People and rangeland biodiversity — North America

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Introduction

Biological diversity refers to life at all levels of organization, from genes within populations to the global arrays of species (Wilson 1992). It is often assumed, though, that any discussion of biodiversity is focused at the species level. However, this discussion is still highly complex given that we know only a small percentage of the total microbial, plant, and animal species. Though estimates of microbial, plant, and animal species are as imprecise as between 5 and 50 million (Tilman 1999), there are many arguments for conserving biodiversity. For rangeland management, the concept of biological diversity is typically focused on plant species richness, evenness, and heterogeneity at community-level spatial scales (West 1993). Species diversity is clearly a major determinant of many ecological processes, especially those related to resilience and resistance (Tilman 1998). Given that over 75% of the earth's ecosystems are manipulated for human purposes (Moguel & Toledo 1999), maintaining abilities of these systems to buffer (resistence) and recover from disturbances (resilience) is key to conserving inherent ecological functions (Peterson *et al.* 1998). For example, one consequence of species loss is a limit to the potential ways an ecosystem can reorganize following disturbance, an important component of ecosystem resilience (Peterson *et al.* 1998). It is well recognized that we are currently in the midst of the sixth major period of extinction on earth and that this current extinction period is biologically driven (Chapin *et al.* 1998). The potential for significant impacts on ecosystem functions due to losses of biodiversity is large and immediate.

Despite the acknowledged biological, social, and economical importance of biodiversity, we lack clarity on many aspects of this concept (Ricklefs 1987; Chapin *et al.* 1998; Gustafson 1998; Tilman 1998). We have limited understanding of the interactions between biodiversity and ecosystem processes and the relationships between biodiversity and ecosystem functions across multiple spatial and temporal scales (Callicott *et al.* 1999). Furthermore, application of these aspects of biological diversity to land management has proven difficult. Management decisions are often related to spatial and temporal scale relationships not yet understood (West 1993). Because of the large numbers of plant and animal species as well as other taxonomic orders such as microbes that may be found in ecosystems, it is biologically intractable and economically infeasible to manage for more than a fraction of existing diversity on a species basis (Franklin 1993). Over 500 species and subspecies of native plants and animals have become extinct in North America over the past 400 years, and further loss of species is inevitable (Scott *et al.* 1987). However, management for taxonomic richness at the local community (alpha) and the landscape (beta) scales is an important goal for conservation of ecosystem function of rangelands (Hobbs & Huenneke 1992). For North America, we need to determine how to conserve biological diversity as our rangelands are increasingly fragmented and our management units are often at relatively small scales. Presently, we are lacking coherent scientific principles that link our management actions to the maintenance of landscape level properties and that reflect how a landscape context affects our management units and decisions. The goal of this paper is to outline emerging strategies for managing North American rangelands based on a recognition of landscape connections between human activities and conservation of biological diversity.

Current setting

Unifying theories that link biodiversity and rangeland management across spatial scales are not currently available, yet are critical if we are to conserve biodiversity and ecosystem function. As a result, adaptive management strategies to conserve biodiversity lack clarity of resolve and evidence supporting their effectiveness. There are, however, a number of relevant points about biodiversity of rangelands that are supported by scientific evidence: (a) biodiversity is scale (spatial and temporal) dependent, (b) the principal factors which structure plant communities differentially impact biodiversity, (c) arid and semi-arid rangeland environments support many species at the edge of their tolerance limits, and (d) rangeland management-related impacts may be the least significant of the human caused impacts on biodiversity. Each of these points will be discussed in further detail.

Biological mechanisms

Any discussion of rangelands, their management, and the importance of biodiversity needs to recognize the episodic and catastrophic characteristics of arid and semi-arid environments. Spatially, North American rangelands are a heterogeneous matrix of semi-natural communities, which may be properly or improperly grazed by livestock and with fragmented ownership. Temporally, this land type is best described as a pulse–trigger–reserve environment (Ludwig & Tongway 1997). The patchy nature of this environment creates edges, interiors, and highly ephemeral conditions. In combination with highly variable abiotic conditions, many species operate at extreme limits to their tolerances for germination, establishment, and persistence.

In general, biodiversity within a landscape is a function of two very different but interactive patterns: environmental gradients related to limiting factors; and processes associated with sites recovering from natural and human-induced disturbances (Romme 1982). We do not have well articulated theories which adequately explain these various scales dependent functions, their interactions, and their influences on biodiversity.

Four mechanisms, all dependent upon spatial scales, have been related to maintenance of species diversity. These are niche relations at community scales (< 100 ha), habitat diversity and mass effects at landscape scales (100–10^4 ha), and ecological equivalency at regional scales (>10^4ha) (Shmida & Wilson 1985). 'Niche relations' refers to interactions among species as well as interactions between species and their environment that can generate patterns of species coexistence, for example by resource partitioning. 'Mass effects' refers to the occurrence of species outside their core habitats, a characteristic of pulse-reserve environments which can include numerous microsites created by ephemeral, episodic events. Habitat diversity refers to heterogeneity in habitat conditions, and ecological equivalency refers to co-presence of species with identical habitat requirements. Species richness in a region is related to all four factors. However, habitat diversity has been regarded as the most important at community and landscape scales.

Environmental factors that affect biodiversity are also scale-dependent. Pressures which act to decrease biodiversity, such as predation, disease, drought, and disturbance, frequently occur at local (community and landscape) scales (Ricklefs 1987). High local biodiversity is often controlled by the temporal scale of disturbances (or the time since a site was disturbed) and localised voids or biological limits on species recruitment (Tilman 1999). Pressures which operate to increase species diversity, such as pervasive climatic events, speciation, migration, and intrinsic productivity, operate from local to regional scales. Management actions will more typically affect factors which decrease species diversity, such as fragmentation of habitat, facilitation or restriction of seed dispersal, and modifications of vegetation structure. In addition, management actions will more characteristically operate at smaller spatial scales and at relatively shorter time spans than other biological drivers, such as drought or predation.

Structural features of arid and semi-arid rangelands are strongly shaped by four major drivers: grazing, fire, competition, and other disturbances including drought (Belsky 1992). It is very difficult to distinguish between direct effects of these drivers on biodiversity and their indirect effects on the environment (Huenneke & Noble 1996). It is also extremely difficult to separate out effects of individual drivers. None of the effects of these disturbances, including grazing and fire, upon diversity are easily predictable (Chaneton & Facelli 1991). Disturbance effects are scale dependent. Responses in diversity to disturbance are strongly shaped by initial conditions prior to disturbance, the history of disturbances for a particular site or matrix of sites, and the life history traits of plants available to respond to disturbances (Pickett & White 1985). The stability of rangeland ecosystems resides in maintenance of resistence and resilience to change in the environment (Johnson & Mayeux 1992). Species flux rates, at local spatial and temporal scales, can be high, and these systems are constantly adjusting and adapting to disturbances and invasions by exotic species that act to either increase or decrease biodiversity (Hobbs & Huenneke 1992).

Human activities

Within this dynamic environment humans' activities are of six general types: (i) introduction and management of grazing animals (livestock, non-native game species, feral animals), (ii) rangeland improvements, including development of water sources for animals, (iii) introduction and management of non-native plants, (iv) removal of competing animal (predators, other herbivores) and plant (fuelwood) species, (v) implementation of agricultural practices (especially irrigated agriculture), and (vi) fragmentation by urbanization (housing developments, road construction, recreational activities) (Huenneke & Noble 1996). Much of the legacy of rangelands in North America is a product of the first four of these activities. The general impacts on biodiversity of livestock overgrazing, establishment of monocultures of forage species, introduction of exotics, and removal of predators are either well documented or intuitively obvious (Smith 1899; Buffington & Herbel 1965; Hastings & Turner 1965; McNaughton 1993). Specific effects of a particular impact upon biodiversity within a particular region are often debated. However, the collective impacts of these activities over the past century on the structural and functional characteristics of North American rangelands have been substantial (Huenneke & Noble 1996). In the United States, legislation has been enacted over the past three decades in response to these impacts, real and perceived, on a number of attributes of rangelands, including biodiversity.

Today, the potential impacts of properly applied rangeland management practices (activities (i) and (ii)) on biodiversity are much less adverse than in prior decades. Classic practices of grazing management and rangeland improvement are generally directed under institutionalized constraints. For example, in many regions of the US stocking rates are well below historical peaks of the early 20th century. In addition, carrying capacities that reflect a recognition of ecological limits have been widely established for many areas during the last half of the 20th century. There has been a recognized improvement in rangeland conditions over the last sixty years, though a corresponding positive impact on biodiversity is not known. Unfortunately, many areas have remained in degraded states due to prior mismanagement, and are impervious to management practices designed for rangeland improvement (USDI, BLM 1997).

Two of the other human activities, (iii) and (iv), with impacts on biodiversity are also subjected to much tighter controls or restrictions regulated by public opinion or legal constraints. The introduction of non-native species and the target removal of native species, especially predators, are rarer activities than in the past. Most activities of management entities today work within our increasingly predominant philosophy of maintaining native plant and animal species. In addition, government agencies often work diligently to reintroduced extirpated species to selected environments.

The two activities which will have the most impact on rangeland biodiversity are agriculture and urbanization. Agricultural impacts, including water use, land conversions, and chemical usage, are well documented. These impacts will continue in the future, but their extent is extremely difficult to predict.

Urbanization may be the greatest present threat with unknown consequences to rangeland biodiversity. Impacts occur in two different fashions. First, urbanization further fragments an already heterogeneous environment. There are many examples where fragmentation of habitat or restrictions to geographic ranges results in lower species density or overall reduction in diversity (Rosenzweig 1995). The rate at which land in North America is being converted to urban uses has been well documented (Sorenson *et al.* 1997). Even in relatively remote regions, the value of land for urban development can be 4–100 times its value as grazing land for livestock. The economics of this conversion are perverse, and the transition of rangelands from predominately grazing land or even multiple use functions to urban or suburban functions will continue. For example, the United States has been categorized into 187 major land resource areas based on aggregations of nearly homogeneous areas of land use, elevation, topography, climate, water resources, potential natural vegetation, and soils. Sorenson *et al.* (1997) determined that the farm and ranch land within 127 (68%) of these major land resource areas are significantly threatened by land development. Though it is widely recognized that we are currently dealing with fragmented and semi-natural habitats, the grain of fragmentation will continue to be reduced by urbanization with significant negative impacts on biodiversity.

The second manner of impact is the compounding effect of urbanization beyond spatial fragmentation. Effects associated with human habitation cascade through these systems. Small scale disturbances can have compounding negative impacts well beyond their individual areal effects. Examples include roads, introduction of domestic pets, landscaping with exotic species, and use of pesticides and herbicides. Some of these impacts can be 15 to 20 times their original dimensions (Foreman & Alexander 1998).

Emerging strategies

From a range management standpoint, we need to recognize two overriding factors which govern our capacities for actions in response to the current situation. First, we have two primary management options: we can manipulate vegetation structure in direct and indirect ways; and/or we can affect plant and animal production by adjusting our controls over livestock (Stafford Smith 1996). Often, any manipulations of vegetation in North America are expensive, subject to regulatory controls, of limited biological effectiveness, or can have unwanted secondary effects. It is also common that our scale of management actions does not correspond to the scale of landscape functions.

Fortunately, our knowledge of the effects of livestock grazing management is applicable and well-supported. We have a general understanding of the importance of controlling timing, intensity, and frequency of grazing (Trlica & Rittenhouse 1993; Holechek *et al.* 1998). Though the relative effects of various management combinations are debated for specific rangeland systems, it is generally recognized that managed grazing can have neutral, positive, or negative effects on biodiversity depending upon the environment (Milchunas *et al.* 1998). For most environments, livestock grazing has negative impacts on species diversity under poor management or excessive use (Miller *et al.* 1994; Laycock 1994). Our primary problems related to livestock grazing are the same ones we have continually dealt with during the 20th century: (a) coping with variations (spatial and temporal) in forage production, (b) manipulating an animal behavioural process (grazing) that is plant species specific, and (c) managing grazing across landscapes with limited (if any) measurements to monitor or assess impacts. Effectively managing these sources of variation should greatly limit negative effects of livestock grazing on biodiversity.

Second, we need to recognize that non-equilibrium systems, which can behave for short periods in a relatively stable state, are weakly responsive to management controls. The driving forces in these grazed systems are typically interactions between management practices and environmental stresses, especially drought (Tainton *et al.* 1996). Inherent environmental variability of North American rangelands results in unpredictable forage supplies. For example, the most significant impacts of grazing occur during drought periods when available forage is low and overgrazing is a chronic factor. These two factors (management actions and non-equilibrium systems) need to be recognized and acknowledged before any management strategy related to biodiversity can be constructed.

A general approach

There are few effective and affordable tools for management, and any intentional, planned impacts on species diversity will be indirectly achieved, at best. One approach would be to manage rangelands to maintain spatially heterogeneous habitats that will support a wide variety of species. Biodiversity has been related to patchiness of the landscape, and patches often are viewed as a consequence of disturbances and processes related to species renewal (Longlands & Young 1995). However, there are not many guidelines for deciding at what scale and for what purpose habitat heterogeneity should be maintained. Given the spatial scale at which management actions can be economically applied on most North Americans rangelands, it should be recognized that manipulations targeted at relatively small landscape scales (<100 ha) will be the most realistic and achievable (Herrick *et al.* 1997). Because one of the greatest threats of land degradation is the loss of local biodiversity (Parmenter *et al.* 1995), implementing small scale actions which are directed to remediate or maintain site capacities for resilience and resistence to degradation should be the goal. With this strategy, alpha diversity can be maintained at small spatial scales, and beta diversity can be maintained by a mosaic of small patches distributed across the landscape being managed.

Operationally, management could be structured upon a spatial scale that can be observed and manipulated. Observations can be based on relatively simple kinds of measurements for populations within a community, such as species richness and evenness, or more elaborate measures related to watershed function, soil stability, and plant demography. Relative to species diversity, populations can be evaluated by specified thresholds of extinction or explosion (Pyke 1995).

High local diversity is created both by the occurrence of small, patchy disturbances and by patchy species distributions (Tilman 1999). Intermediate disturbances which occur at frequent intervals create a mosaic of sites at differing stages of recovery that can directly increase local spatial heterogeneity. Recruitment limitations create local absences or voids in particular species. However, human-induced disturbances, such as chronic overgrazing, can effectively counteract these factors to reduce spatial heterogeneity (Milchunas *et al.* 1998). Overgrazing can reduce the frequency of other natural disturbances, such as fire, and can be a dispersal process which removes recruitment limitations for certain species.

These perceptions concerning biodiversity are emerging in spite of lack of a theoretical structure for the application of landscape ecology to the management of heterogeneous landscapes (Wiens 1995). Despite this knowledge limitation, the first principle must be to structure our management at workable spatial scales within the landscape. Management can then be directed towards two options: generating heterogeneity of small sites or patches within a landscape, or altering the external forces which further create heavily fragmented landscapes (Saunders *et al.* 1991). Many heavily fragmented landscapes in North America reflect a myriad of external interacting forces including biological, cultural, sociological, political, and economical factors. Frequently they cannot be managed by an individual and require partnerships among individuals within an area. In North America, it is becoming increasingly common for individuals within a region to join forces in an effort to manage fragmented landscapes, or landscapes under threats from external forces. Saunders *et al.* (1991) stated that integrated management requires neighbours to interact. Stafford Smith (1996) argued for a new management paradigm based on participation within the local community. Generically, these community-based approaches are supported because problems facing individual land managers are ill-defined, clear courses of actions are often lacking, and there are few, if any, feedback systems in place to gauge effects of management decisions. These criteria certainly apply to the problem of conserving biodiversity on rangelands.

Local examples

There are two recent examples of a participatory paradigm being employed in the southwestern United States for resource management. The Malpai Borderlands Group is an association of ranchers supported by government agencies, community organizations, scientists, and the general public. The goal of this group is to restore and maintain the natural processes that create and protect a healthy, unfragmented landscape to support a diverse, flourishing community of human, plant, and animal life in the Borderlands Region (southwestern New Mexico and southeastern Arizona). A major threat to this region is urbanization of ranch lands causing continued fragmentation throughout the region. Participation with this group provides access to numerous resources, including forage that would not normally be available to an individual. Participation requires assurances that an individual will not subdivide their ranch land. Collectively the group is able to pool enormous resources which are synergistically greater than the sum of their parts, and practice adaptive management. This community based approach, primarily in response to externally driven forces, has been described as creation of a 'radical center' (Brown & McDonald 1995).

The Catron County Citizens Group in western New Mexico is another community organization that is defining a participatory paradigm for resource management in North America. The mission of this group is to serve as a forum for honest and open debate over issues in the community and to take action on projects that ensure an economic, social, and environmentally sound future. Land management debates in Catron County have been as contentious as almost anywhere in the US over the past decade. This Citizens Group, however, has painstakingly worked to identify projects within the county that can result in improved resource conditions. The Catron County Group has identified very specific projects that will both maintain traditional extractive industries in the community and sustain species diversity, especially threatened and endangered species. For example, one project has a goal of protecting endangered species that occur in local riparian zones while managing livestock grazing in the adjacent uplands of the watershed.

In contrast to the Malpai Borderlands Group, the Catron County Group is primarily taking a small site approach and involving the community in managing remnants. Both approaches, though similar as participatory activities, are quite dissimilar in addressing forces that are impacting resource heterogeneity and landscape fragmentation. The Catron County Group is trying to address driving forces impacting rangelands within their community. In this fashion they are attempting to control project goals and demonstrate site specific achievements. The Malpai Borderlands Group has recognized that a major driving force impacting their community is external — the demand to develop less expensive land to support expansion of urban populations in the south-western US.

It is important to recognize that both groups are employing available information in identifying actions in support of their overall objectives. Each group interacts with various scientific and bureaucratic organizations. The Malpai Group has a science advisory committee, and the Catron County Group conducts workshops using subject matter experts as participants in addressing specific project areas. However, it is also important to recognize that both groups are addressing issues that are realistically outside current scientific theories. At local levels, for different reasons, each of these groups is trying to effectively manage lands at landscape, even regional scales. They are using adaptive management principles by basing practices on current ecological concepts and adjusting management given subsequent responses. For example, the Malpai Group, which is further along in this process, is now providing much of the new information on the role of prescribed fire at landscape scales in the northern Chihuahuan Desert.

In attempting to manage these diverse landscapes that contain a heterogeneity of private and public lands, rangeland conditions, histories of use, and commodity values, these groups are addressing conservation of biodiversity from a functional perspective. Their primary needs are not the descriptive perspectives currently characteristic of landscape ecological theory, but rather consist of conceptual frameworks that link management of community scale heterogeneity with their sustained use of the landscape. Management requires extrapolations across scales (Turner & Gardner 1991), and managers need a basis for these extrapolations.

Unfortunately, in North America the scientific community is not yet able to service this need for an effective conceptual framework for managing beta scale heterogeneity while maintaining or improving resistence and resilience of landscapes. Fortunately, the scientific community can contribute information on management strategies for community scale practices that are conceptually well-based. Currently, this conceptual framework is based on a state-and-transition perspective, though other conceptual frameworks are still employed. At this relatively small local scale, there are increasing numbers of examples of conservation of biological diversity of managed landscapes. Application of scientific theory at this small scale has occurred for decades and will continue for both individuals and larger community-based groups such as the Malpai Borderlands and the Catron County Groups.

Implementation of a participatory paradigm by more communities will become increasingly common in order to exert local controls on landscape fragmentation. There are several general questions that need to be addressed in order to contribute to the management of rangelands in support of communities in North America:

1. what are the relationships between beta diversity and landscape resilience and resistence?
2. what are the appropriate scales for assessing and monitoring landscapes?
3. what are the state and transition models that have application to rangeland management in North America?
4. which alpha scale processes can be managed (manipulated) that effect larger scale processes and impact the maintenance of goods and services?

Conclusions

The United States Endangered Species Act is probably the strongest law affecting the conservation of biodiversity in North America. Yet, this legislation, currently being evaluated for revision, only affects species listed as threatened or endangered and their habitats. Conservation of biodiversity is legally constrained to the relatively narrow species limits of this law.

A major threat to rangeland biodiversity in North America is the increasing fragmentation of local landscapes. Though there are instances of continued rangeland degradation due to traditional disturbances of overgrazing and general resource mismanagement, these are relatively rare. The expansion of introduced species and the increasing value of rangeland for urban development represent the more intrusive and pervasive disturbances at the end of this millennium. These disturbance forces operate at multiple spatial and temporal scales. Rangeland management practices to counteract these forces are currently applicable only at small spatial and short temporal scales.

Local communities have begun to try to control fragmentation of local landscapes despite a deficit of supporting conceptual frameworks for directing their actions. In an effort to sustain desired goods and services from the local landscape, these community groups are initiating small scale projects which maintain heterogeneity and conserve species diversity. This scale of management actions can be generally supported by existing and emerging scientific information. In addition, this scale of management actions will continue to be appropriate for ecological, political, social, economical, and cultural reasons. The scientific community needs to work to provide information on linkages between these small spatial scale management actions and larger landscape scale attributes of importance to these communities. These attributes will primarily be the maintenance of goods and services from these landscape. Addressing scientific issues related to landscape resilience and resistence will directly support the conservation of biodiversity on North American rangelands. The strongest evidence in support of this conclusion is the increasing occurrence of local communities developing mechanisms to control their landscapes in spite of acrimonious and contentious debate over rangeland resources in North America. The opportunity exists to both advance scientific theory and practically apply that theory in support of local rangeland management.

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