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,