual practices are assembled into conservation systems to meet the needs of clients (e.g., a resource management system). This process involves map-based decisions about where to implement particular practices within individual field and property boundaries. Although this level of planning may involve a high degree of precision regarding the placement of practices (i.e., fencing, water development, roads), there is seldom a clearly defined link to important processes occurring at larger spatial scales. For example, a water development may be planned and implemented within an individual pasture with the objective of improving grazing distribution. However, the improved grazing distribution is seldom documented quantitatively and the assumed larger-scale effects of improved grazing distribution (water quality, habitat improvement) also lack consistent measurement. Notable exceptions to this generalization include cases in which planning considered sage grouse movements between nesting and lekking sites (Connelly et al. 2000) or where planning was designed to prevent

A semiarid grassland (foreground) and creosotebush shrub savanna (background on hill) in southcentral New Mexico. (Photo: Brandon Bestelmeyer)

the spread of waterborne bacteria (*Escherichia coli* O157:H7) from rangelands into green leafy produce fields (Tian et al. 2002; Jay et al. 2007).

One limitation to the broader use of landscape perspectives is the current focus on the relatively homogeneous "site" as the fundamental spatial unit of rangeland inventory, planning, and assessment (Brown et al. 2002; Washington-Allen 2006). The size of a site varies, but it is roughly 5–50 ha. Following current NRCS protocols, sites are classified to an "ecological site" based on the relationship of potential vegetation to differences in climate, soils, and landscape position relative to water movement or solar energy inputs. The potential vegetation and associated vegetation dynamics described for ecological sites (using STMs) have been used as a benchmark with which to gauge the effects of management on rangelands. The current ecological site system defines benchmarks on a "piece-by-piece" basis, however, and does not directly address how the composition and arrangement of multiple sites (i.e., landscape context) or variability within sites should influence decisions (USDA NRCS 2003a). Research in landscape ecology and the practical experiences of planners suggest that the landscape context of a land unit can be critically important (explanation below). Although management units commonly contain multiple ecological sites, there has been little attempt to formalize procedures for using information about the composition and arrangement of sites or variability within sites in planning or assessment. Historically, planning and implementation of practices did not consider landscape context in the placement of water facilities, fences, or other facilitating practices. Multiple-site planning most often occurred when designing grazing systems to accommodate livestock movement, unique plant growth patterns, and nutritional needs. However, there has been little effort to search for principles relating to the assessment of practice effects at multiple spatial scales. Furthermore, there is little guidance for integrating multiple properties into planning or assessment, which represents an even broader scale of heterogeneity and pattern.

There is some precedent for planning and assessment at multiple scales in the form of "area-wide conservation plans" referred to in the *National Planning Procedures Handbook* (USDA NRCS 2003b).

"Area-wide conservation plans are voluntary, comprehensive plans for a watershed or other large geographic area. Area-wide conservation plan development considers all natural resources in the planning area as well as social and economic considerations. Plan development follows the established planning process to assist local people, through a voluntary locally led effort, to assess their natural resource conditions and needs; set goals; identify programs and other resources to solve those needs; develop proposals and recommendations to do so; implement solutions; and measure their success."

This statement allows for the scale of the Conservation Management Unit to be determined by the planner and acknowledges that goals can be defined for multiple scales, but provides little guidance for planning and assessment across the scales.

The Watershed Protection and Flood Prevention Act (83-566), referred to as PL 566, authorizes NRCS to work with clients at spatial scales greater than the property level. Guidance for this program is found primarily in the *National Watershed Manual* (USDA NRCS 2009). The tools for planning, however, are primarily those used for individual property planning. Assessment tools differ slightly in that outcomes are usually expressed at the watershed scale, often using economic variables (e.g., flood prevention benefits).

The *National Biology Handbook* (USDA NRCS 2004) also considers multiple scales for programs to enhance wildlife habitat. In this document, the concepts of core reserves (nodes), corridors, and buffer zones are introduced and applied by example. In addition, qualitative metrics for assessing patch, corridor, matrix, and structural attributes are introduced. In discussions of these attributes, it is clearly stated that planning, implementation, and assessment may span several spatial scales.

In spite of these precedents, NRCS policy, guidance documents, and reports of accomplishments have focused largely on planning and outcomes at the individual management unit or ranch scale. In addition, the dominance of the Environmental Quality Incentives Program (EQIP) in agency activities often dictates that planners work on Conservation Management Units that are defined by management unit boundaries. The lack of tools to assist in planning or quantitative assessment of ranch-, watershed-, or landscape-level outcomes limits the ability of planners to address broad-scale problems.

Measurement of Practice Effects

There is currently no specific NRCS protocol or program designed to measure conservation effects at multiple spatial scales. The only dataset that covers all nonfederal rangelands (and all other nonfederal lands) is the National Resource Inventory (NRI). The NRI is a "longitudinal survey of soil, water, and related environmental resources designed to assess conditions and trends every five years on non-federal US lands" (Nusser and Goebel 1997). The sampling framework for NRI is based on sampling areas (primary sampling units or segments) of 160 acres (64.8 ha). The number of segments has varied throughout the life of the NRI, ranging from 108 000 to over 300 000 nationwide. Within each segment, three sample points are randomly located and various attributes are measured in them. This structure permits statistical inferences about changes in land use and practice application at regional, state, and national levels. The NRI was designed initially to detect changes in broad classes of land cover and use (e.g., cropland to pasture, or wheat to corn). Most NRI analyses are based on photointerpretation. In 2003, a special study was established for

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TABLE 2. Examples of specific applications of landscape concepts discussed in this chapter to rangeland conservation practices discussed in other chapters.

Conservation practice type	Spatial heterogeneity	Spatial pattern	Scaling
Brush management	Shrub encroachment rates vary with ecological site	Grass fragmentation with encroachment determines erosion rates	Mosaics of shrublands and grasslands can be conservation goals for wildlife
Invasive species management	Certain sites are more likely to be invaded first and serve as early warning	Formation of gaps in vegetation may facilitate initial invasion	When scale of infestation is large, it is difficult to control with herbicides
Prescribed burning	Fire prevalence depends on climate	Fuel connectivity determines scale of fire spread	Scale of fire-affected vegetation determines habitat values to wildlife
Prescribed grazing	Certain landforms or soils attract more use by livestock	Patch grazing breaks down with increased stocking rate	Increases in scale of pasture results in less uniform livestock distribution
Range planting	Species plantings more likely to be successful on certain soils	Plantings lead to reduced bare ground connectivity and runoff	Use of key microsites for establishment may lead to spread of seed to larger areas
Riparian management	Geomorphic valley type determines potential riparian function	Spatial pattern of channel (meander width ratio) determines riparian vegetation states	Upland vegetation management affects sediment deposition in riparian areas
Wildlife habitat management	Multiple plant communities may constitute desirable habitat	Fragmentation of forest leads to increased predation or parasitism rates	Dispersal limitation due to landscape fragmentation may override quality of local habitat

rangelands that included field data collection. This will allow detection of more subtle changes in ecological state or condition within rangeland vegetation in future evaluations, but only at very broad scales (http://www.ncgc.nrcs.usda.gov/products/nri/range/2006range.html).

There are two significant limitations to using existing NRI data to evaluate conservation practices. The first is that it is virtually impossible to obtain reliable information about the practices being applied at any particular random point. Although observable conservation practices are recorded, most of the more significant practices are not easily documented without expensive (and often unavailable) landowner interviews. Some of these data do exist, but they are housed in separate databases, are subject to privacy limitations, and generally lack sufficiently precise location information to relate them to field measurements.

The second limitation is that the sampling design was intended to provide estimates of change in major land use and land cover classes at regional to national scales. The design was not intended to detect differences among ecological sites or among variations in the arrangement of practices within a landscape. Such analyses require sampling that is structured with respect to the distribution of treatments and ecological sites in particular landscapes. Analysis must also consider particular models of interactions between locations in a landscape. Further, consideration of within-site heterogeneity may require sampling protocols that are tailored to specific processes (e.g., habitat use by birds). Thus, we assert that NRI is insufficient to provide an understanding of the effects of landscape context on conservation outcomes. This assertion suggests that additional conservation assessment strategies are needed at local to regional levels.

PLACING CONSERVATION PRACTICES IN A LANDSCAPE CONTEXT

Most chapters in this volume review literature to provide science-based evaluations of the effectiveness of individual practices and several of them point to the importance of landscape context. In the last 15 yr, the number of rangeland studies that mentioned "spatial" or "landscape" increased from about 100 to over 700 per year, suggesting a growing awareness of the value of landscape perspectives within the rangeland science community. The number of studies referencing these terms that were associated with conservation practices, however, increased only slightly and is still a small number (< 40/yr). Consequently, this chapter does not on focus on specific practices, but instead offers a forward-looking synthesis of concepts and approaches that could facilitate a shift from the typically site-scale or local perspective toward a landscape perspective (Fig. 1). The local perspective uses information about the ecological state or plant community phase (plant community variants within states of an STM) to select practices to achieve target states or phases, sometimes tailored to an ecological site. The landscape perspective additionally considers how the surrounding landscape—including its states, ecological sites, and connections among them—influence the effects of practices. Attention is also paid to spatial heterogeneity or patchiness within the site as a regulator or objective of conservation effects. Finally, the landscape perspective considers how a localized conservation effect influences the landscape in which it is embedded. Below, we offer a review of several concepts underpinning the landscape perspective. Examples of how these concepts apply to practices discussed in other chapters are presented in Table 2.

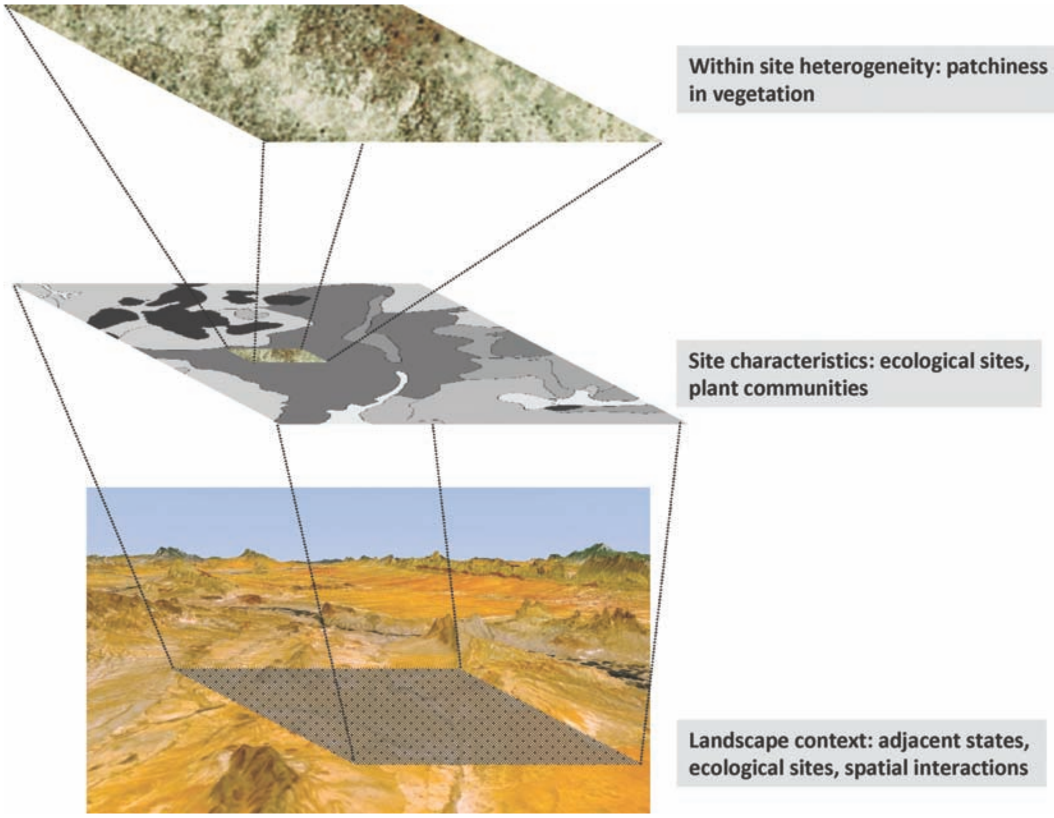
Spatial Heterogeneity

Spatial heterogeneity refers to the variability of properties over space including soils, vegetation, and process rates such as water runoff or erosion (Turner and Chapin 2005). Spatial heterogeneity within rangelands has been accounted for at some scales, but not others. For example, spatial heterogeneity due to relatively static variations in the landscape (reflected in the ecological site) and differences in past vegetation dynamics or "historical legacies" (reflected in ecological states or community phases) is known to influence the success of practices and is widely used in planning (Creque et al. 1999; Bestelmeyer et al. 2009). Different ecological sites or states/phases are predicted to differ in both the possible changes that can be observed and the success of certain management interventions. Different soils vary in their susceptibility to shrub invasion and the loss of perennial grasses, and in the potential for recovery of different grass species (Fuhlendorf and Smeins 1998; Hamerlynck et al. 2000; Wu and Archer 2005). For example, sites with well-developed argillic (clay-rich) horizons may limit invasion by both *Larrea* (McAuliffe 1994) and *Prosopis* (Archer 1995; Miller et al. 2001). On the other hand, the phytotoxicity of chemicals used to control shrubs may also be limited on clay-rich soils (Duncan and Scifres 1983). Once shrubs are removed, the recovery of grasses in response to removal may depend on clay content (Bestelmeyer et al. 2006a) and the degree of soil interspace erosion that is captured in classifications of alternative states (Herrick et al. 2006). The linkage of this type of information to ESDs and maps is potentially very useful. ESDs, however, often do not clearly present information on the likelihood of practice success or failure among ecological sites or states.

> "…a forward-looking synthesis of concepts and approaches that could facilitate a shift from the local perspective toward a landscape perspective"

Spatial heterogeneity may also be of value in its own right, although considering heterogeneity in this way for planning is not common. It has long been recognized that habitat heterogeneity promotes biodiversity, but conventional measurements tend to emphasize within-habitat heterogeneity (such as the vertical complexity of vegetation at a point in space; Tews et al. 2004) rather than landscape heterogeneity (the composition and arrangement of different habitats across a landscape). Landscape heterogeneity in rangelands can be produced by the simultaneous and dynamic coexistence of different plant communities due to patchy disturbances and corresponding asynchrony in successional stages among patches. Such patch dynamics or a "shifting mosaic" (Bormann and Likens 1979) can yield desirable properties at broad scales. For example, shifting mosaics in grasslands caused by grazing and fire disturbances can create a mix of structurally simple and structurally complex plant communities that sustain wildlife populations that exploit resources in different plant communities over time (Archibald et al. 2005; Fuhlendorf et al. 2006). Spatiotemporal variation in vegetation can also be used as a tool to manage livestock distributions (Fuhlendorf and Engle 2004). Some valued wildlife species may even be associated with elements characterizing traditionally undesirable states, such as mesquite shrubs in semidesert

FIGURE 1. A schematic of the landscape perspective. The top panel illustrates patchiness in vegetation cover (dark) interspersed with sparsely vegetated areas (light). The middle panel is a map of ecological sites, each representing one or more vegetation states. The bottom panel illustrates the landscape within which the mosaic of ecological sites is embedded. These are three scales of pattern that promote a landscape perspective.



grasslands (Lloyd et al. 1998; Saiwana et al. 1998) or bare and structurally simplified areas in the Great Plains (Knopf 1996; Derner et al. 2009). Thus, the presence of even persistently altered states in parts of a landscape can have management value. In rangelands, however, management typically seeks to maximize the coverage of a single, desired plant community and deemphasizes landscape heterogeneity (Fuhlendorf and Engle 2001).

Spatial heterogeneity may be similarly important to the basic functioning of ecosystems. Shifting mosaics caused by localized fire and succession are seen as prerequisites for long-term sustainability of desirable forest structures (Allen et al. 2002; Bond et al. 2005). At a finer scale, alternating areas of bare ground that generate water runoff and vegetated patches that intercept runoff can sustain productive vegetation in arid environments where a more homogeneous distribution of vegetation could not be sustained (Noy-Meir 1973; Ludwig and Tongway 1995; Rietkerk and van de Koppel 2008). Consequently, conservation and restoration approaches to increase cover and production of desired species may focus on maintaining or creating heterogeneity and specific spatial patterns in vegetation (Noble et al. 1997; Miller and Urban 1999). In such cases, the existence of patches of bare ground or "early seral" vegetation may not reflect the initiation of ecosystem degradation, as is sometimes assumed, but rather are necessary components of some ecosystems that sustain productivity and biodiversity.

Spatial heterogeneity of the kind noted above is especially important for vegetation sampling. Point samples gathered without sufficient replication or stratification to different patch or community types can lead to misinterpretations about vegetation conditions when extrapolated to the management unit or landscape scale. Sampling that ignores spatial heterogeneity and replication needs is a common source of monitoring failure (Herrick et al. 2005).

We conclude that explicit consideration of spatial heterogeneity in conservation planning and assessment is valuable because it contains information about practice effects. Spatial heterogeneity can also be a goal of practices. The information provided by spatial heterogeneity can only be obtained if our treatment of data moves from reporting averages or percentages to explaining the variability in the data as a function of ecological site, state, and patchiness variables. Such data can then be used to improve ESDs so that they better predict variations in success. Our concepts of ecological sites and states also need to describe spatial heterogeneity as an attribute of interest, rather than solely describing spatially averaged attributes and implicitly treating management for desired plant communities as an "all-or-nothing" proposition.

Spatial Pattern

Without direct consideration of heterogeneity, attributes such as soils, cover, and species composition become nonspatial. Spatial pattern within and among plant communities in a landscape, however, may have predictive value (Reitkerk et al. 2004; Barbier et al. 2006). Whereas spatial heterogeneity describes the condition of possessing dissimilar patch types in an area, spatial patterns communicate where those patch types are and how they are shaped and arranged. Spatial pattern can be measured by the density, size, shape, and location/adjacency of patches and plant communities, typically via a geographic information system (GIS), spatial statistics, and derived maps (Gustafson 1998; Turner 2005). Spatial patterns are descriptors of the potential for—and consequences of—spatial interactions involving the flow of matter (e.g., water, seeds, or animals), disturbance (fire or erosion), or information (cues for habitat selection). They thus communicate valuable information about specific ecological processes that affect the success of practices (Turner 2005).

Spatial patterns in vegetation give rise to variations in the availability of limiting resources or the intensity of disturbance. These variations subsequently alter the spatial patterns—a phenomenon sometimes referred to as "self-organization" (Watt 1947; Rietkerk et al. 2002). Self-organized spatial patterns are the consequences of feedbacks between the initial spatial pattern in vegetation and processes such as water redistribution. For example, a vegetation patch in a matrix of open ground will intercept the flow of water and increase infiltration and plant-available water resulting in increased plant growth. Increased plant growth may enable the patch to intercept more water and so on until resources from the adjacent bare area become limiting. As a consequence of feedback effects, rangeland landscapes often exhibit characteristic spatial patterns. As overall resource availability to the site changes (e.g., due to aridity), the spatial pattern may change in predictable ways (Rietkerk et al. 2004). Disturbances to patches that interfere with feedbacks also lead to predictable effects on spatial patterns and feedbacks (Ares et al. 2003; Kéfi et al. 2007). Large patches tend to become fragmented, leading to decreased resource capture, production, and soil degradation (Wu et al. 2000; Bestelmeyer et al. 2006b). The decrease in large patches also leads to characteristic changes in the distribution of patch sizes in an area. Thus, changes to spatial patterns have been promoted as early warning indicators of rangeland degradation (and presumably restoration success; Kéfi et al. 2007; Scanlon et al. 2007). The value of pattern-based indicators relative to standard measures of ground cover is being debated (Maestre et al. 2009).

The spatial pattern of ecological states, soils, and topography in a landscape governs the flow of resources and disturbances. The impact of practices on potential water yield is especially sensitive to the locations in a landscape where the practices are applied (Ludwig et al. 2005). For example, Wu et al. (2001) found that policy incentives for brush control on the Edwards Plateau need to clearly specify the optimal locations for treatment in order to influence water yield. Spatial specification is important because there are tradeoffs between strategies designed to increase potential forage productivity vs. water yield potential (Redeker et al. 1998). Similarly, the spatial design of infrastructure including the locations of

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"...the spatial pattern of vegetation cover can be even more important than the average amount of cover in determining runoff and erosion under some conditions."

fences, watering points, and feeders are used to modify patterns of animal movement and forage utilization, taking into account livestock behavior and the template of topography and plant communities to which livestock respond (Laca 2009). In this way, the spatial locations of rangeland infrastructure can have a large, indirect impact on overall vegetation change. Similarly, the spread of fire in a landscape depends upon where the fire is initiated relative to the spatial arrangement of ridges and valley bottoms (Swanson et al. 1988) as well as the connectivity of fuel loads (Allen 2007).

The spatial pattern of vegetation patches resulting from practices, in turn, affects key processes and services. For example, Ludwig et al. (2007a) showed that the spatial configuration of vegetated and bare patches had a significant influence on erosion and sediment loss in northeastern Australia. A catchment with a coarse-grained patch structure (few large patches) and 54% grass cover had 43 times greater sediment loss than a catchment with fine-grained patch structure (many small patches) and 43% grass cover. This result suggests that the spatial pattern of vegetation cover can be even more important than the average amount of cover in determining runoff and erosion under some conditions. With similar amounts of total cover, the pattern of vegetation patches comprising this cover can be arranged such that they either slow runoff and retain sediment or allow it to leave the site. Larger bare patches, bare patches that are elongated parallel to the direction of flow, and bare patches that occur lower on a hill slope are less able to slow the movement of water and sediment and prevent its transfer into channels.

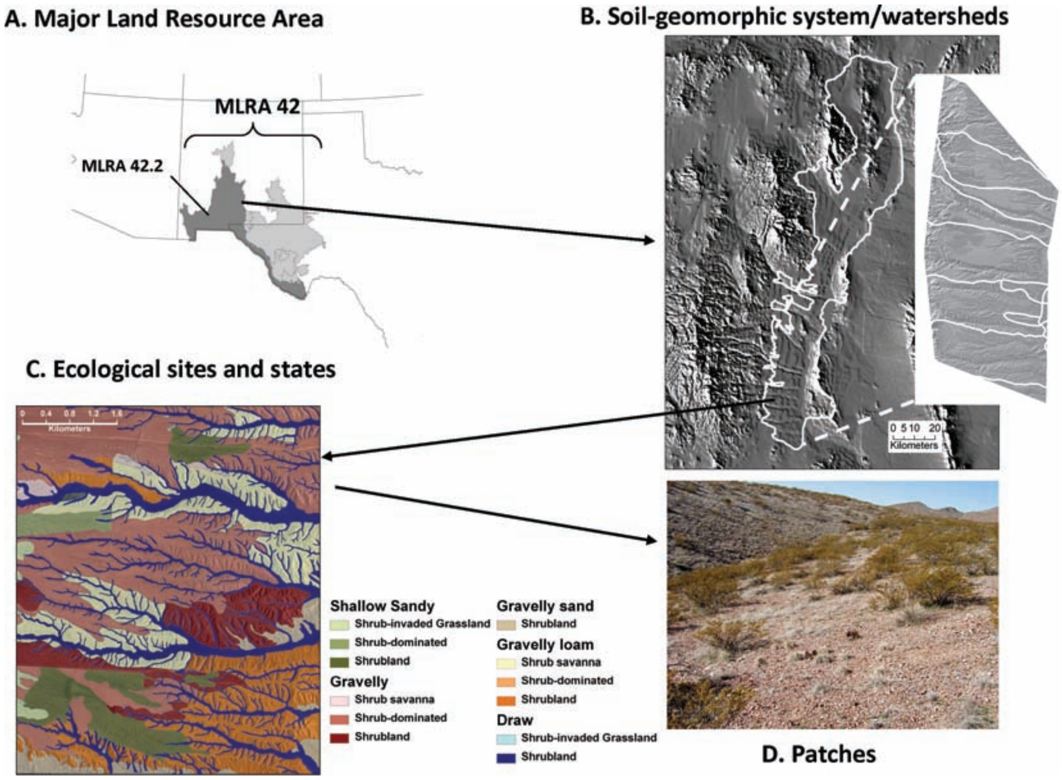
Water and sediment loss rates at sites with intermediate values of cover close to "percolation thresholds" can be very sensitive to changes in plant cover (Davenport et al. 1998). Percolation thresholds describe how a small change in cover over a defined area can result in a cover type becoming connected with respect to a process, such as the spread of fire (Turner et al. 1989) or the dispersal of species through certain cover types (King and With 2002). The shape and size of patches comprising cover affects the critical threshold value, but in general the shift from fragmented to connected occurs at intermediate cover values (Miller and Urban 2000). Areas with very low cover are necessarily fragmented and areas with very high cover are necessarily connected via a single large patch. Spatial pattern matters most when cover is intermediate because of the wide variety of possible arrangements of patches (Gergel 2005). Thus, attention to changes in connectivity can help us understand nonlinear relationships between cover and ecological processes in some conditions (Peters et al. 2004).

These examples suggest that various measurements of connectivity (and its converse, fragmentation) could be used to estimate critical ecological processes that mediate the effects of practices (Debinski and Holt 2000; Tischendorf and Fahrig 2000; Goodwin 2003). Some specific measures include the frequency of patches of different size or weighted mean patch size (Li and Archer 1997), the aggregation index (He et al. 2000), and the landscape leakiness index (Ludwig et al. 2002, 2007b) as well as several others (McGarigal and Marks 1995; Calabrese and Fagan 2004). As one example, we would expect that decreasing connectivity of vegetation patches and increased connectivity of bare patches would be reflected in a "landscape leakiness index." Increases in this index are correlated with reduced water infiltration and nutrient retention. In selecting a metric, it is important to link it to a conceptual model of the pattern–process relationship, which, in turn, should indicate the appropriate spatial scale at which the metric is measured.

Biophysical Scaling Effects

The spatial extent of observation determines how we perceive natural resource problems as well as the practices we use to solve them. For example, woody plant encroachment in landscapes of South Texas was shown to be strongly scale-dependent. In areas encompassing multiple soils, woody plant cover was associated with high-clay soils and wetter portions of the landscape (Wu and Archer 2005). At finer scales within upland soils, woody plant cover was negatively related to soil clay content and was unrelated to surface hydrology (Archer 1995; Wu and Archer 2005). Thus, the correlates of woody plant dominance depend upon the spatial scale of measurement. Correlations with particular

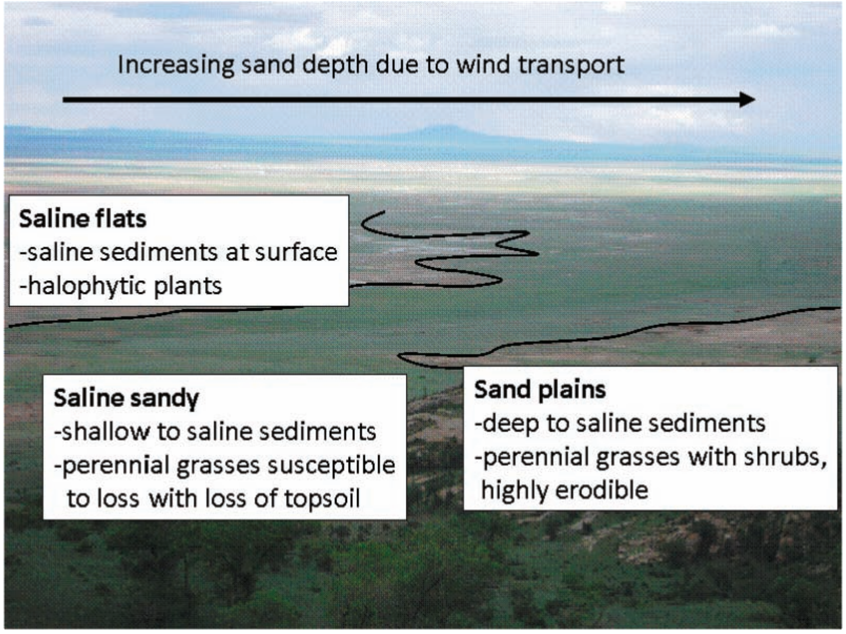
FIGURE 2. Example of a spatial hierarchy in a Chihuahuan Desert rangeland, following Table 2 and Bestelmeyer et al. (2009). **A**. Major Land Resource Area (MLRA) 42 and the Land Resource Unit MLRA 42.2, shaded. **B**. A soil–geomorphic system along the Rio Grande River valley comprising relict piedmont, ballena, and inset fan landforms associated with the Pleistocene–Holocene entrenchment of the river. The inset shows detail on soil map units and watershed boundaries (US Geological Survey hydrological unit code 11), illustrating the repeating patterns along the north–south axis. **C**. Each ecological site is represented by a different primary color. Various states occur within these delineations, including shrub savanna, shrub-dominated, and shrubland. States within sites are different shades of the primary color. **D**. Patches of *Bouteloua eriopoda* grass occurring in the interspaces of *Larrea tridentata* shrubs, a spatial pattern in the shrub-dominated state that indicates restoration of grass cover is possible.



variables (e.g., clay content) can change sign across scales due to interactions with factors that vary at broad scales (e.g., topographic position overrides the negative effect of clay on woody plant cover). Management models used to predict patterns of woody plant encroachment thus need to recognize the scale dependence of variables governing encroachment.

Different animal species or groups respond to rangeland attributes that are measured at different spatial scales. For example, although habitat quality for many bird species focuses on local vegetation structure, practices designed to promote highly mobile wetland bird species should focus on the distribution of a spatially dispersed mosaic of sites that are used at different points in the annual cycle (Haig et al. 1998). Considering the response of insect communities to grazing and mowing for hay in tallgrass prairie, Stoner and Joern (2004) showed that the species diversity of generalist and herbivore insect guilds in prairie fragments was largely controlled by local (within-fragment) plant community composition. This suggests that practices should focus on plant community attributes at the local scale to maintain populations of these insects. The predator insect guild, however, responded more to broader-scale attributes such as the shape of the fragments, thereby producing indirect effects on the other generalist/herbivore guilds. Attention to both fragment quality and fragment shape would be important conservation objectives in this case.

FIGURE 3. An example of ecological site concepts developed for an area near Dalinour Lake in the Inner Mongolia autonomous region near Xilinhot, China. The ecological site boundaries are defined by varying depths of sandy sediments over saline lake sediments that are associated with shifts in vegetation and vegetation change/soil processes. Because the three ecological sites are coevolving in response to a common process (wind-driven deposition of sand over saline lake sediments) they can be considered as part of the same soil–geomorphic system.



The spatial scale of observation also determines our ability to perceive processes that link the success of local practices to conditions in adjacent parts of a landscape. This perception is important when practices carried out at a fine scale influence adjacent areas due to spatial interactions. Conversely, the consequences of a practice within a local area may depend on influences from adjacent areas; both of these effects are termed "cross-scale interactions" (Peters et al. 2004). Clearly, some of the primary benefits of erosion control occur off-site in the form of decreased erosion in adjacent sites, improved water quality, and decreased siltation of streams and reservoirs (Pimentel et al. 1995; Pringle et al. 2006). Cross-scale interactions can also thwart local management efforts. Historical overstocking and drought initially converted extensive areas from perennial grassland to eroding shrubland until the 1950s in the Jornada Basin of southern New Mexico. Once the eroding shrublands became sufficiently extensive, many remaining grassland areas were converted to coppice-dune shrublands even when domestic grazers (and native grazers) were excluded via fencing (Peters et al. 2006). Grassland-to-shrubland transitions after the 1950s had become decoupled from the local processes that had previously caused them. Instead, they were controlled by broad-scale erosion and sediment movement that led to local soil instability with abrasion, burial, and mortality of grasses, occurring even in ungrazed areas (Okin et al. 2009). In such cases, the local management of vegetation or soils may not be adequate to predict the trajectory of vegetation change, as is often assumed in the use of assessment and monitoring indicators. A characterization of the functioning of the broader landscape would be required.

These examples indicate that in addition to variation in the properties of a specific land area (e.g., its ecological site or state) planners should carefully consider the landscape context within which an area is embedded. Furthermore, carefully chosen intervention points can induce nonlinear responses or emergent effects that are not predicted based on a simple linear scaling of the areas that are treated. One example might be to increase grass cover in critical portions of a watershed to reduce watershed-scale sediment loss. In both cases, there may be critical spatial scales at which the effects of key spatial interactions can be predicted, determined by factors such as geomorphology, hydrology, or species behavior (Turner et al. 2001). Our ability to predict these scales remains limited, but planners can integrate expert judgment with GIS to make informed (and testable) predictions.

Societal Heterogeneity and Scaling

Consideration of biophysical scaling will often lead planners to look to scales larger than a management unit. In these cases, the identity, heterogeneity, and spatial arrangement of management units in different ownership or tenure must be considered. Activities on one management unit may have off-site effects on adjacent units that are unrecognized, or diffuse effects from multiple units may influence attributes of communal interest, such as water table depth (Swallow et al. 2001; Standish et al. 2009). Collaborative approaches are thus necessary precursors to broad-scale practices such as fire management or species conservation (Sayre 2005). In order for multiproperty practices to be successful, an atmosphere of trust and cooperation is required, calling for careful attention to an inclusive process in strategic and tactical planning (Duff et al. 2009). Much like the processes that lead to patchiness and sustainability in vegetation, "self-organized" groups of interested property owners working with agency representatives and scientists (e.g., prescribed fire associations) can lead to successful broad-scale conservation efforts (Biggs and Rogers 2003). Ultimately, however, successful practices must foremost be perceived to benefit individual landowners (Swallow et al. 2001). As with the use of ecological sites and states, it would be useful to document the societal contexts within which certain practices succeed or fail as a means of developing more effective approaches to conservation planning (Paulson 1998).

A SPATIAL HIERARCHY FOR CONSERVATION PLANNING AND EVALUATION

The preceding review makes a compelling case for the value of landscape perspectives in conservation planning and assessment, but how can we most effectively incorporate these perspectives? Some approaches, including the use of spatial simulation models, are too technically complex to be widely implemented at the present time. We suggest that informed judgment combined with GIS and selected use of some existing models (e.g., hydrologic models) provide a practical means to develop landscape perspectives. The concepts described here rely on spatial data. Such data can be used to detect patterns at different scales and then design and evaluate practices based on the patterns. Mapping activities—usually of management units, vegetation, and ecological

Water running in a rill after a rainfall event within a desertified grassland, southcentral New Mexico. (Photo: David Toledo)

TABLE 3. General levels of the land hierarchy discussed in this chapter, distinguishing characteristics for each level, approximate map scales, and analogous levels in the National Hierarchy of Ecological Units and Terrestrial Ecological Unit Inventory of US Forest Service. Entries in parentheses are not formal levels but are discussed in literature. This hierarchy mixes a spatial hierarchy (Major Land Resource Area to watershed) with elements of a classification hierarchy (ecological sites to patch) that can be delineated as nested spatial units.

General level used in this chapter[1]	Distinguishing characteristic	Map scale	USFS[2]
Major Land Resource Area	An area of similar gross physiography and continental weather pattern	1:3 500 000	Section
Land Resource Unit	A class or area based on regional climate variation or geology within Major Land Resource Areas; may or may not be spatially explicit	1:1 000 000	Section/subsection
Soil–geomorphic system	An area of similar geology and linked geomorphic/biotic processes that control landscape evolution	1:250 000	Land-type association
Watershed/Airshed	An area that is internally connected by a dominant spatial interaction (typically water flow, but could be eolian soil redistribution, fire, or animal movement)	~ 1:100 000, variable (hydrologic unit code 11)	(Watershed)
Ecological site	A class of land of similar potential vegetation, soil, geomorphic setting, topographic position, and microclimate	1:24 000 to ~ 1:150 000	Ecological type
Plant community/state	An area of similar plant species composition	~ 1:5 000 to 1:12 000	Plant association/ structural stage
Patch	A discrete unit of homogeneous vegetation and soil surface properties, ca. 1–100 m^2	1:1	(Patch)

[1]USDA NRCS (2003a) and Bestelmeyer et al. (2009).
[2]Winthers et al. (2005) and Cleland et al. (1997).

sites—are already part of planning and assessment process. Our recommendation is to plan and evaluate conservation practices with regard to multiple hierarchical levels of pattern in rangelands (Fig. 2) and forge more explicit connections with existing databases on soils and ecological sites. First, we describe a series of hierarchical levels in rangelands (from fine to broad scales) and the data that can be used to represent them (Table 3).

Patches

A patch is a relatively homogeneous area, often defined by local aggregations of plants or the absence of plants (e.g., a bare ground patch). Patch is a concept that can be used at any scale depending on the process or species of interest. The use of patch in this document is similar to the concept of the "pedon" used to describe a homogeneous unit of soil. Patch spatial patterns (the size, arrangement, and composition of patches) at fine scales (e.g., 0.1–1 ha) are used to define patterns within a plant community that affect specific processes such as erosion or habitat use. Information about patchiness serves three functions: 1) it changes our predictions when compared to the default assumption of uniformity (e.g., patchy vs. uniform grazing pressure), 2) it can be an objective of management (e.g., using fire to increase habitat heterogeneity), and 3) once recognized, patchiness can be altered to affect processes (e.g., to decrease landscape leakiness). For example, patchiness in grazing pressure can produce localized changes in vegetation that would not be predicted assuming a uniform grazing distribution, including changes considered to be useful or to be degradation (Adler and Lauenroth 2000; Augustine 2003). Patchiness produced by grazing, fire, or their interaction has been promoted as a means to increase biodiversity (Fuhlendorf and Engle 2001). Practices can also take advantage of existing patchiness; erosion control structures can connect vegetation patches to form large obstructions to overland flow (see Ludwig et al. 1997; Reid et al. 1999). Slash additions combined with seeding may initiate the development of grass patches in eroded areas (Stoddard 2006) and the new grass patches may then expand over time.The alteration of patch spatial patterns can be measured using a number of tools. Ground-based transect approaches, including gap intercept, measure changes in the frequency distribution of fine-scale bare patches (Kuehl et al. 2001; Herrick et al. 2005). Aerial photography or high-resolution satellite imagery coupled to image classification are used to map vegetated patches (Bastin et al. 2002; Laliberte et al. 2004; Bestelmeyer et al. 2006b) and calculate a variety of patch metrics (Gergel and Turner 2001). More easily, Google Earth can be used to detect patch patterns across the globe and can now be linked to traditional GIS shapefiles (http://earth.google.com). Finally, the USDA Agricultural Research Service (ARS) Jornada Experimental Range has produced simple nominal and ordinal indicators that capture patch spatial patterns and associated soil redistribution processes (http://jornada.nmsu.edu/sites/default/files/FieldGuidePedodermPattern.pdf). The success of these approaches depends upon the spatial grain and the detectability of the patches involved. Measurements of patch spatial patterns (types, sizes, density, connectivity) could be explicitly defined as objectives (or preconditions) for state or Major Land Resource Area (MLRA)-level practice guidelines.

Plant Community

Plant communities, considered as spatial units, are assemblages of plant species and patches that exist at a particular place and time (Vellend 2010). Different communities in a landscape can be distinguished based on spatiotemporal shifts in the composition and abundance of species. Plant communities can be classified to states based on their responses to natural and management drivers and ecological sites may exhibit one or more plant communities (phases) or states (Bestelmeyer et al. 2009; Fig. 2). Practices are expected to produce or limit shifts among communities occurring in different states (accelerating practices) or produce shifts among communities within a state (vegetation management or facilitating practices; Stringham et al. 2003). Thus the identity of a community carries with it explicit predictions about its likely response to a practice.

In addition to the obvious role of plant communities as planning tools and assessment strata, data on the success of practices could be linked to communities and states in the Ecological Site Information System. These data can be used to refine STMs. Classifications of plant communities can also be linked to the responses of key animal species (Holmes and Miller 2010). Plant communities can be mapped using vegetation maps based on standardized vegetation classifications available through some gap analysis programs and other detailed mapping efforts (http://www.natureserve.org/prodServices/ecomapping.jsp). It is important to recognize, however, that coarse vegetation maps may combine several plant communities (and states) that are distinguished in STMs. Alternatively, plant communities can be mapped directly against a background ecological site layer using aerial photography or satellite imagery and derived spectral indices, often resulting in map units featuring associations or complexes of communities.

Watersheds in Kalalau valley, Kauai, Hawaii. (Photo: Brandon Bestelmeyer)

Currently, maps of plant communities are seldom available so they can be produced as needed for setting landowner objectives and implementing practices.

Ecological Site

Ecological sites are classes of land that differ in potential natural vegetation (Landres et al. 1999), historical range of variation, and response to disturbance as a function of differences in soils, landforms, and climate (USDA NRCS 2003a). The Terrestrial Ecosystem Survey of the US Forest Service provides land units similar to ecological sites (Winthers et al. 2005). Ecological sites are nested within climate-based classes called Land Resource Units (LRUs) or MLRAs (similar to ecoregions). In conjunction with its STM, the ecological site communicates the breadth of possible plant communities known to exist on a site. Even when STMs are similar, soil variations represented in ecological sites may influence the effects of management. Examples include the success of herbicide use with varying soil clay content, or variation in the success of grass seeding with climate variation among LRUs. Thus, planning and evaluation should be linked directly to the ecological site, and better still, to local information on soil/landform variations within ecological sites (Bestelmeyer et al. 2009). In this way, the classification of ecological sites can be updated to better reflect differences in ecological resilience or other responses, or the effects of important variations within ecological sites can be described.

Ecological sites are correlated to soil map unit components and are represented spatially via soil map units of the National Cooperative Soil Survey (e.g., http://soildatamart.nrcs.usda.gov). Given the scale of soils mapping in many rangelands, soil map units usually have a one-to-many relationship with soil components. As a result, soil map units describe soil complexes, associations or consociations that often translate to multiple ecological sites per soil map unit. For the purposes of initial stratification, many soil map units can usefully be classified according to the spatially dominant ecological site within them while recognizing that they are not necessarily homogeneous. Visual cues obtained in the field (e.g., surface soil color, gravel, slope) can be used to more precisely classify areas.

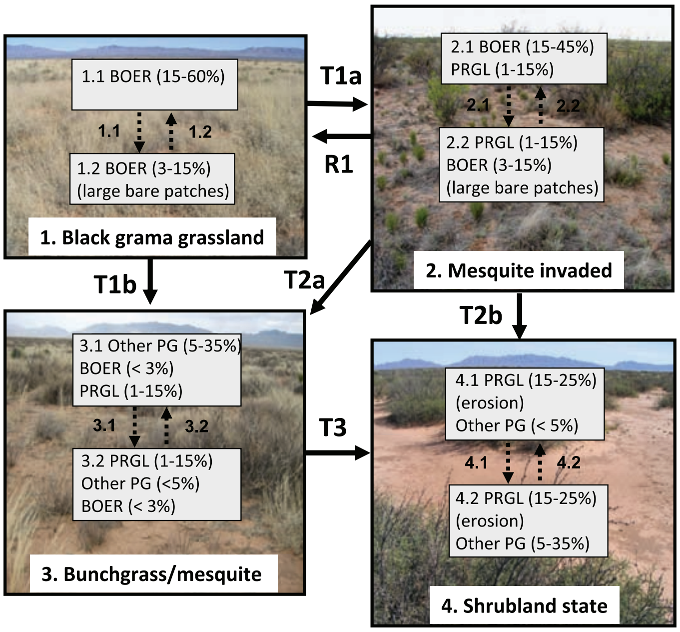
Watersheds/Airsheds/Firesheds

Ecological site and state units in many landscapes are connected to one another via hydrology and eolian transport or potential fire spread. Thus, management in one unit will impact others in connected landscapes. Watershed manipulations are often designed to take advantage of these connections; this has been reviewed elsewhere (Williams et al. 1997). To understand hydrological consequences, the appropriate order (or scale) of a watershed (i.e., Hydrological Units of the US Geological Survey) should be specified, alongside an expectation of the response to a hydrological manipulation at different scales and in specific parts of the watershed. Such predictions are more common within riparian zones compared with upland areas affecting riparian zones (Goodwin et al. 1997). Informal conceptual models or distributed hydrologic models can be used to develop such predictions. The concept of airsheds may be especially important in arid zones where wind erosion and sediment deposition processes are important (Okin et al. 2006). Airsheds could also assist prediction of the movement of smoke from prescribed fires. Similarly, "firesheds" have been conceived as the possible or expected area influenced by a single fire ignition as constrained by natural barriers, fuel, terrain, and weather within a given period of time (Stratton 2006). Formal units for the latter types of "-sheds" may not exist, but can be estimated from models or in a GIS. Accounting for physical connections is a key element in developing estimates of off-site effects of practices.

Soil–Geomorphic Systems(SGSs)

The SGS is a new concept and refers to a discrete land area with a characteristic spatial arrangement of ecological sites (and often plant communities) that are linked by fluxes of materials, organisms and disturbances, soil-forming processes, and ecological processes (Bestelmeyer et al. 2009; see Figs. 2 and 3). They are similar in scale to the landtype association of the National Hierarchy of Ecological Units (Cleland et al. 1997). Land areas within an SGS feature similar landscape organization and may encompass multiple watersheds or airsheds (depending on their scale). The interaction of management and soil or landscape attributes should be similar across an SGS. In other words, the rules

FIGURE 4. An example of a state-and-transition model developed for the Sandy ecological site, Major Land Resource Area 42 and Land Resource Unit 42.2. BOER = *Bouteloua eriopoda* (Torr.), Other PG = other perennial grasses (including dropseeds, *Sporobolus* spp.), and PRGL = *Prosopis glandulosa* (Torr.). Cover values (%) reported are foliar canopy cover from a variety of datasets. Key portions of state narratives relating to spatial patterns are highlighted to illustrate their use (right column). Specific predictions for transitions (T) and restoration pathways (R) occur below the figure.



State 1. Appearance of large (> 1m) bare patches signals risk of limited black grama recovery

State 2. Mesquite may preferentially invade in bare patches and then populations expand

State 3. Dropseeds die out in drought, ample opportunities for mesquite establishment

State 4. State spreads due to cascading soil erosion.

T1a. Mesquite establishment facilitated by seed transport by cattle, bare patches > 50 cm, and relatively wet springs
R1. Shrub removal with black grama recovery to > 15% may inhibit reinvasion
T1b, T2a. If black grama is reduced below ca. 3% cover (grazing in drought), it will not recover dominance
T2b, T3. At perennial grass cover < 5%, wind and storm events can trigger deep, spreading soil erosion

governing spatial interactions and determining the success or failure of a management action in a land area are similar within an SGS and will differ in distinct SGSs. Additionally, the spatial scales at which responses will be manifested, and should be monitored, can also differ among SGSs. SGSs can be used to tailor management and monitoring programs to landscapes that are structured differently. The extent of SGSs can be hand-digitized in a GIS using a digital elevation model and geology maps alongside a basic knowledge of hydrology and geomorphology or created by aggregating State Soil Geographic (SSURGO) database soil map units (sometimes State Soil Geographic (STATSGO) database map units can be used).

MLRAs and LRUs

Considerations at these scales are similar to those at the ecological site scale. It is useful to understand the location of an ecological site within a MLRA or LRU, given the continuous variations in climate that exist across the extent of these broad areas. Modeled climate products with national coverage, such as the PRISM model (Daly et al. 2002) can be used to quantify within-MLRA/LRU variations. The types of practices used and their outcomes vary strongly among MLRAs/LRUs, so it would be useful to assemble guidelines at these levels. For example, rangeland seeding has been recommended in the 10–14-inch precipitation zone (LRU) of MLRA 35 (Colorado Plateau) but not in the 6–10-inch precipitation zone.

A MODEL-BASED, LANDSCAPE APPROACH TO CONSERVATION PRACTICES

Using the spatial data and concepts described above, we recommend that the following steps be considered to design and test the effects of practices and then link what we have learned from these tests to an expanding database. Planning starts with collaborative development of a conceptual model of the intended effects and ends with an update to the model, paralleling statistical approaches that are advocated for adaptive environmental management (Ellison 1996). In this case, the "model" includes recognition of spatial heterogeneity, spatial pattern, and landscape context.

Define Boundaries of the Management Area and Critical Natural Resource Issues

This is perhaps the most important step to incorporate biophysical and societal scaling. Knowledge of the primary conservation problems, the biophysical and social mechanisms involved, and the scales of spatial interactions associated with those mechanisms are used to delineate the spatial extent of a management area. General information about MLRAs, LRUs, and SGSs can be used to identify the types of mechanisms and critical scales that characterize an area (based on patterns in soils, geomorphology, hydrology, and climate) and therefore, the land area that needs to be considered to solve a problem. The extent of the management area alongside patterns of land ownership is defined in a GIS. Patterns of land ownership within the focal area will determine what resource concerns can be addressed on individual properties and therefore, which conditions can be expected to improve.

Collaborating stakeholders and planners then identify and prioritize natural resource problems and the specific locations of interest. The causes of the problems are identified, and historical perspectives on ecosystem conditions and drivers can help to recognize the key issues. Participatory mapping exercises (Reed et al. 2008) and workshops structured around general conceptual models of land change for an area (Reynolds et al. 2007) are useful approaches. The goal of this step is to focus limited resources on the highest-priority problems.

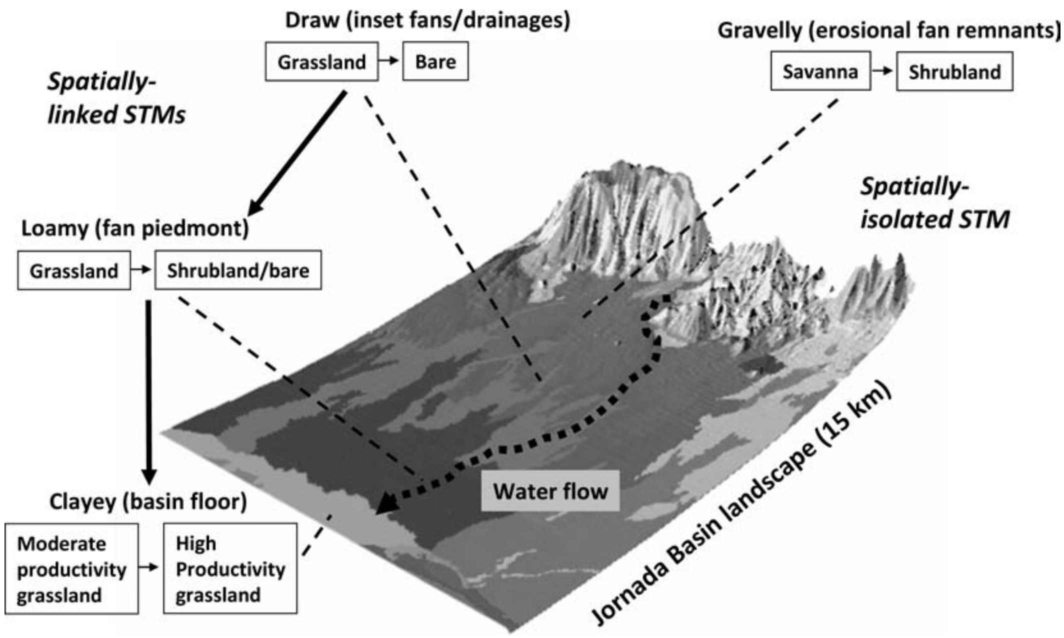
Develop Models of Conservation Effects

Soil or landform mapping is used to identify and locate the set of ecological sites present within the management area. This activity effectively stratifies the management area according to different conservation objectives and expected responses to practices. Each ecological site is then linked to a specific STM that describes the plant communities that are possible for each site and the drivers or interventions needed to achieve them. Ecological sites and STMs may already exist for the management area. Alternatively, ecological sites and STMs may need to be developed by project personnel in cooperation with NRCS and other agencies. Existing STMs must often be expanded to provide explicit predictions about practice effects.

A primary means to develop or expand STMs that serves the design of conservation practices is to examine historical applications and reconstruct their effects. This can be accomplished in many areas via comparisons of historical aerial photography and sometimes via ground-based data or photography. Local knowledge on how the practices were applied and information on the initial state and ecological site are essential. It would be useful to store information on past effects of conservation practices for each ecological site and state. Such a database does not yet exist at the national level, but the Land Treatment Digital Library provides a model for such a database (http://pubs.usgs.gov/fs/2009/3095) and similar databases could be developed locally. Additional sources of information include inventory data of the properties of plant communities associated with the same ecological sites and different management histories, recent monitoring data including responses to climate change and management interventions, and process-based studies that test for the mechanisms causing or constraining ecosystem responses, often associated with long-term research sites.

ESDs and STMs are used to subdivide the landscape according to conservation objectives and to specify the target states or plant community phases for each ecological site. A reasonable target depends partly on ecological potential, which depends on soil variations reflected in ecological site classification (e.g., the depth to saline sediments strongly affects the potential composition of plants; Fig. 3). The selection of targets and practices also depends upon either the risk of degradation or the nature of restoration thresholds that must be

FIGURE 5. A schematic of the relationships between state transitions (boxes and arrows) in state-and-transition models (STMs) associated with different ecological sites sharing a spatially interactive landscape. Transitions within the hydrologically isolated STM (Gravelly ecological site) have no effect on other transitions. A transition within the Draw ecological site has cascading effects downslope in the linked STMs. See text for additional explanation (from Bestelmeyer et al. 2011).



"Spatial data on states and ecological sites are the critical elements needed to connect predictions to specific sites and to assess spatial interactions across a landscape."

overcome to achieve the target state. STMs can be used to formally define predictions about the responses of a discrete land area to conservation practices (Fig. 4). Most important, the STMs then become the repository for information learned from monitoring of practice outcomes. STMs linked to ecological site classifications function as evolving libraries and the interface between knowledge and action.

Although not generally available in existing STMs, local and landscape spatial patterns may be described as attributes defining at-risk community phases or alternative states (e.g., Ludwig and Tongway 1997; Fig. 4). For example, the presence of large open ground patches may signal an increased risk of invasion. Indicators of risk may also occur elsewhere in the landscape. For example, a head-cut gully several kilometers away might soon affect the vegetation and hydrology of an upslope area.

Identify Natural Resource Goals Across Ownership Boundaries

When conservation objectives suggest that cross-boundary coordination will be needed, stakeholders may differ in their preferences and perceptions of tradeoffs. Goals must sometimes be negotiated alongside building of trust between coordinating parties. Kitchen-table to community-level discussions and gradual consideration of the options are essential.

Develop Maps of Ecological Sites, States, and Landscape Models

Spatial data on states and ecological sites are the critical elements needed to connect predictions from STMs to specific sites and to assess spatial interactions across a landscape. Several tools currently exist to support this (e.g., Web Soil Survey, http://websoilsurvey.nrcs.usda.gov; Soil Web, http://casoilresource.lawr.ucdavis.edu/drupal/node/902). In relatively small areas, the solution is simple: conduct field assessments of vegetation and other state attributes (e.g., soil surface properties, patch patterns) alongside verification of the ecological sites. In large landscapes, we have used aerial photography and other layers (e.g., digital elevation models, soil maps)

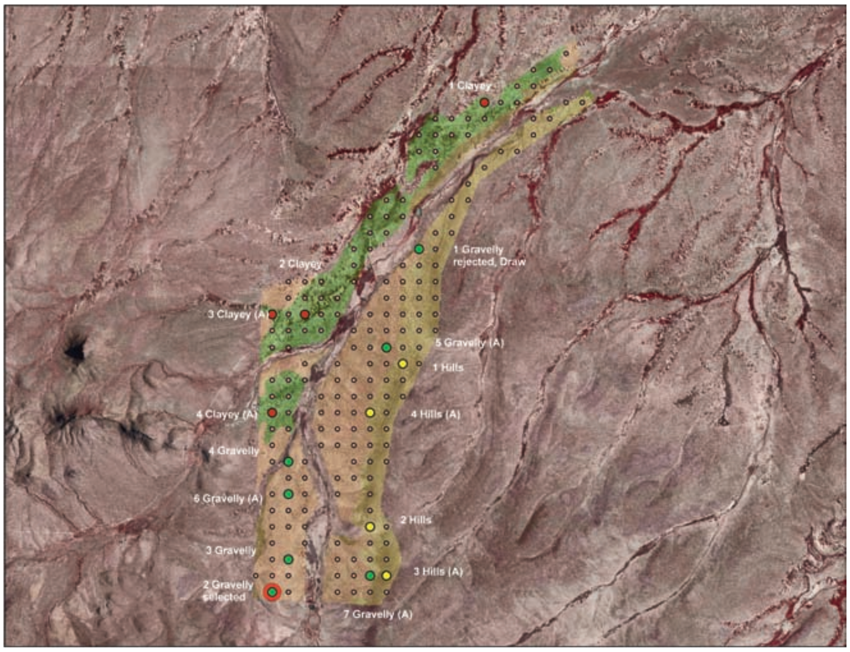


FIGURE 6. An example of a conservation effects assessment monitoring design within a brush management area treated with herbicide. The treated area was classified as having two ecological sites (green = Clayey, tan = Gravelly). Subsequent field assessments identified the Hills ecological site as an inclusion within one Gravelly polygon. ArcGIS was used to create a grid of systematically distributed points. A random number generator was used to select a subset of points for assessment of treatment efficacy and low-intensity monitoring (shrub mortality assessment and repeat photography points) in each ecological site (colored circles). One random subset point located in a nontarget ecological site (a draw that was misclassified in the spatial dataset) was rejected for use. An intensive monitoring point (outlined in red) in the spatially -dominant ecological site was randomly selected from the subset points at which standard monitoring procedures were employed.

in GIS to produce maps of ecological states and ecological sites (hereafter a "state map"). In using this approach in arid ecosystems, staff members at the Jornada Experimental Range have delineated map units that were believed to be internally homogeneous with respect to ecological site and state, but the identity of the state or ecological site was uncertain due to data limitations. Thus, they used the map to structure rapid field assessments and subsequently attributed the polygons. Assessments that were coupled to state mapping allowed a trained technician to evaluate the ecological site and state of 13 000–25 000 ha · day^{-1}.

The resulting maps can be combined with conceptual landscape models of likely spatial interactions or with GIS-based landscape models (e.g., Soil and Water Assessment Tool; Di Luzio et al. 2004) to predict how practices will influence broader scales via spatial interactions. As a simple example of the former, a planner can consider how STMs operating on different ecological sites may be linked within a watershed via a map of soils and a digital elevation model (Fig. 5; Bestelmeyer et al. 2011). Some models associated with hydrologically isolated landforms (e.g., Gravelly sites on erosional fan remnants, sensu Peterson [1981]) need not involve consideration of interactions with other models. On other sets of landforms, transitions among states are linked among ecological sites. A transition from a grassland to sparsely vegetated or bare state in a draw (inset fan) would result in a shift to shrubs and grassland species tolerant of drier conditions in a downslope ecological site (e.g., Devine et al. 1998). The sparser cover and increased runoff from this site, in turn, might lead to increased production at the edge of a Clayey basin floor and a shift from drier- to wetter-adapted grass species (e.g., Peters et al. 2006). Thus, in this example, a practice to repair the gully in the draw might result in both desirable and undesirable transitions in downslope ecological sites. This general set of interactions would operate throughout the SGS. In this way, the linkage of state and ecological site maps to landscape models can be used to delineate land units that require different practices and predict how they and adjacent units will respond.

Design Practices for Individual or Multiple Combined Land Units

For each land unit or for groups of land units that interact spatially (e.g., via hydrological connections), practices are specified to maximize the likelihood of successful maintenance of, or restoration to, the target state or phase. Where possible, an experimental component can be included using matched controls and pre-intervention measurements to allow a before–after–control-intervention statistical design (Block et al. 2001), preferably over suitable periods of time. Decisions are made for every land unit, including the decision not to intervene. The rationale for these decisions, based on the STM and stakeholder goals, can be stored with the state map database in ArcGIS software.

Monitor and Update ESDs

Monitoring stratified to different land units can test both the effectiveness of the practice application and the effects of the practice given its successful application. The monitoring program should be able to distinguish between application effectiveness and effect given successful application to provide fair evaluations of the causes of failure. Stratification by ecological state, ecological site, and surrounding states and sites allows context-dependent tests of practice effects. There should also be careful consideration of the hypothesized response attributes and timelines for change. Without careful consideration of these design elements, monitoring programs are often incapable of providing valid tests of practice effects.

In keeping with the collaborative nature of this approach, the interpretation of the monitoring data should be discussed among planners, science specialists, and stakeholders. Because the effects of intervention often unfold over long time periods and are influenced by short-term climate variability and other events, the results are sometimes not straightforward to interpret. The limitations of the data obtained at any given time should be recognized and interpretations can evolve with additional data.

The evidence obtained is used to modify or revise the appropriate STM, ecological site classification, and local landscape or general SGS models. As a result, the criteria for states and ecological sites may be changed and the likelihood of success of a practice within a state or ecological site can be quantified. The attributes of state maps can be updated and subsequent practices modified.

An Example: What Are the Benefits of the Model-Based, Landscape Approach?

The sequence of activities discussed above are new proposals, therefore we cannot provide direct evidence of their effectiveness. We can, however, provide an example of how they are currently being applied and the benefits we anticipate. The USDA ARS Jornada Experimental Range has worked with the Bureau of Land Management (BLM) in New Mexico to evaluate the effects of brush management practices that have been supported by both BLM and USDA EQIP funds as part of the Restore New Mexico program. Initial meetings with BLM staff were used to create explicit descriptions of the expected benefits of brush management. These meetings were also used to specify explicit hypotheses for vegetation responses in different ecological states and ecological sites. Digital maps were then used to design the brush control treatments. Soils, landform, and the pattern and cover of ecological states from aerial photography were interpreted according to STMs to identify suitable treatment areas within selected allotments. This selection procedure was based on interactions between BLM staff and grazing permittees. For example, shrubland states on thin, rocky soils are unlikely to yield much herbaceous response from brush control and were avoided in favor of slightly deeper soils. They used a simple spatial pattern indicator—the aggregation of perennial grasses under shrub canopies— to identify suitable states for treatment during field visits. They focused on states where remnant perennial grasses were distributed throughout shrub interspaces and in which vegetative recruitment could lead to rapid recovery (Bestelmeyer et al. 2009).

Once brush control treatments were applied, the same spatial data were used to design a monitoring program. Spatial data were used to stratify plots to target ecological sites and states and then randomly select plots within target ecological sites to achieve a spatially balanced sample (Fig. 6). They also established sampling plots in areas outside of treated areas on the same leased properties to evaluate changes to herbaceous cover that may occur when stock numbers are redistributed to other pastures due to grazing deferments in treated pastures. The size of the monitoring units (50-m transects) and monitoring methods were chosen considering the size and distribution of remnant grass cover patches. Line–point intercept, gap intercept, and belt transects (Herrick et al. 2005) are being used to monitor trends in herbaceous plant recovery and shrub mortality and recruitment, tailored to the expected responses of the brush control treatment. We have planned to obtain repeated readings over a 12-yr time horizon; desert grassland recovery is slow at best. Across the body of brush control “planning data entered by field staff need to be integrated with spatial data and followed up by monitoring in treated areas as well as offsite areas”

treatments (over 50 sites), herbaceous response will be modeled as a function of soil properties, landform, landscape position, size of the treatment, weather, and posttreatment grazing management. Thus, this design integrates data on both local and landscape factors to design and test the practice. The need for local stratification, randomization, and measurements with respect to the specific treatments indicates the need for a carefully designed monitoring and spatial information system. The results of the Restore New Mexico monitoring will be used to update STMs for the area.

We anticipate that the model-based approach will produce results and benefits that have heretofore not been achieved. In spite of a long history of brush management activities, the lack of an ecological site and STM-guided experimentation and monitoring has circumvented a quantitative understanding of the conditions under which brush management succeeds and the characteristics of success. We anticipate that the Restore New Mexico monitoring will be able to more precisely target brush management activities to achieve desired results in the future. Model-based conservation planning might save millions of dollars that would be spent on ineffective treatments in southern New Mexico alone.

RECOMMENDATIONS

Incorporate Landscape Perspectives into Conservation Planning

How can agencies and conservation planners implement model-based approaches such as those discussed above? First, it is unreasonable to assume that all conservation planners will be able to access, manipulate, and store spatial data and process models underpinning the model-based approach. Expertise in GIS, remote sensing, and model implementation needs to be available to planners given a general knowledge of the uses of these tools. Such support within NRCS could occur initially via training and collaborations with other action and science agencies and with academic partners to make spatial data available in the planning process. The production of training materials and simple Web-based tools should be a priority. There should also be institutional support within NRCS to make expertise in GIS, remote sensing, and model implementation available to planners. Such support within NRCS could occur via spatial data specialist positions; similar expertise already exists in support of soil surveys.

Second, agencies alongside academic programs at universities should invest in longer-term training in landscape ecology and related tools and concepts, particularly GIS, hydrology, and soils/geomorphology. Most programs already emphasize elements of this training, but these elements are seldom integrated with ideas including ecological sites, STMs, monitoring, and approaches to specific rangeland practices. There is a clear need to develop integrative courses and texts that link the more disciplinarily specialized bodies of knowledge.

Third, administrative changes are needed in the development of conservation plans to include systematic consideration of off-site effects. Modifications to the Conservation Practice Physical Effects planning document to add an “off-site effect” category would be one approach. Historically, conservation planners incorporated landscape perspectives via rules of thumb such as “look across the fence to see what is coming at you,” but administrative requirements would ensure that spatial interactions are taken into account when needed. Finally, there must be an institutional structure within which STMs can be updated. This is a critical step if we are to learn from the study of conservation effects.

Develop Landscape Approaches to Conservation Effects Monitoring

Structured monitoring should be part of the budget for broad-scale conservation practices. Clear guidelines for design, institutional support for the implementation of the design, and a mechanism to incorporate what is learned from the monitoring within ESDs should be developed (as illustrated above). To accomplish this, planning data entered by field staff need to be integrated with spatial data and followed up by monitoring in treated areas as well as off-site areas. Careful monitoring design, including stratification and sufficient replication, must be supported if the monitoring is to be useful.

There should be a system in place to document the cumulative benefits of practices with regard to large areas, long timeframes, and a broader range of services (Tanaka et al. 2005). Although there have been few attempts to project cumulative benefits of widely dispersed, small-scale restoration projects, Kondolf et al. (2008) developed a relatively simple approach to prioritizing projects to achieve maximum off-site benefits. This approach allows for an explicit statement of the assumptions about the delivery of ecosystem services offsite via the application of traditional site-specific management practices. Similar approaches merit closer consideration by NRCS.

Develop a Spatial Information Support System and Associated Tools

A spatial information system designed to support both the planning process and the design of monitoring programs requires a spatial database of maps and tools that can be used by planners, perhaps with assistance by spatial data specialists. The Web Soil Survey (http://websoilsurvey.nrcs.usda.gov) already provides a remarkable array of data and tools that can be used in the planning process and mirrors some of the steps described above. Soil Web, another recent online soil survey tool, links soil maps to Google Earth imagery (http://casoilresource.lawr.ucdavis.edu/drupal/node/902). It should be possible to expand Web Soil Survey and Soil Web by linking their data layers to others via Web-based distributed networks (e.g., climate from PRISM, http://www.prism.oregonstate.edu, or fire locations from GeoMAC, http://www.geomac.gov) and to project-level data housed in local servers.

The Landscape Toolbox (http://landscapetoolbox.org) provides an example of how various tools can be linked to specific

A restoration treatment using woody debris to create vegetated bands (patches) in eroded soils of Big Bend National Park, Texas. (Photo: Brandon Bestelmeyer)

Recently (left) and less recently (right) burned tallgrass prairie in east-central Kansas. (Photo: Brandon Bestelmeyer)

natural resource problems. The Landscape Toolbox employs an analytical framework to link specific problems to tools that provide information at different spatial scales. Projects such as the Landscape Toolbox highlight the value of linking disparate information sources and reveal the need for standards of information transfer (e.g., data formats and metadata) in the use of spatial data from multiple sources.

Maps that Facilitate Evaluation of Landscape Context. Such maps can be based on stable physical attributes derived from digital elevation models, including drainage networks, flow directions, and flow accumulation developed using GIS-based models. In addition to these more static attributes, climate data could be used to evaluate patterns of wind direction and velocity and fine-scale patterns of precipitation (e.g., via Doppler radar maps). Fire extents and characteristics can also be mapped. Readily available, preprocessed spectral data from the MODIS satellite can be used to document variations in production at landscape scales (e.g., maps of the Normalized Difference Vegetation Index with 250-m resolution). These data, in conjunction with visual interpretation of land surface characteristics in aerial imagery, can allow a trained specialist to recognize several important physical spatial processes.

Simple Spatial Models. Expanding upon the maps developed above, hydrological, fire or other models can be used to simulate landscape processes using digital elevation models, soil maps, vegetation maps, and other data as inputs (e.g., Stratton 2006). Such models could be used to target, for example, particular portions of a watershed for treatment based on the sensitivity of water output to a change in plant cover expected in an area (Wu et al. 2001). More complex process-based models are being developed, but they are often designed for the purpose of exploring the effects of specific processes and it is unlikely that these models will be useful for guiding management anytime soon. On the other hand, simple models with user-friendly interfaces could enable the planners to develop alternative spatial designs for practices and compare the anticipated effectiveness of these options.

Spatial Pattern Indicators. Such indicators would be used to evaluate differences in patch or landscape pattern (e.g., patch size, patch density, connectivity, landscape leakiness) that are important mediators of ecological processes including water redistribution, erosion, or wildlife movement. Calculation of these indicators usually relies on classified satellite data or aerial imagery and the type of classification used depends on the process in question. Consideration of spatial pattern need not involve GIS-based calculations, however. Simple ground-based indicators for field inventory are available for upland (Tongway and Hindley 2004; Herrick et al. 2005; http://jornada.nmsu.edu/sites/default/files/FieldGuidePedodermPattern.pdf) and riparian areas (Prichard 1998). The specific indicators required will vary with the ecological process involved. As with vegetation composition, the measurements will change in response to practices and thus will be useful for monitoring. Assistance from spatial data specialists should be made available to support remote-sensing based monitoring. Protocols for the selection and use of specific indicators could be specified for MLRAs or groups of MLRAs that share similar ecological processes. Research support will be needed to identify useful spatial pattern indicators and to interpret their values. In this vein, reference values should be associated with descriptions of state and plant community phases.

Link Results to a National Ecological Site Database

Although the spatial information support system discussed above facilitates conservation planning, we must also consider where to house the monitoring data and the lessons learned from them. Similar to existing databases such as the National Soil Information System and the Ecological Site Information System, raw monitoring data at points and the interpretations derived from those data, respectively, will likely require separate, but linked databases. Current revisions to database structures planned with the reorganization of responsibilities for production of ESDs within NRCS provide an opportunity to consider how monitoring data on conservation effects could be linked to ESD interpretations. In any case, a database system and process to link CEAP monitoring data to ESDs must be a high priority within NRCS.

Support Research to Better Integrate Concepts, Tools, and Applications

Development of a systematic approach to conservation planning at the landscape level would benefit from research that addresses how to integrate information from landscape scales, spatial patterns, models, conservation planning field data, and NRI and other monitoring data. Specifically, such research would illustrate how field measurements should be gathered so that they can be scaled up or integrated with models and spatial data from broader spatial scales. Case studies centered on specific landscapes or MLRAs, supported by the USDA National Institute of Food and Agriculture and USDA Conservation Innovation Grants, could be used to explore how to bring together the variety of tools and approaches. Research is also needed to determine how best to scale up interpretations of conservation effects to state, regional, and national levels. Case studies illustrating how real-world conservation planning is linked to landscape research could provide an effective assessment of the benefits of the landscape perspective.

KNOWLEDGE GAPS

We identify the following knowledge and administrative gaps that need to be overcome in order move forward with our recommendations.

1. ESDs and STMs need refinement and elaboration so that they contain explicit predictions about how plant communities and dynamic soil properties are assumed

to change as a function of conservation practices. For predictions involving spatial interactions, levels that aggregate multiple ecological sites (such as SGSs) will need to be specified and carry the predictions. These predictions become the hypotheses for monitoring efforts and tests of them are used to update ESDs and refine our use of practices.

2. The lack of synthetic models at the level of MLRAs or LRUs is a clear limitation to developing consistent ESDs and STMs across the United States. Such models are needed to develop comparisons among different ecological sites and regions (e.g., the comparative likelihood of success or the magnitude of an effect in different land areas) and to represent spatial interactions at landscape scales (e.g., wildlife populations that cross multiple ecological sites).
3. Readily available maps of ecological sites, and especially ecological states, are generally not available to assist planning. Without maps that are connected to ESDs and STMs, planners will find it difficult to use these tools.
4. Models for spatial interactions in landscapes that specify how conservation practices in one state/ecological site mapping unit should affect the states of adjacent mapping units are poorly developed. This will require creative research and modeling coupled to field studies.
5. Spatial pattern indicators that aid in predicting the trajectories that states will take under different conservation practices are seldom available when they could be useful. Addressing this gap will require the integration of pattern analysis, using field- or image-based approaches, coupled to monitoring of conservation effects.
6. We lack administrative and database mechanisms to update ESDs and STMs. If the ESDs and STMs are not improved as a function of the monitoring tests, then learning cannot occur and the efficiency of conservation practices will not improve.

CONCLUSIONS

Our assessment indicates that landscape perspectives and applications are needed to promote long-term success and effectiveness of conservation practices on rangelands. A large body of literature supports the utility of a landscape perspective (e.g., Naveh and Lieberman 1984; Turner et al. 2001). We reviewed the implications of spatial heterogeneity, spatial pattern, and spatial scaling for the design of practices and interpretation of conservation effects. Spatial heterogeneity is used to understand why a practice succeeds or fails in areas of differing climate, soil, and spatial context. Spatial heterogeneity can also be a primary goal of practices, for example, by supporting the varying habitat elements used by animal species. Spatial patterns are used to indicate critical processes that are not reflected in other measures, such as connectivity for wildlife movement or runoff and erosion potential. Patterns too can be a conservation objective (e.g., wildlife corridors or areas of low landscape leakiness). Spatial scaling is used to understand the dimensions of the land area over which spatial interactions link practices in one place to effects in other places, and conversely, how characteristics of the landscape affect the local success of a practice.

Landscape perspectives encompassing spatial heterogeneity, pattern, and scale are increasingly being connected to practical tools that can be used by conservation planners. Such tools include indicators, classifications and maps of ecological sites and states, and hydrologic models. These tools can be used both to help design practices and to design the monitoring programs that evaluate their effects. A spatial hierarchy focuses attention on the data needed at each spatial scale governing ecological processes of interest. In order of decreasing scale, MRLAs, SGSs, watersheds, ecological sites, plant communities, and patches each relate to processes governing the management of rangelands. Furthermore, consideration of societal information such as land ownership is usually needed at broad scales. Each of these data sources can be consulted in a systematic way, which we described in six steps, to design and evaluate conservation practices in a landscape.

We recommend that conservation practitioners consider several scales of spatial pattern and related spatial processes, including cumulative effects, each *time a practice is applied*. This synthesis indicates that a systematic approach to planning that incorporates landscape perspectives would, in many cases, lead to more effective interventions by 1) recognizing indicators foretelling the likelihood of success; 2) targeting interventions to ecological sites, states, and spatial contexts in which success is most likely; and 3) maximizing (and measuring) the cumulative, positive effects of practices over the long term at broad spatial scales.

Although some of the tools and approaches supporting landscape perspectives are already used by conservation planners, the development of others will require a scientific and institutional investment by the federal government and support by universities and funding agencies. Spatial data information systems should be developed that link maps, models, and pattern-based metrics to support planning and monitoring design. Databases are needed to house the resulting data. The interpretations of these data should be linked to ESDs. Foremost, we must invest in training and research to instill an understanding of the concepts and a capacity for reasoning about landscape processes (e.g., Gergel and Turner 2001). We suspect that such investments would pay for themselves, and then some, by improving conservation effectiveness in the millions of acres of rangelands that will be treated in years to come.

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