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The Trail Boss

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- to properly take care of the basic rangeland resources of soil, plants and water;
- to develop an understanding of range ecosystems and of the principles applicable to the management of range resources;
- to assist all who work with range resources to keep abreast of new findings and techniques in the science and art of range management;
- to improve the effectiveness of range management to obtain from range resources the products and values necessary for man's welfare;
- to create a public appreciation of the economic and social benefits to be obtained from the range environment;
- to promote professional development of its members.

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Invited Synthesis Paper



NEIL E. WEST has offered his overview of how the issues of biodiversity may influence planning, research, and management of rangelands.

Neil grew up on a cattle and sheep ranch in southern Oregon. He went to Oregon State University with the intention of becoming a chemist. His career was irrevocably changed, however, when he learned, in an introductory zoology class, about the outdoor orientation of the then esoteric science of ecology. He attained a B.S. in general science and continued directly into a Ph.D. program in plant ecology at OSU. His doctoral work focused on the effects of fire exclusion on ponderosa pine forests in central Oregon. After a short assignment at the

Oregon Forest Research Lab in Corvallis, West decided to relocate to the drier terrain of Utah. West has been at Utah State University, Logan, most of the time since 1964. L.A. Stoddart and C.W. Cook were important early mentors. He has tried to overcome his western biases by spending a semester or more at the University of Georgia; Yale University; Adelaide University, South Australia; and Hebrew University, Jerusalem, Israel.

West has chosen the difficult and often unappreciated professional task of attempting to keep up with the output of theoretical synecologists and translating the applicable portions to the improvement of rangeland management. His current efforts focus on the scientific underpinnings of sustainability, desertification, monitoring of trend and interpretation of rangeland condition. Unless his wife Alexa can dissuade him otherwise, Neil is usually to be found each weekend sampling the aquatic and upland ecosystems of the Intermountain regions behind a gun or fishing rod.

Biodiversity of rangelands

NEIL E. WEST

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Abstract

Biodiversity is a multifaceted phenomenon involving the variety of organisms present, the genetic differences among them, and the communities, ecosystems, and landscape patterns in which they occur. Society will increasingly value biodiversity and influence the passage of laws and writing of regulations involving biodiversity which rangeland managers will have to abide by over the coming decades. Even private and developing world rangelands will be affected.

While taxonomic knowledge of vertebrates and vascular plants and their abundance, rarity, and distribution, in the developed nations is generally adequate, the same cannot be said of the developing world. Furthermore, adequate knowledge of invertebrates, nonvascular plants, and microbes is deficient everywhere.

Although the basis of variation at all higher levels, genetic variation within rangeland species, even the major ones, has barely been assessed. Obtaining statistically adequate data on populations of rare species that are small and secretive is well nigh impossible.

We have many means of measuring community diversity, but all of them are value laden. That is, choice of variables to measure and how they are indexed betrays what we consider are important. We should be more forthright in stating to the users the biases of these methods. There are many other, more useful ways to describe community-level diversity besides the traditional focus on species.

Ungulate grazing is an important process in many ecosystems. Thus, removal of grazing destabilizes some systems. Livestock grazing will actually increase the chances of survival of some species. Moderate livestock grazing can also enhance community and landscape-level diversity in many instances.

Attention is now shifting from "charismatic" species to defensively managing larger tracts of land with habitat or ecosystems holding suites of sensitive species. Since some accelerated extinction of isolated populations and species is inevitable, we need to know which species and ecotypes are most valuable. Understanding of modular, guild, and functional group structure would also help us identify keystone or critical link species and better focus our attention on truly important tracts of land where they live.

It is probably more important to sustain soils and ecosystem processes than any randomly selected species, especially if functionally redundant species can be identified. Similarly, not all introduced, alien, or exotic species are equal threats; it depends on how they fit into ecosystems. Sustainable development will depend on finding balance between use and protection, from range sites to landscapes, and even on a global basis.

Key Words: extinction, invasion, keystone species, functional groups, guilds, critical link species, sustainability, fragmentation, National Forest Management Act, Surface Mining Control and Reclamation Act

The threat of global environmental change, accelerated species extinctions, and changing societal values has caused biodiversity to become a topic that has captured the attention of the public as well as the scientific community (Anon. 1991a). Laws regulating biodiversity have been passed and more legislation is being proposed. Even if no new laws are passed, it appears that further expansion of regulations and enforcement will have considerable effects on how rangelands will be managed in coming decades. Although the alteration of management because of concerns for biodiversity will occur first on federal lands in the western U.S., the management of private rangelands will also probably be increasingly influenced by regulations involving biodiversity. Indeed, even the style of assistance in developing nations is being changed by concerns for biodiversity from the World Bank and the governments of donor or assisting nations (McNeely et al. 1989). Thus, all those concerned with rangeland management worldwide will eventually become influenced by this issue. We can either wait for the influences to become evident and react after regulations are in place or we can become proactive. If rangeland management professionals are to have any input to the writing of laws and regulations or redirection of monitoring and research programs, they must understand what biodiversity is, why it is valued, and how its monitoring and regulation will influence future rangeland policies and management practices.

Biodiversity Defined

The broadest concept of diversity is that of variety. Although environmental variety includes physical, chemical, topographic, edaphic (soils), and meteorological (climatic) factors, mankind is usually most interested in the variety of other organisms. In simplest terms, biological diversity (hereafter shortened to biodiversity), is the variety of life and its processes, including the variety of living organisms, the genetic differences among them, the communities, ecosystems, and landscapes in which they occur, plus the interactions of these components (Fig. 1). Some (Salwasser 1991, Gomez-Pompa and Kaus 1992) would add the local peoples, their culture, and their "indigenous knowledge" to the list of what should be retained. The definition used here is considerably broader than that used historically by biogeographers, community, and evolutionary ecologists. Magurran (1988) recently observed that "Diver-

sity is rather like an optical illusion, the more it is looked at, the less clearly defined it appears to be and viewing it from different angles can lead to different perceptions of what is involved." Peters (1991) further warns that biodiversity should not be viewed as a single phenomenon, but as a "concept cluster". Stability and sustainability are also related concept clusters; thus if these notions are combined with biodiversity, particularly complex issues arise.

Genetic Diversity

Each plant, animal, and microbial species occurring on rangelands has variation in its DNA-based genetic structure over space and time. Genetic diversity found within species is the ultimate source of biodiversity at higher levels, determining how species interact with their environments and each other. Generally, the shorter the life span and the greater the role of sexual reproduction, the greater the genetic variation, although other features such as mating systems (Pimm 1991) may influence these traits. While range scientists have long recognized the importance of empirically identifying ecotypes of important plants used in planting efforts, the genetic reasons for these differences are only beginning to be investigated. Durant McArthur's work with sagebrushes (*Artemisia* spp.) and Douglas Dewey's work with wheatgrasses and ryegrasses (Triticeae tribe of grasses) are notable exceptions. Less conspicuous and less economically important species have rarely had their genetic structure considered. Fortunately, new techniques are becoming available to accelerate this type of work in the future.

Species Populations

As mankind has gained increasing power to change the biosphere, there is little doubt that rates of extinction have begun to greatly exceed rates of evolution of new species (Ehrlich and Wilson 1991). Because of the inertia of various human tendencies now in place, loss of many more species is inevitable (Holdgate 1991). The majority of the world's species are naturally rare (Williams 1964), many being relicts from former times with different environments. About half of the world's species, mostly insects, occur in the tropics. The reputation of the world's drylands as having low biodiversity is, however, false (Cody 1989, Pimm and Gittleman 1992).

Disturbance tends to favor those organisms with broad ecologi-

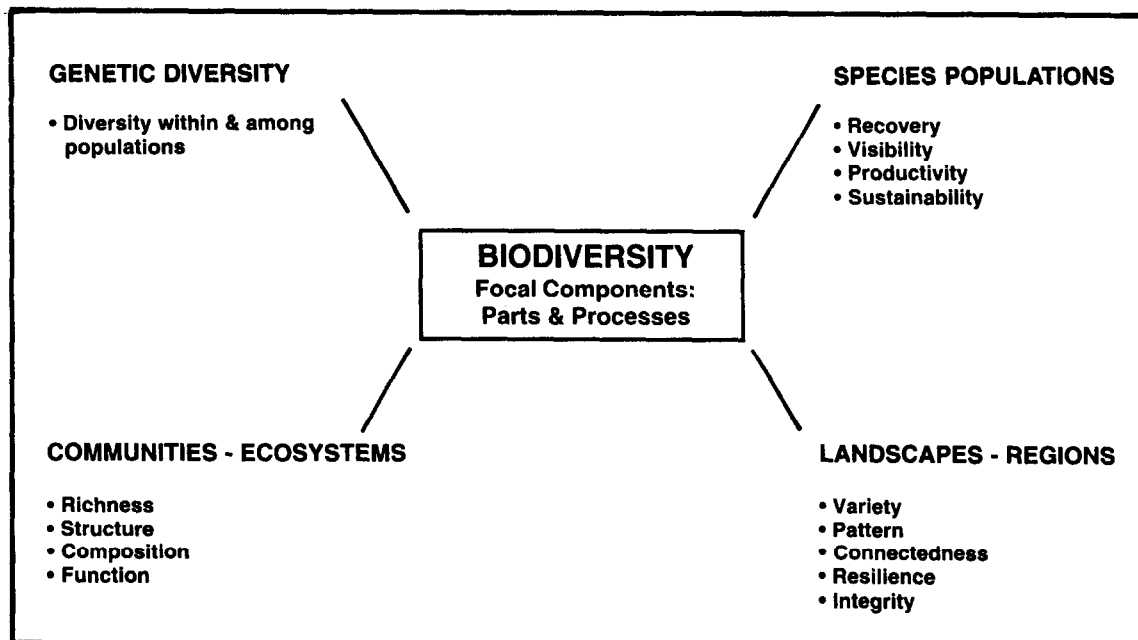


Fig. 1. Components of biodiversity. From Anon 1991b. Reprinted with permission of the Keystone Center.

cal tolerances and shorter life cycles. Because agriculture has historically favored cultivation of short-lived species with rapid growth rates, "weedy" species (both plant and animal) have increased in proportionate abundance over the nonweedy ones. Moreover, as the proportion of the earth that was wildland has diminished, we have observed the loss of species requiring the more stable conditions of nondeveloped areas. The cosmopolitization of the world's fauna and flora in disturbed habitats is viewed as "biological pollution" to some (Howarth 1983).

Communities

Associations or assemblages of species make up communities. Any management action or inaction (failure to actively manage) involves a trade-off between species that benefit from subsequent changes and those that do not. Consequently, species occurrences and their relative abundances can be altered due to management actions, such as changes in livestock grazing and/or fire regimes. Although the data have rarely been evaluated in ways to bring out the impacts of various management systems on community diversity, there are a myriad of techniques to collect and synthesize such data at the community level (Magurran 1988). These will be discussed later.

Ecosystems

An ecosystem is the biological community in an area plus the physical environment with which it interacts. The variety of species, growth forms, life cycles, and phenological and physiological modes (e.g., C₃, C₄, CAM plants) present in a community qualitatively and quantitatively influence the environment and processes operative at the ecosystem level. For instance, whether leguminous or nonleguminous or otherwise N-fixing shrubs are allowed to prosper in savannas has a profound influence on the spatial variability in micro-climate, water budgets, nitrogen fixation, and animals present and their plane of nutrition.

Landscapes and Regions

Biodiversity is also patterned at the scales of hillslopes, watersheds, and landscapes. Romme and Knight (1982) suggest that changes in landscape pattern may influence a variety of other natural features including animal use, water, and nutrient cycles, and even the probability of different kinds of natural disturbances. Kinds of ecosystems, their level of various functions (e.g., productivity, nutrient cycling) and linkages (e.g., water, sediment) are being increasingly scrutinized as advanced remote sensing, geographic information systems, computerized image interpretation, and multivariate statistical techniques allow data to be organized across broad scales (Pickup 1989). These tools are also allowing a new consideration of regional patterns of diversity now becoming known as epsilon diversity (Burton et al. 1992). Gap analysis (Scott et al. 1987, 1988, 1989) is attempting to recognize where discontinuities occur in protection, largely between the national reserve and refuge system.

Importance of Scale

It should now be obvious that biodiversity can be considered at a variety of scales in both space and time. Whether a short-term change in local species richness can lead to a significant long-term change in overall biodiversity is a question of temporal and geographic scales. If the organisms that are harmed by a given management action are already rare or more imperiled than ones that benefit, or if the manipulation eliminates a local population or a species throughout its range, a community, or an ecosystem process, then biodiversity is reduced. If the manipulation eliminates an element that is common elsewhere in the landscape or provides an opportunity for an imperiled element in the landscape to migrate or increase, then biodiversity is more secure. It is thus obvious that we have to consider the impacts of any project at place and times

well beyond the point of action. Because benefits are typically economic, immediate, and known, while costs are ecological, long term, and unknown, it is very difficult to objectively balance costs and benefits (Solbrig 1992).

The fundamental problem with more tightly defining biodiversity is that different people favor looking at different phenomena at much different scales in space and time (Table 1). The "diversity" of scientists, technicians, users and other interested parties with different value systems involved in such a vast array of features is bound to create difficulties in reaching consensus on what to measure and how to measure it. Thus, the only reliable means of knowing what people really mean when they use the term biodiversity is to ask what and how they would measure. Their answers must clearly spell out which levels of integration and temporal and spatial scales they favor. Furthermore, without a tightly stated purpose for estimating diversity, we cannot decide on the most relevant measure(s) of biodiversity. Failure to make objectives explicit insures continued frustration because what constitutes ecological good remains as much a matter of human opinion as it is a subject of science.

One must be aware that patterns and processes at 1 spatial or temporal scale affect those at other scales. For instance, diminishment of migratory small birds, largely because of loss of tropical winter habitat, may eventually influence ecosystem structure at higher latitudes because of changed seed predation, seed dissemination, pollination, etc., which could lead to changing vegetation structure. Greater diversity at 1 spatial scale need not translate to greater diversity at another scale (Hoover and Parker 1991). Monitoring a suite of phenomena simultaneously at several scales becomes prohibitively expensive. Thus, we must learn how to relate the various features and levels to each other and then match the combinations with the most efficient monitoring approaches.

Why Biodiversity Is Important

Changes in biodiversity, by whatever measure used, should be of concern to everyone for at least 1 of 4 basic reasons. The *first* reason involves morality. Many believe that humans have a moral obligation to protect their fellow creatures, whether any extrinsic value is placed on them or not. The relative strength of this feeling depends on one's own philosophical and religious orientation, namely mankind's place in nature. The *second* reason involves aesthetics. The human desire to see and appreciate the living parts of nature and add interest, value, and quality to their lives is reflected in the popularity of wildlife films, visits to zoos, botanical gardens, and ecotourism. The *third* reason involves economics. In addition to the economic spin-offs of the aesthetic values (e.g., ecotourism), there are many direct benefits to humans from biodiversity in the form of "goods" such as foods, medicines, fuel, building materials, and industrial products. Crops and domestic animals are "borrowed" from the "genetic library" of nature (Ehrlich and Wilson 1991). The potential of that "library" has scarcely been tapped. Some of these potential uses will be forever lost by allowing extinction to accelerate.

The *fourth*, least understood, but most important value of biodiversity is the array of "services" provided by natural ecosystems (Westman 1977, Walker 1992). Essential ecosystem services include maintenance of the gaseous composition of the atmosphere; amelioration of climates; genesis, fertility, and stability of soils; disposal of wastes; cycling of nutrients; natural control of pathogenic and parasitic organisms, etc. Loss of biodiversity can negatively influence both the quality and quantity of ecosystem services and ultimately have economic consequences. Although it is logical to expect that reducing the biological variability of a system reduces its resilience and increases the probability that rare events which formerly could be "absorbed" will cause dramatic change and

Table 1. Indicator variables for inventorying, monitoring, and assessing terrestrial biodiversity at 4 levels of organization, including compositional, structural and functional components, inventory and monitoring tools and techniques (from Noss 1990). Reprinted with permission of the Society for Conservation Biology and Blackwell Scientific Publications, Inc.

Levels of	Classes of Indicators			
	Composition	Structure	Function	Inventory and monitoring tools
Regional Landscape	Identify, distribution richness and proportions of patch (habitat) types and multi-patch landscape type; collection patterns of species distributions (richness, endemism)	Heterogeneity; connectivity; spatial linkage; patchiness; porosity; contrast, grain size; fragmentation; configuration; juxtaposition; patch size frequency distribution; perimeter-area ratio; pattern of habitat layer distribution	Disturbance processes (areal extent, frequency or return interval, rotation period, predictability, intensity, severity, seasonality); nutrient cycling rates; energy flow rates; patch persistence and turnover rates; rates of erosion and geomorphic and hydrologic processes; human land-use trends	Aerial photographs (satellite and conventional aircraft) and other remote sensing data; Geographic information Systems (GIS) technology; time series analyses; spatial statistics; mathematical indices (of pattern, heterogeneity, connectivity, layering diversity, edge, morphology, autocorrelation, fractal dimension)
Ecosystem Community	Identify, relative abundance, frequency, richness, evenness, and diversity of species and guilds; proportions of endemic, exotic, threatened and endangered species; dominance—diversity curves; life-form proportions; similarity coefficients; C ₃ -C ₄ plant species ratios	Substrate and soil variables; slope and aspect; vegetation biomass and physiognomy; foliage density and layering; horizontal patchiness; canopy openness and gap proportions; abundance, density and distribution of key physical features (e.g., cliffs, outcrops, sinks) and structural elements (snags, down logs); water and resource (e.g. mast availability; snow cover	Biomass and resource productivity, herbivory, parasitism, and predation rates, colonization and local extinction rates; patch dynamics (fine-scale disturbance processes), nutrient cycling rates; human intrusion rates and intensities	Aerial photographs and other remote sensing data; ground-level photo stations; time series analysis; physical habitat measures and resource inventories; habitat suitability indices (HSI, multispecies); observations, censuses and inventories, captures, and other sampling methodologies; mathematical indices (e.g. of diversity, heterogeneity, layering dispersion, biotic integrity)
Species Population	Absolute or relative abundance; frequency; importance or cover values; biomass, density	Dispersion (Microdistribution); range (macrodistribution); population structure (sex ratio, age ratio); habitat variables (see community-ecosystem structure, above); within-individual morphological variability	Demographic processes (fertility, recruitment rate, survivorship, mortality); metapopulation dynamics; population genetics (see below); population fluctuations; physiology; life history; phenology; growth rate (of individuals); accumulation; adaptation	Censuses (observations, counts, captures, signs, radio-marking); remote sensing; habitat suitability index (HSI) species habitat modeling; population viability analysis
Genetic	Allelic diversity; presence of particular rare alleles, deleterious recessives, or karyotypic variants	Census and effective population size; heterozygosity; chromosomal or phenotypic polymorphism; generation overlap heritability	Inbreeding depression; outbreeding rate: rate of genetic drift; gene flow; mutation rate; selection intensity	Electrophoresis; karyotypic analysis; DNA sequencing; offspring-parent regression; sib analysis; morphological analysis

injure the system (Archer and Smeins 1991), the science so far brought to bear on this topic has not been strong (Peters 1991, Solbrig 1992, Walker 1992, Johnson and Mayeux 1992, Allen and Hoesktra 1992).

Measurement of Diversity at the Community Level

Methods for assessing biodiversity vary among hierarchial levels

of organization (Table 1). Some basic concepts and principles of measurement are similar, however. Because community level aspects of measuring biodiversity are of greatest inherent interest to range professionals and have had the longest history of development, methodological focus herein will be for assessing biodiversity at the community level of organization. Some of these techniques can also be employed at landscape levels (Turner et al.

1990, Burton et al. 1992).

Community biodiversity, also sometimes known as intra-stand or alpha biodiversity, deals with the numbers of taxa and their relative abundances within a stand (an area of sufficient homogeneity with regard to vegetation, soils, topography, micro-climate, and past disturbance history to be treated as a single unit). The usual compound expressions of community biodiversity really involve 2 fundamental features: (1) richness, also sometimes known as variety or species density; and (2) relative abundances of component taxa, also called equitability or evenness. Most of the suggested indices of community level biodiversity (over 90 proposed indices have been published, but an infinite number of indices are possible [Ludwig and Reynolds 1988]) involve both richness and equitability, often in ill-defined ways.

Richness

Because increasing number of taxa are usually encountered as the cumulative area sampled increases (species-area curve, Fig. 2), any small sample will usually underestimate the maximum richness

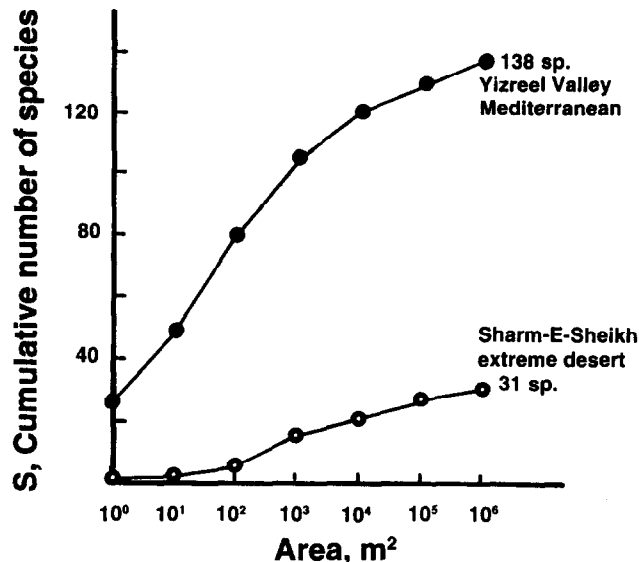


Fig. 2. Species-area curves for Yizreel Valley and Sharm-E-Sheikh regions of Israel (Shmida and Wilson 1985). Reprinted with permission of Blackwell Scientific Publications, Ltd.

possible within a stand. When a sample is used to estimate stand richness, it is assumed some predictable relationship exists between the actual number of taxa sampled (s) and the real total number of taxa (S) present. It is often assumed that s is directly proportional to the logarithm of area (a) sampled. If true, one could theoretically extrapolate from any size sample to an entire area about which one chooses to generalize. Unfortunately, no single equation has been found to be valid in all circumstances, and all empirically derived curves have been nonlinear. Furthermore, in certain instances, species richness has been found to not increase as size of area sampled increased (Boeklen and Gotelli 1984, Dunn and Loehle 1988). The rate at which species richness increases, or not, with area is related to the combined effects of 4 biological determinants: niche relations, habitat diversity, mass effects, and ecological equivalency (Shmida and Wilson 1985).

Richness is affected also by evenness, spatial pattern (patchiness), and natural disturbances or human-induced treatments. Thus, adjusting plot size and shape and increasing plot numbers to compare different stands may not yield statistically valid results (Christensen and Peet 1984, Palmer 1990). All we currently know is that, given equally rich communities, numbers of taxa will usually increase toward an asymptote (levelling off of the curve) most rapidly in those communities in which the constituent taxa are most equally abundant, small in size, and randomly intermingled in

space. As community equitability decreases, organism size increases (due to human treatments or natural causes), or taxa become spatially segregated, more samples are required to observe a particular portion of the total taxa potentially present.

Equitability

Relative abundance or evenness of taxa is frequently viewed as more important to land resource managers than richness. For instance, rangeland managers often prefer an abundance of only a few highly palatable plant species when their goals center on livestock production. Thus, they, do not want to treat all plant species equally in a diversity index.

Several metrics of equitability have been developed from heterogeneity (richness combined with equitability) measures by standardizing against the maximum possible value. The major difficulty with these metrics is that they require knowing the actual number of species (S) present in the sampling universe. Because S is usually underestimated from subsampling (Palmer 1990), equitability is usually overestimated. Obviously, variations in sample size, as well as random variations in s , or the distributions of abundances among species affect these measures (Peet 1974). Under most circumstances, it is best to avoid trying to express equitability by itself (Christensen and Peet 1984). Where species presence is nearly equivalent, however, it is useful to graphically tease out differences in equitability over either time or space to show how communities change in terms of relative abundances of species and growth forms (e.g., Lewis et al. 1988, Yorks et al. 1992).

Heterogeneity

Most ecologists have preferred to combine richness and equitability into a single measure known as heterogeneity (Peet 1974). Heterogeneity can be equated with the uncertainty that exists regarding the taxonomic or other group membership of an individual selected at random from a community (Pielou 1966). The uncertainty of membership increases with either increased richness or decreased equitability. Because information content is a measure of uncertainty, information theory-based indices are often used to express heterogeneity (Ludwig and Reynolds 1988).

Simpson (1949) first proposed that heterogeneity is inversely related to the probability that 2 individuals selected at random from a community will belong to the same species. Routledge's (1979, 1984) modification of Simpson's index of concentration can be used at several hierarchical levels within and beyond stands. Estimates of variance and tests of statistical significance can be done with computer-intensive methods (Hatton and West 1987).

Information theory-based heterogeneity indices are more sensitive to changes in rare species than the probability theory-based indices (e.g., Simpson's and its modifications), which are more sensitive to changes in common species. The appropriate use of these indices thus varies depending upon whether one wants to stress the total biota or the more abundant taxa within the community. It should be realized, however, that information of usually greatest interest to a manager becomes submerged from view by such data compressions into an index number (which can be arrived at an infinite number of ways).

Cardinal measures (Cousins 1991) such as richness treat all taxa equally. All of the information theory-derived indices (Williams alpha, Shannon's H and J , as well as others, see Hill 1973) are cardinal indices.

Ordinal measures, which treat each taxon differently and express their ranking in a particular order, are more ecologically sound (Walker 1992) and offer advantages for environmental assessment (Cousins 1991). Relative statistical abundance distributions (such as log-normal, Fig. 3) are not the only ordinal tool we have for analyzing diversity, but ordering by size and functional

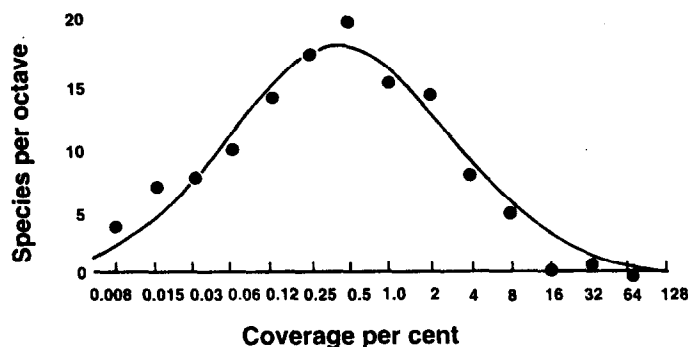


Fig. 3. Log-normal statistical distribution of Sonoran Desert plants (Whittaker 1965). Reprinted with permission of the American Association for the Advancement of Science. Octaves are classes of abundance ordered by doubling of cover values.

aspects should be more common in the future (Cousins 1991).

Wayne and Bazzaz (1991) point out that although indices based merely on species may be useful, species (except microorganisms [Chapin et al. 1992]) are generally unreliable units for quantifying the ecological diversity perceived and utilized by most organisms in communities because they are based on taxonomic, not functional bases (Walker 1992). Limiting our analyses to just species designations greatly under-estimates the complexity of communities, ecosystems, and landscapes. Some species-rich communities may be much less ecologically diverse than species-poor communities. For instance, a community with a lot of similarly sized grasses and forbs would be less structurally and functionally diverse than a community with a single abundant shrub species and fewer grasses and forbs. Wayne and Bazzaz (1991) propose categorizing organisms by genetic, phenetic, gender, developmental state and/or maternal effects to obtain additional insight about how communities are structured and function.

Choice of Method

There is no such thing as an "objective" index of biodiversity. The numerical values produced are dependent on human choices of the area to be sampled, how often it was sampled, any selection of variable(s) to express presence or dominance and the particular formulae used to synthesize the data (West and Reese 1991). Interpretation and comparison of numbers expressing community biodiversity are difficult. The unaware or unscrupulous can, intentionally or not, "bend" the numbers to favor certain conclusions. Methods should be selected on the basis of the question(s) being asked of the survey, experiment, or monitoring program.

A numerical index may stay the same or nearly so, but a community could change significantly, such as reductions in the abundance of major species. This is because most indices treat all species as equals. Several species could be lost, but if several new species replace them at about the same relative abundance, the index values would remain about the same. One thus needs to compare similarity or dissimilarity as well as diversity. In other words, turnover of taxa and functional groups over time or space needs to be specified. Simply maintaining community diversity as expressed in an index does not ensure maintenance of quality. We have to also be concerned with maintaining the desired species in the desired communities. Choice of benchmarks for comparison is also critical (West 1991a).

Impact of Current Laws and Regulations

Biodiversity has had such intuitive appeal to the public that their legislators have passed laws despite immaturity of the underlying science (Redford and Sanderson 1992). The 2 major U.S. laws in

place that deal with biodiversity on rangelands are the National Forest Management Act of 1976 (NFMA) and the Surface Mining Control and Reclamation Act of 1977 (SMCRA). There are, however, numerous other laws, regulations, and executive orders that also require consideration of biodiversity (Kirby 1984, Stockwell 1990).

National Forest Management Act

The National Forest Management Act requires the U.S. Forest Service to consider natural diversity as a requirement, criterion, and output of good wildland management. Section 6 (h) defines diversity as "the distribution and abundance of different plant and animal communities and species within the area covered by a land and resource management plan." Furthermore, the management prescription for national forests "where appropriate and to the extent practicable, will preserve and enhance the diversity of plant and animal communities." Sec. 219.13g also calls for maintaining viable populations of native vertebrates. Regulations implementing the act require maintenance of natural diversity on national forest lands and call for environmental monitoring to insure that natural diversity is maintained. The Forest Service is still grappling, however, with ways to meet these goals (McMinn 1991). The Klamath National Forest in northern California is probably the lead Forest Service unit in devising inventory, planning, action, and monitoring procedures to accommodate biodiversity (Williams and Marcot 1991).

Surface Mining Control and Reclamation Act

The Surface Mining Control and Reclamation Act requires the establishment of a: "...diverse, effective, and permanent vegetation cover...capable of self-regeneration and plant succession at least equal in extent of cover to the natural vegetation of the area..." on all lands after disturbance from surface mining. Of the sundry requirements for revegetation success, the re-establishment of diversity may be the most difficult to satisfy (Laycock 1980, DePuit 1984, Hatton et al. 1986, Hatton and West 1987).

The state regulatory agencies for coal stripmining have not taken a broad view of how to measure diversity. Currently, their focus is on intra-community (alpha) plant diversity only (Krabbenhoft et al. 1991), although inter-community (beta) and landscape (mosaic or gamma) diversity may be equally important in reestablishing wildlife habitat desirability (Laycock 1980, Krabbenhoft et al. 1991). The prevailing choice of an alpha index is the Shannon-Wiener index, which weights species equally via information theory and thus favors the establishment and persistence of rare species over abundant species. Consequently, it is more difficult to show an improvement in biodiversity and to subsequently obtain bond release than if a probability theory-based index were used. Moreover, the use of information theory-based indices limits habitat quality in terms of wildlife, livestock, and watershed considerations. Some erroneous assumptions about how diversity relates to stability (Peters 1991, Allen and Hoeskstra 1992) lie at the core of these missed opportunities.

Biodiversity and Rangeland Management

Having reviewed the general problems with defining and applying these principles and measuring biodiversity in special circumstances, let us turn to how these issues relate to all kinds of rangelands.

Concentration on Vertebrates

The biodiversity issue first gained public attention in the U.S. because of the concern for diminishing prospects for survival of some vertebrates, (the often-called "charismatic fauna" or "flagship" species [Noss 1990]). These organisms were first formally dealt with after passage of the Endangered Species Act in 1973.

Those species, already having low densities and isolated distributions, are the ones most vulnerable from any type of single influence or cumulative effects. Monitoring of population dynamics of vertebrates and separating natural from human-induced impacts on densities is notoriously difficult (Pechmann et al. 1991). Furthermore, management policies that focus monitoring efforts too high on the food chain run the risk of permanent damage to more basic components (e.g., soils [Sidle and Sharpely 1991]) and functions (Walker 1992) of ecosystems. This is because statistically significant changes in animal populations usually lag behind significant changes in lower trophic levels and the supporting environment.

The Natural Heritage programs in each of the United States and European countries have already identified the rarer vertebrate and vascular plant species and where they are located (Stolzenberg 1992). We can now use care in managing around such taxa. We do not, however, generally have similar information in other parts of the world. We lack similar data for invertebrate animals, nonvascular plants, and microbes, even in Europe, the USA, and other developed countries. Such less noticeable species may even be more important for maintaining ecosystem "health" or affecting ecosystem recovery than the more visible organisms (Perry et al. 1989). Large terrestrial mammals make up only 0.02% of all North American species (Grumbine 1990), yet they have received the "lion's share" of our attention.

Wide-ranging vertebrates, particularly predators and carrion-eaters, have been the most visibly impacted animals on rangelands. I can point to several that are jeopardized by activities ancillary to the management of livestock on rangelands—the grizzly bear, Mexican wolf, black-footed ferret, and California condor. The Mexican wolf now survives only in *ex situ* (off-site) protection. That is, they have been removed from the wild and are being bred in captivity. The black-footed ferret has been and continues to be captive-bred; captive-bred individuals were recently released on rangelands in Wyoming's Shirley Basin. Two captive-bred condors were released in Southern California in 1992. Reestablishment of the ferret, condor, and Mexican wolf in the wild is questionable. The cost of such programs is enormous and detracts from proactive efforts to keep other species from joining the rare and endangered lists. We can expect the management of the most promising sites for reintroduction of such "emergency room" species to be carefully monitored and controlled to try and save these and other publicly favored species. If you deal with rangeland in such areas, be prepared for these influences on your style of land management.

Causes for Declining Diversity on Rangelands

Species Level

While continuous, heavy livestock grazing and trampling can reduce some rare plant species (e.g., the uncharismatic, but taxonomically distinctive *Oryctes nevadensis* in California [Holsinger and Gottlieb 1991]), especially palatable herbs, indirect effects are probably more significant for animals. For instance, diminishment of the Hawaiian crow and many other species in the tropics is apparently due to clearing of forests to create pastures for cattle (Holden 1992). Changing vegetation structure does indeed influence the variety and abundance of particular native animals. Where livestock grazing has gone on more moderately, without associated mechanical or chemical vegetation manipulations, what is cause and effect is far more questionable. For instance, decline of the masked bobwhite quail is correlated with the expansion of tall shrubs into formerly desert grasslands along the Arizona-Sonora border. The change to greater amounts of shrubs and succulents has, however, been going on for a long time (Van Devender 1990) and how much of this change can be attributed to livestock versus climate and other causes of change is debatable (Bahre 1991). Expect similar debates to develop concerning mountain quail, sage grouse, and sharp-tail grouse in the Pacific Northwest.

Another herbivore that seems to be declining in the southwestern deserts is the desert tortoise. Although some attempt connecting the decline of this animal to livestock grazing (Berry 1978), livestock use had declined well before tortoise densities slipped and recent studies have shown that it prefers grazing on the introduced annuals (Bostick 1990). It thus seems that other influences such as increased predator densities (e.g., a 15-fold increase in ravens over the past few decades), increased off-road vehicular traffic, diseases, and direct human removal could well be more important than vegetation changes and cattle trampling of animals and dens (Huxtable 1992). Removal of livestock from public rangelands occupied by the desert tortoise is an action that is more politically than scientifically supportable.

Some rare species are favored by ungulate grazing. For instance, the mountain plover nests only in relatively heavily grazed short-grass steppe (Graul 1973, 1975; Ryder 1980; Leachman and Osmondson 1990). This bird probably evolved with bison influences that only cattle can now replace. Ryder (1980) notes that bird species can be placed into decreaser, increaser, and invader categories as easily as plants. The buttercup, *Ranunculus ophioglossifolius*, in Great Britain can only be sustained where heavy sheep grazing is maintained (Frost 1981). Sheep grazing was also found to be beneficial to populations of the early spider orchid (*Ophrys sphegodes*) in England, apparently because maintenance of a short turf reduces interspecific competition (Hutchings 1987). Earlier cattle grazing was leading to declines of this species, apparently because of less close cropping of competitors and mechanical hoof damage. We can thus expect livestock to be used in some instances to actually enhance conservation of species preferring certain successional niches.

Any comprehensive biodiversity program must include threatened and endangered species as a major priority. High priority must also be given to approaches that are more holistic than the piecemeal, crisis efforts into which threatened and endangered species force the scientific and land management communities. Working with individual species *ad finitum* focuses on symptoms rather than underlying causes.

Community Level

We have long known that herbivores influence community-level diversity through differentially utilizing or trampling plants variously susceptible to defoliation and other physical damage. For instance, Waser and Price (1981) show how heavy, year-long grazing can diminish the variety of herbs near water and fence corners in the Sonoran Desert. Moderate intensities and periodicities of grazing and trampling usually increase community level diversity of plants by decreasing the ability of community dominants to competitively exclude other species (Quinn and Robinson 1987) and by creating physical gaps and freeing resources such as light, moisture, and nutrients (Archer and Smeins 1991). Collins and Barber (1985) evaluated the effects of fire, grazing, bison wallows, and prairie dog towns on plant diversity in a mixed-grass prairie in southwestern Oklahoma. Plant species diversity was highest on grazed wallows without fire. Lightly to moderately grazed areas had somewhat less diversity. The lowest diversity was associated with recently burned, ungrazed sites. Bonham and Lerwick (1976) found that plant community richness was greater on prairie dog towns than for adjacent uncolonized areas within shortgrass prairie of northeastern Colorado. Similar trends were noted elsewhere by Hansen and Gold (1977), Archer et al. (1987), and Coppock et al. (1983).

The effects of grazing on plant community-level diversity depends on grazing intensity, evolutionary history of the site, and climatic regimes (Milchunas et al. 1988). In semiarid grasslands with a strong evolutionary history of grazing, herbivory appears to have a relatively small effect on community composition (Fig. 4;

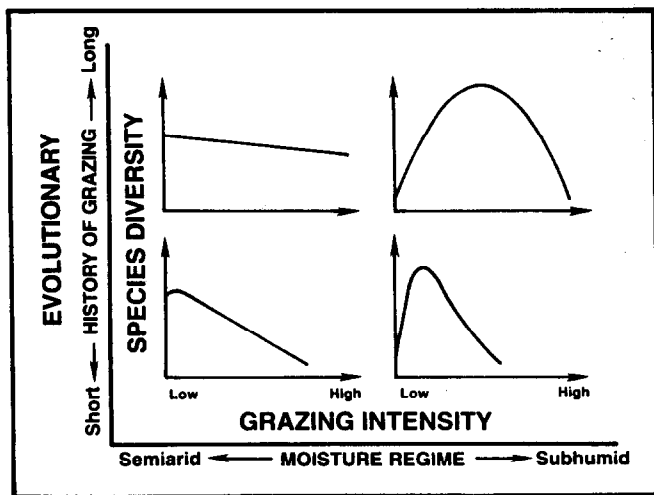


Fig. 4. Hypothesized relationships between relative plant species diversity in grasslands in relation to grazing intensity along gradients of moisture and evolutionary history of grazing (Milchunas et al. 1988). Published with permission of the University of Chicago Press.

e.g., the shortgrass plains of the U.S. and Canada). In contrast, climatically similar grasslands with a shorter evolutionary history of large mammal grazing lose diversity at much lower grazing intensities; e.g., Argentine pampas. Under wetter conditions, moderate grazing usually enhances diversity regardless of differing evolutionary history (Fig. 4). For instance, Ellenberg (1988) proposes that livestock grazing will have to be increased to enhance diversity of herbaceous plant communities of central Europe which are experiencing dramatic nitrogen and sulfur enrichment due to air pollution which drives dominance by a few species thriving under high nutrient inputs.

Another demonstration of how livestock grazing can have a role in enhancing community biodiversity comes from Israel where Naveh and Whittaker (1979) showed that greater plant diversity for several rangeland sites in Israel existed on moderately grazed areas rather than either ungrazed or heavily grazed areas (Fig. 5). Mueggler (1984) demonstrated similar patterns for sagebrush

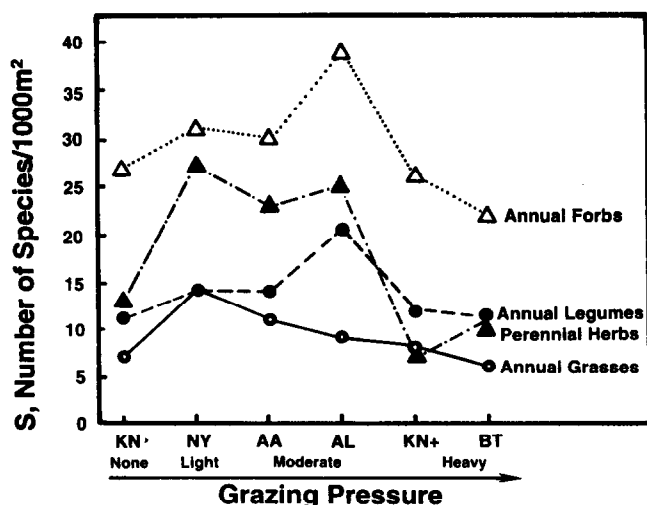


Fig. 5. Species richness in major herb groups of open woodlands and shrublands in relation to grazing pressure (Naveh and Whittaker 1979). Reprinted with permission of Kluwer Academic Publishers.

steppe and aspen rangelands in Montana. Ungulate grazing is an important process in many rangeland ecosystems. If grazing is excluded, biodiversity may increase in the short term, but may decline long-term because the system itself changes and in the future may be less able to withstand other disturbances such as drought and fire (Walker 1989).

Ecosystem Level

Whether losses of species lead to community-level reorganizations which are functionally important, requires careful, long-term observation of ecosystems (Holdgate 1991, Walker 1992, Solbrig 1992). For instance, although loss of the American chestnut and the passenger pigeon from the eastern deciduous forest region of the U.S. led to some short-term economic impacts for particular people who harvested them for fiber and food, other species filled in the space and used resources these species once tied up (Johnson and Mayeaux 1992). No undesirable long-term changes in total productivity and watershed protection were noticed. Resilience at the community and ecosystem levels was thus high for these particular losses because of apparent structural and functional redundancy.

The functional consequences of species-level biodiversity at the ecosystem level is still being debated (Chapin et al. 1992, Walker 1992). Most ecosystems exhibit species richness higher than that required for efficient biogeochemical and trophic functions (di Castri 1991). Functional groupings have been proposed as a means of aggregating species having similar effects on ecosystem processes (Walker 1992). Because we can never determine how each species affects all ecosystem functions, such groupings may be a practical necessity. Where there is more than 1 species per functional group, it implies that species within a functional group may be equivalent or redundant in their impacts on ecosystem processes and that ecosystems could function equally well with fewer species.

Theories of community-ecosystem interaction range from the "rivet" hypothesis (Ehrlich and Ehrlich 1981), where each species plays an incrementally important role (like rivets in an airplane), to that of communities composed of a few functional groups, each with several ecologically equivalent species such that some may be lost with little effect on ecosystem processes (the full redundancy hypothesis). The truth probably lies somewhere between these extremes (Walker 1992).

Redundancy can be thought of as "insurance" or "backup" because each species can tolerate only a limited range of climatic and biotic conditions. If there are several species in each structural and functional group, those structural and functional features are more likely to persist under global environmental change. We know, for instance, that loss of major trees or shrubs in some rangeland systems can have cascading effects through food chains, energy flow, water, and nutrient cycles. For instance, removal of trees and shrubs from subtropical rangelands results in a reorganization of native bird (Johnson et al. 1980) and mammal communities, micro climates (Belsky et al. 1989), water budgets (Blackburn 1983, Carlson et al. 1990), and the nitrogen cycle (West 1991b). Leaving a patchwork of trees and openings can actually optimize the mix of cattle and wild ungulates, apparently because of enhancement of browse production and edge (Fulbright and Beasom 1987). Whether those changes are desirable or not depends on who is paying the bills and benefiting from the consequences.

Landscapes

If paddocks or allotments are large enough, and water and salt are available only at certain points, then gradients of disturbance and consequent successional status should be available across the landscape (Andrew 1988, Laycock 1992). Thus, it is conceivable that total diversity across a landscape could be enhanced by some

patterns of livestock use. Furthermore, some treatments such as prescribed burning and chaining can actually enhance community and landscape diversity in some rangeland settings (Mueggler 1984).

Regions

A major threat to sustaining biodiversity at all lower hierarchical levels is reduced size of contiguous habitats. Habitat fragmentation is regarded by many as a major problem influencing land management (Burton et al. 1992). Blockstein (1990) maintains that preservation of multiple examples of all natural communities occurring in the U.S. should be a national goal. The assumption is that maintaining communities or ecosystems automatically ensures perpetuation of a large component of species. The validity of this approach depends on selection and application of community and ecosystem classifications. Most classification systems are based on a small number of dominant species; the mere occurrence of those species at any age or state of community development does not ensure that all potential component species and functions exist (McMinn 1991).

Potential Solutions

Rather than wait for others to impose their will, I strongly suggest that academia, the professional societies, and the resource management and advisory agencies take a proactive stance and begin making their own suggestions of what diversity is, how it should be inventoried, monitored, and incorporated into management plans and actions.

Genetic Resources

With the potential of transgenic (gene splicing between taxonomically disparate organisms) manipulation now proven, any species could be genetically useful to mankind in the future. Furthermore, particular genetic populations could become more valuable than the common ones we have used thus far. For instance, ecotypes of grasses with unusual tolerance to heavy metals have become valuable in revegetating areas disturbed by mining (Wainwright and Woolhouse 1975). Other populations are naturally selected by grazing, fire, herbicides, etc. Range practitioners can use such material to advantage in rehabilitation activities. Until details of genetic variation of many more species are available, it behooves us to protect isolated populations.

Species Protection

It is rapidly becoming evident that not all species now found on earth will remain during the 21st century (Ehrlich and Wilson 1991, Holdgate 1991). Furthermore, it has become evident that passive management or attempts at preservation do not protect all species. For instance, Newmark (1987) found that between 0 and 43% of the original (pre-establishment) large mammal fauna have been lost from our large western national parks in the U.S., the variation depending largely on their size and time since establishment. Furthermore, broadening "preservation" attempts will eventually curtail economic growth. We will thus be forced to decide which aspects of biodiversity are most important. Adherence to policies that place equal emphasis on every species is both ecologically unsound and tactically unachievable (Walker 1992).

Communities, Ecosystems, Landscapes

With the creation of national plans for further wildlife reserves and corridors (Scott et al. 1987), the greater ecosystem concept (Grumbine 1990), and regional assessment of environmental health through the Environmental Protection Agency's Environmental Monitoring and Assessment Program (EMAP) (Hunsaker and Carpenter 1990), even private landholders will probably see further governmental control of how their lands can be managed, in the name of maintaining or enhancing biodiversity. The public is beginning to realize that not all biota are preserved or protected in

existing parks, preserves, and refuges (Pimentel et al. 1992). Furthermore, buffers of surrounding land with less heavy use act as corridors of seminatural environments allowing movements of organisms between the parks and preserves, particularly under climatic change scenarios (Peters and Darling 1985). Thus, the importance of other federal, state, and private wildland is being recognized for conservation biology efforts (Brussard 1991).

While most U.S. rangeland surviving until the twentieth century has been marginal for agricultural, urban, and industrial development and consequently less fragmented than ecosystems in wetter parts of the country, even low levels of fragmentation may result in ill-functioning communities (Hanson and Angelstam 1991). We must expect greater efforts to protect declining and under represented ecosystem types (e.g., tall grass prairie, California steppe) in national preserve systems. There will also be increasing attempts to restore missing species and community types (e.g., blackland prairie, longleaf pine-wire grass).

It should be recognized that most of the ideas being pushed by conservation biologists are based on the theory of island biogeography. Not all accept that such theory is applicable to mainlands (Mann 1991). Some continental species can escape into bordering areas once their prime habitat is removed (Boeklen and Gotelli 1984, and Dunn and Loehle 1988). The vast majority of workers in conservation biology have been trained in a species orientation. Accordingly, they have been slow in realizing that the best way to minimize species losses is to maintain ecosystem integrity.

I favor prioritization of maintenance of soil profiles and ecosystem functions as the most important aspects of sustainability (West 1991b). Without these features, overall potentials for all biota are seriously diminished. Furthermore, holding the soils on upland sites and out of aquatic environments seems very important to maintain options for future generations. Global warming may displace temperature zones in middle latitudes of 4 to 6° northward or 460 to 1,100 m upslope over the next 100 years (Nielsen et al. 1989). Thus, we do not know where our farmlands will have to be in the future. If the tools of the Green Revolution lose their edge, more marginal arable land will have to be brought into production (Moffat 1992). The world's food supply will have to be doubled by 2010 in order to feed the 90 million more people per year being born. Soils spread across a landscape are better insurance than only rocks on hillsides and deep sediments in depressions.

Research Needs

Either active (multiple use) or passive management (preservation) of an ecosystem results in adjustments in the population densities of all species and inevitably leads to a shift in relative abundances of co-existing species (community composition). All species are not created ecologically equal. Some are "drivers" and some are "passengers" (Walker 1992). The extent of change will largely depend on the tightness of coupling of the major species to others in the food web (Pimm 1991).

Species whose direct or indirect effects on the survival of other species or on ecosystem function are disproportionately large in relation to their abundance are called "keystone" species (Westman 1990). By definition, there is no redundancy in the critical function of a keystone species. For instance, Brown and Heske (1990), in an experimental study at the Chihuahuan-Sonoran Desert ecotone in Arizona, found that without kangaroo rats (*Dipodomys* spp.) a shrub steppe quickly changed to grassland. The digging of these rodents apparently favors establishment of shrub seedlings. Without them, grass competitively squeezes out shrubs. Despite the importance of keystone species to management, knowledge of them is scant, another deficiency we could begin to correct.

Policies calling for removal of all exotic, alien, or introduced species appear to rest on the old notion of a totally interdependent

community in which any change in the abundance of any species is a threat to the entire community (the "rivet" hypothesis). However, wildland communities receive continuous new arrivals, yet the adjustments usually do not result in a net loss of species. In fact, plant species richness of the California annual grasslands is probably much higher today than it was prior to coming of European man (Johnson and Mayeaux 1992). Most communities do not consist of highly co-evolved species pairs, but exhibit some substitutability by species within groups (Westman 1990). This is not to deny mutualism and the existence of keystone or critical link species, but rather to acknowledge that not all species play those roles. We need to differentiate between the exotics to worry about and those that are of less worry, based on what they do.

Critical link species are those that play a vital role in ecosystem function, regardless of their biomass, place in a food web, or possible role as a keystone species (Westman 1990). An example of a group of critical link species on rangelands is mycorrhizal fungi. These organisms exchange carbon fixed by green plants for enhanced uptake of phosphorus. Their absence may severely inhibit recovery of about 90% of the green plants that depend on them. For instance, Wicklow-Howard (1989) has shown that repeated fires promoted by cheatgrass in former sagebrush steppe in the Intermountain West can lead to extinction of mycorrhizae over vast areas and impede reestablishment of shrubs and perennial grasses. Identifying who these critical link species are and finding out whether substitutable species exist is important in setting policy and acting on it. We thus need to get on with the task of identifying critical link species for all kinds of rangelands.

The concepts of minimal and configurational structure (Pickett et al. 1989) are also worth considering. Species composition (configurational structure) within functional groups (minimal structure) may vary widely without collapse of biotic communities or ecosystems. Stennhauser (1991) has recently demonstrated how qualitatively defining stability as the maintenance of minimal structure is a practical way to approach assessment of ecosystem change.

If you agree with Walker (1989, 1992), and me, that it is more important to preserve processes than all organisms when functional redundancy can be demonstrated, we also need to discover guild or modular structure. Although Hawkins and MacMahon (1989) and Simberloff and Dayan (1991) have outlined the difficulties of tightly defining such organizational units, the existence of groups of functionally related species within communities having stronger interactions with each other than with the larger food web is undeniable and of great importance for land management (Westman 1990). Grabherr (1989) gives us an example of how we could proceed to identify keystone species, modules, guilds, or functional groups on rangelands.

If sustainable development is to be realized, we need to find ways that natural resources can be both used and maintained. Finding balancing points would be easier if we better understood the role species play in ecosystem functioning. If we can come to identify functional groups, minimal structure, modules, guilds, keystone species, critical link species, and exotics to be worried or not worried about, the older, overly simplistic, species-only notions of biodiversity will be replaced and we will gain a larger role in setting policies on use of rangelands.

Recommendations

Unfortunately, policy makers have quickly turned what were academic working concepts about biodiversity into packaging buzzwords to fund politically popular programs. The evolving understanding of biodiversity being built by researchers was thus prematurely uncoupled from strong science (Redford and Sander-son 1991). We have land managers trying to implement actions

based on fuzzy definitions, loosely worded objectives and inadequate methods of measurement and monitoring because a concerned and impatient public is breathing down their necks.

We have barely begun the tasks of quantifying biodiversity of rangelands and planning management to accommodate these values. Biodiversity is now being viewed from many new perspectives. These expanding views are driving needs for further inventories and monitoring efforts and research beyond the livestock grazing focus the range profession has stressed in the past.

Livestock grazing of any kind does not automatically reduce biodiversity. There are numerous examples of livestock use enhancing retention of some rare species, richness, and equitability of species diversities in communities and over landscapes. Although grazing is an important activity influencing ecosystem structure and function, other tendencies such as global climatic changes and biological invasions may soon overwhelm the influences of livestock in changing the biosphere.

Since it will be tactically impossible to save all species and not all species are equally valuable in maintenance of essential ecosystem states and functions, we should begin a concerted effort to identify which organisms are most important (e.g., keystone and critical link species) and focus our attention to protecting them and the modules, guilds, and functional groups in which they occur. Similarly, not all invading species are equal threats.

Passive, solely protective management of a small fraction of a countryside will not automatically result in a return to earlier conditions there, such as might have existed before the Ages of Discovery and Industrialization, and preservation of a major fraction of the species, genotypes, communities, or ecosystems. Mankind now dominates, either directly or indirectly, every part of the globe. Most biodiversity exists in human-managed areas (Pimentel et al. 1992). The inertia of changes we have already set in motion is too powerful to deny. The only rational response is for us to now recognize and understand these inevitable changes and ameliorate them where possible. We must learn to use lands in a sustainable way such that most biodiversity is also retained (Olsson et al. 1991). The basic guideline should be to prevent the loss of the inherent capacity of the land to produce and so maintain mankind's options for the future.

Literature Cited

- Allen, T.F.H., and T.W. Hoekstra. 1992. Toward a unified ecology. Columbia Univ. Press, N.Y.
- Andrew, M.H. 1988. Grazing impact in relation to livestock water points. *Trends Res. Ecol. Evol.* 3:336-339.
- Anon. 1991a. Threats to biodiversity in the United States. U.S. Environ. Protection Agency, Washington, D.C.
- Anon. 1991b. Biological diversity on Federal Lands: A report of a Keystone policy dialogue. Keystone Center, Keystone, Colo.
- Archer, S.A., and F.M. Smelns. 1991. Ecosystem-level processes. p. 109-139. In: R.K. Heitschmidt and J.W. Stuth (eds.) *Grazing management: An ecological perspective*. Timber Press, Portland, Ore.
- Archer, S., M.G. Garrett, and J.K. Detling. 1987. Rates of vegetation change associated with prairie dog (*Cynomys ludovicianus*) grazing in North American mixed-grass prairie. *Vegetatio* 72:159-166.
- Bahre, C.J. 1991. A legacy of change: Historic human impact on vegetation in southern Arizona. Univ. Arizona Press, Tucson.
- Belsky, A.J., R.G. Amundson, J.M. Duxberry, S.J. Richa, A.R. Ali, and S.M. Mwonga. 1989. The effects of trees on their physical, chemical, and biological environments in a semi-arid savanna in Kenya. *J. Appl. Ecol.* 26:1005-1024.
- Berry, K.H. 1978. Livestock grazing and the desert tortoise. *Trans. North American Wildlife and Natural Resources Conf.* 43:505-519.
- Blackburn, W.H. 1983. Influence of brush control on hydrologic characteristics of range watersheds. p. 73-88. In: K.C. McDaniel (ed.) *Proc. Brush Manage. Symp.* Texas Tech Press, Lubbock.
- Blockstein, D.C. 1990. Toward a federal plan for biological diversity. *J. Forest.* 88:15-19.

- Boeklen, W.J., and N.J. Gotelli. 1984. Island biogeographic theory and conservation practice; species-area or specious area relationships? *Biol. Conserv.* 29:63-80.
- Bonham, C.D., and A. Lerwick. 1976. Vegetation changes induced by prairie dogs on shortgrass range. *J. Range Manage.* 29:221-225.
- Bostick, V. 1990. The desert tortoise in relation to cattle grazing. *Rangelands*. 12:149-151.
- Brown, J.H., and E.J. Heske. 1990. Control of a desert-grassland transition by a keystone rodent guild. *Science* 250:1705-1707.
- Brussard, P.F. 1991. The role of ecology in biological conservation. *Ecol. Appl.* 1:6-12.
- Burton, P.J., A.C. Balesky, L.P. Coward, S.G. Cumming, and D.D. Kneeshaw. 1992. The value of managing for biodiversity. *Forest. Chronicle*. 68:225-237.
- Carlson, D.H., T.L. Thurow, R.W. Knight, and R.K. Heitschmidt. 1990. Effect of honey mesquite on the water balance of Texas Rolling Plains rangeland. *J. Range Manage.* 43:491-496.
- Chapin, F.S. III, E-D Schulze, and H.A. Mooney. 1992. Biodiversity and ecosystem processes. *Trends Res. Ecol. Evol.* 7:107-108.
- Christensen, N.L., and R.K. Peet. 1984. Measures of natural diversity. p. 43-538. *In*: J.L. Cooley and J.H. Cooley (eds.) *Natural diversity in forest ecosystems*. Proc. Workshop. Univ. Georgia, Athens.
- Cody, M.L. 1989. Growth form diversity and community structure in desert plants. *J. Arid. Environ.* 17:199-209.
- Collins, S.L., and S.C. Barber. 1985. Effects of disturbance on diversity in mixed-grass prairie. *Vegetatio* 64:87-94.
- Coppock, D.L., J.K. Detling, J.E. Ellis, and M.I. Dyer. 1983. Plant-herbivore interactions in a North American mixed grass prairie. I. Effects of blacktailed prairie dogs on intra-seasonal aboveground plant biomass, nutrient dynamics and plant species diversity. *Oecologia* 56:1-9.
- Cousins, S.H. 1991. Species diversity measurement: choosing the right index. *Trends Research Ecol. Evol.* 6:190-192.
- DePuit, E.J. 1984. Potential topsoiling strategies for enhancement of vegetation diversity on mined lands. *In*: Third Bien. Symp. on Surface Coal Mine Reclamation on the Great Plains, Billings, Mont.
- diCastri, F. 1991. Ecosystem evolution and global change. p. 189-218. *In*: O.T. Solbrig and G. Nicolis (eds.) *Perspectives in biological complexity*. Int. Union Biol. Sci., Paris, France.
- Dunn, C.P., and C. Loehle. 1988. Species-area parameter estimation: testing the null model of lack of relationship. *J. Biogeogr.* 15:721-728.
- Ehrlich, P.R., and A.H. Ehrlich. 1981. *Extinction: The causes and consequences of the disappearance of species*. Random House, N.Y.
- Ehrlich, P.R., and E.O. Wilson. 1991. Biodiversity studies: Science and policy. *Science* 253:758-762.
- Ellenberg, H. 1988. Floristic changes due to nitrogen deposition in central Europe. *In*: J. Nilsson and P. Grennfelt (eds.) *Critical loads for sulfur and nitrogen*. Nordic Council of Ministers.
- Frost, L.C. 1981. The study of *Ranunculus ophioglossifolius* and its successful conservation in the Badgeworth Reserve, Gloucestershire. p. 481-489. *In*: H. Synge (ed.) *Biological aspects of rare plant conservation*. John Wiley and Sons, N.Y.
- Fullbright, T.E., and S.L. Beasom. 1987. Long-term effects of mechanical shrub treatment on white-tailed deer browse. *Wildl. Soc. Bull.* 15:560-564.
- Gomez-Pompa, A., and A. Kaus. 1992. Taming the wilderness myth. *BioScience* 42:271-279.
- Grabherr, R. 1989. On community structure in high alpine grasslands. *Vegetatio* 83:223-227.
- Graul, W.D. 1973. Adaptive aspects of the mountain plover social system. *Living Bird* 12:69-94.
- Graul, W.D. 1975. Breeding biology of the mountain plover. *Wilson Bull.* 87:6-31.
- Grumbine, E. 1990. Protecting biological diversity through the greater ecosystem concept. *Natural Areas J.* 10:114-120.
- Hansson, L., and P. Angelstam. 1991. Landscape ecology as a theoretical basis for nature conservation. *Landscape Ecol.* 5:199-201.
- Hansen, R.M., and I.K. Gold. 1977. Blacktailed prairie dogs, desert cottontails and cattle trophic relations on shortgrass prairie. *J. Range Manage.* 30:210-214.
- Hatton, T.J., S.L. Durham, and N.E. West. 1986. Estimating components of plant community diversity on mined lands. p. 102-105. *In*: D. Williams and S.E. Fisher (eds.) *Proc. Annu. Meet. Amer. Soc. for Surface Mining and Reclam.* Denver, Colo.
- Hatton, T.J., and N.E. West. 1987. Early seral trends in plant communities on a surface coal mine in southwestern Wyoming. *Vegetatio* 73:21-29.
- Hawkins, C.H., and J.A. MacMahon. 1989. Guilds: the multiple meanings of a concept. *Annu. Rev. Entomol.* 34:423-451.
- Hill, M.O. 1973. Diversity and evenness: a unifying notation and its consequences. *Ecology* 54:427-432.
- Holden, C. 1992. A call for action on the Alala. *Science* 256:1136.
- Holdgate, M.W. 1991. The environment of tomorrow. *Environment* 33:15-20, 40-42.
- Holsinger, K.E., and L.E. Gottlieb. 1991. Conservation of rare and endangered plants: Principles and prospects. p. 195-208. *In*: D.A. Falk and K.E. Holsinger (eds.) *Genetics and conservation of rare plants*. Oxford Univ. Press, N.Y.
- Hoover, S.R., and A.J. Parker. 1991. Spatial components of biotic diversity in landscapes of Georgia. *Landscape Ecol.* 5:125-136.
- Howarth, G. 1983. Classical biocontrol: pancea or pandora's box. *Proc. Hawaiian Ent. Soc.* 24:239-244.
- Hunsaker, C.T., and D.E. Carpenter (eds.) 1990. Environmental monitoring and assessment program: Ecological indicators. Environ. Prot. Agency/600/3-90/060. Office of Research and Development, Research Triangle Park, N.C.
- Hutchings, M.J. 1987. The population biology of the early spider orchid. *Opheys spregodes* Mill. I.A. demographic study from 1975-1984. *J. Ecol.* 75:711-727.
- Huxtable, N. 1992. The desert tortoise. *Rangelands* 14:33-34.
- Johnson, R.R., L.T. Haight, M.M. Riffey, and J.M. Simpson. 1980. Brushland/steppe land communities. p. 91-112. *In*: Management of Western forests and grasslands for nongame birds. USDA, Forest Serv., Intermount. Forest and Range Exp. Sta., Ogden, Ut. Gen. Tech. Rep. INT-86.
- Johnson, H.B., and H.S. Mayeaux. 1992. A view on species additions and deletions and the balance of nature. *J. Range Manage.* 45:322-333.
- Kirby, P.E. 1984. Natural diversity requirements in environmental legislation affecting national forest planning (except the National Forest Management Act). p. 11-20. *In*: J.L. Cooley and J.H. Cooley (eds.) *Natural diversity in forest ecosystems*. Proc. Workshop, Univ. Georgia, Athens.
- Krabbenhoft, K., D. Kirby, M. Biondini, and D. Nilson. 1991. Plant diversity patterns on North Dakota mined lands. p. 555-561. *In*: W.R. Oaks and J. Bowden (eds.) *Proc. Nat. Meeting, Amer. Soc. Surface Mining and Reclam.* Princeton, W.V.
- Laycock, W.A. 1980. What is successful reclamation? A look at the concepts of adaptability, productivity, cover, and diversity of seeded species. *Proc. Northwest Colorado Land Reclam. Seminar II*. Steamboat Springs, Colo.
- Laycock, W.A. 1992. Implications of grazing vs no grazing on today's rangelands. *Proc. Symp. on Ecological Implications of Livestock Herbivory in the West*. Soc. Range Manage., Denver. (In Press).
- Leachman, B., and B. Osmundson. 1990. Status of the mountain plover: A literature review. U.S. Fish and Wildlife Serv., Golden, Colo.
- Lewis, C.E., B.F. Swindel, and G.W. Tanner. 1988. Species diversity and diversity profiles: Concept, measurement, and application to timber and range management. *J. Range Manage.* 41:466-469.
- Ludwig, J.A., and J.F. Reynolds. 1988. *Statistical ecology: A primer on methods and computing*. John Wiley and Sons, N.Y.
- Magurran, A.E. 1988. *Ecological diversity and its measurement*. Princeton Univ. Press, Princeton, N.J.
- Mann, C.C. 1991. Extinction: Are ecologists crying wolf? *Science* 253:736-78.
- McMinn, James W. 1991. Biological diversity research: an analysis. Gen. Tech. Rep. SE-71. Asheville, N.C. USDA, Forest Serv., Southeastern Forest Exp. Sta.
- McNeely, J.A., K.R. Miller, W.V. Reid, R.A. Mittermeier, and T.B. Werner. 1989. *Conserving the World's Biological Diversity*. Int. Union for Conserv. of Nature, Gland, Switzerland.
- Meuggler, W.F. 1984. Diversity of western rangelands. p. 211-217. *In*: J.L. Cooley and J.H. Cooley (eds.) *Natural Diversity in Forest Ecosystems*. Proc. Workshop, Univ. Georgia, Athens, Ga.
- Milchunas, D.G., O.E. Sala, and W.K. Lauenroth. 1988. A generalized model of the effects of grazing by large herbivores on grassland community structure. *Amer. Natur.* 132:87-106.
- Moffat, A.S. 1992. Does global change threaten the world food supply? *Science* 256:1140-1141.
- Naveh, Z., and R.H. Whittaker. 1979. Structural and floristic diversity of shrublands and woodlands in northern Israel and other Mediterranean Areas. *Vegetatio* 41:171-190.
- Nielsen, R.P., G.A. King, R.L. DeVelle, J. Lenihan, D. Marks, J. Dolph, B. Campbell, and G. Glick. 1989. Sensitivity of ecological landscapes and regions to global climate change. U.S. Environ. Prot. Agency, EPA/600/3-89/73. Corvallis, Ore.
- Newmark, W.D. 1987. A landbridge perspective on mammalian extinctions in western North American parks. *Nature* 325:430-432.

- Noss, R.F. 1990. Indicators for monitoring biodiversity: A hierarchical approach. *Conserv. Biol.* 4:355-364.
- Olsson, G.A. L-E. Liljelund, and L. Hedlund. 1991. A strategy for the conservation of biodiversity. *Ambio* 20:269-270.
- Palmer, M.W. 1990. The estimation of species richness by extrapolation. *Ecology* 71:1195-1198.
- Pechmann, J.K., D.E. Scott, R.D. Semlitsch, J.P. Caldwell, L.J. Velt, and J.W. Gibbons. 1991. Declining amphibian populations: The problem of separating human impacts from natural fluctuations. *Science* 253:892-895.
- Peet, R.K. 1974. The measurement of species diversity. *Annu. Rev. Ecol. System.* 5:285-307.
- Perry, D.A., M.P. Amaranthus, J.G. Borchers, and R.E. Brainerd. 1989. Bootstrapping in ecosystems. *BioScience* 39:230-237.
- Peters, R.H. 1991. A critique for ecology. Cambridge Univ. Press.
- Peters, R.L., and J.D.S. Darling. 1985. The greenhouse effect and nature reserves. *BioScience* 35:707-717.
- Pickett, S.T.A., J. Kolasa, J. Armesto, and S. Collins. 1989. The ecological concept of disturbance and its expression at various hierarchical levels. *Oikos* 54:129-136.
- Pickup, G. 1989. Remote sensing of landscape processes. p. 221-247. In: R.J. Hobbs and H.A. Mooney (eds.). *Remote sensing of biosphere functioning*. Springer-Verlag, N.Y.
- Pielou, E.C. 1966. The measurement of diversity in different types of biological collections. *J. Theoret. Biol.* 13:131-144.
- Pimentel, D., U. Stachow, D.A. Takas, H.W. Brubaker, A.R. Dumas, J.J. Meaney, J.A.S. O'Neil, D.E. Onsi, and D.B. Corzilius. 1992. Conserving biological diversity in agricultural/forestry systems. *BioScience* 42:354-362.
- Pimm, S.L., and J.L. Gittleman. 1992. Biological diversity: Where is it? *Science* 255-940.
- Pimm, S.L. 1991. The balance of Nature? Ecological issues in the conservation of species and communities. Univ. Chicago Press, Chicago.
- Quinn, J.F., and G.R. Robinson. 1987. The effects of experimental subdivision on flowering plant diversity in a California annual grassland. *J. Ecol.* 75:837-856.
- Redford, K.H., and S.E. Sanderson. 1992. The brief, barren marriage of biodiversity and sustainability. *Bull. Ecol. Soc. Amer.* 73:36-39.
- Romme, W.H., and D.H. Knight. 1982. Landscape diversity: The concept applied to Yellowstone Park. *BioScience* 32:644-670.
- Routledge, R.D. 1979. Diversity indices: Which ones are admissible? *J. Theoret. Biol.* 76:503-515.
- Routledge, R.D. 1984. Estimating ecological components of diversity. *Oikos* 42:23-29.
- Ryder, R.A. 1980. Effects of grazing on bird habitats. p. 51-66. In: R.M. DeGraff and N.G. Tilghmann (eds.). *Management of Western forests and grasslands for nongame birds*. U.S. Forest Serv., Gen. Tech. Rep. INT-86. Intermount. Res. Sta., Ogden, Ut.
- Salwasser, H. 1991. Diversity: new perspectives for sustaining diversity in U.S. Nation Forest Ecosystems. *Conserv. Biol.* 5:567-569.
- Scott, J.M., B. Csuti, J.D. Jacobi, and J.E. Estes. 1987. Species richness: A geographic approach to protecting future biological diversity. *BioScience* 37:782-788.
- Scott, J.M., B. Csuti, K. Smith, J.E. Estes and S. Caicco. 1988. Beyond endangered species: an integrated strategy for the preservation of biological diversity. *Endangered Species Update*. 5:43-48.
- Scott, J.M., B. Csuti, J.E. Estes, and H. Anderson. 1989. Status assessment of biodiversity protection. *Conserv. Biol.* 3:85-87.
- Shmida, A., and M.V. Wilson. 1985. Biological determinants of species diversity. *J. Biogeogr.* 12:1-20.
- Simberloff, D., and T. Dayan. 1991. The guild concept and the structure of ecological communities. *Ann. Rev. Ecol. System.* 22.
- Sidle, R.C., and A.N. Sharpely. 1991. Cumulative effects of land management on soil and water resources: an overview. *J. Environ. Qual.* 20:1-3.
- Simpson, E.H. 1949. Measurement of diversity. *Nature* 163:688.
- Solbrig, O.T. 1992. The IUBS-UNESCO Program for research in biodiversity. *Ecol. Appl.* 2:131-138.
- Stennhauser, E.B. 1991. The concept of stability in connection with the gallery forests of the Chaco region. *Vegetatio* 94:1-13.
- Stockwell, S.S. 1990. U.S. national policies related to maintaining biological diversity in forests. p. 291-295. In: M.L. Hunter Jr. *Wildlife, Forests and Forestry*. Prentice-Hall, Englewood Cliffs, N.J.
- Stolzenburg, W. 1992. The Heritage Network: detectives of diversity. *Nature Conservancy*. Jan. Feb., p. 23-27.
- Turner, S.J., R.V. O'Neill, W. Conley, M.R. Conley, and H.C. Humphries. 1990. Pattern and scale: Statistics for landscape ecology. p. 17-49. In: M.G. Turner and R.H. Gardner (eds.). *Quantitative methods in landscape ecology*. Ecol. Studies 82, Springer-Verlag, N.Y.
- Van Devender, T.R. 1990. Late quaternary vegetation and climate of the Sonoran Desert, United States and Mexico. p. 134-165. In: J.L. Betancourt, T.R. Van Devender and P.S. Martin (eds.) *Packrat middens: The last 40,000 years of biotic change*. Univ. Arizona Press, Tucson.
- Wainwright, S.J., and H.W. Woolhouse. 1975. Physiological mechanisms of heavy metal tolerance in plants. p. 231-257. In: M.J. Chadwick and G.T. Goodman (eds.). *The ecology of resource degradation and renewal*. Blackwell Sci. Publ., Oxford.
- Walker, B. 1989. Diversity and stability in ecosystem conservation. p. 121-130. In: D. Western and M. Pearl (eds.). *Conservation for the Twenty-first Century*. Oxford Univ. Press, N.Y.
- Walker, B.H. 1992. Biological and ecological redundancy. *Conserv. Biol.* 6:18-23.
- Waser, N.M., and M.V. Price. 1981. Effects of grazing on diversity of annual plants in the Sonoran Desert. *Oecologia* 50:407-411.
- Wayne, P.M., and F.A. Bazzaz. 1991. Assessing diversity in plant communities: The importance of within-species variation. *Trends in Ecol. Evol.* 6:400-401.
- West, N.E. 1991a. Benchmarks for rangeland management and environmental quality. p. 30-44. In: L.F. James, J.O. Evans, M.H. Ralphs, and R.D. Child (eds.) *Noxious range weeds*. Westview Press, Boulder, Colo.
- West, N.E. 1991b. Nutrient cycling in soils of semiarid and arid regions. p. 295-232. In: Skujins (ed.). *Semiarid lands and deserts: Soil resource and reclamation*. Marcel Dekker, Inc., N.Y.
- West, N.E., and G.A. Reese. 1991. Comparison of some methods for collecting and analyzing data on aboveground net production and diversity of herbaceous vegetation in a northern Utah sub-alpine context. *Vegetatio* 96:145-163.
- Westman, W.A. 1977. How much are nature's services worth? *Science* 197:960-964.
- Westman, W.A. 1990. Managing for biodiversity. *BioScience* 40:26-33.
- Whittaker, R.H. 1965. Dominance and diversity in land plant communities. *Science* 147:250-260.
- Wicklow-Howard, M. 1989. The occurrence of vesicular-arbuscular mycorrhizae in burned areas on the Snake River Birds of Prey Area, Idaho. *Mycotaxon* 34:253-257.
- Williams, C.B. 1964. Patterns in the balance of Nature and related problems in quantitative ecology. Academic Press, N.Y.
- Williams, B.L., and B.G. Marcot. 1991. Use of biodiversity indicators for analyzing and managing forest landscapes. *Trans. N.A. Wildl. & Nat. Res. Conf.* 56:613-627.
- Yorks, T.P., N.E. West, and K.M. Capels. 1992. Vegetation change in desert shrublands of western Utah's Pine Valley, 1933-1989. *J. Range Manage.* 45:569-578.

Dependence of 3 Nebraska Sandhills warm-season grasses on vesicular-arbuscular mycorrhizae

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Abstract

Vesicular-arbuscular mycorrhizae (VAM) are rare or absent in actively eroding soils of the Sandhills. The objective of this study was to determine if 3 major Sandhills warm-season grasses used in reseeded eroded Sandhills sites are highly mycorrhizal dependent, and evaluate the response of VAM at different phosphorus (P) levels. In 2 greenhouse experiments, sand bluestem [*Andropogon gerardii* var. *paucipilus* (Nash) Fern.], switchgrass (*Panicum virgatum* L.), and prairie sandreed [*Calamovilfa longifolia* (Hook) Scribn.] were grown in steam-sterilized sand in pots and inoculated with either indigenous Sandhills VAM, *Glomus deserticola*, or noninoculated. In the second experiment, VAM inoculated and control plants were treated with 5 P levels ranging from 5.4 to 27.0 mg P pot⁻¹. Increasing levels of P fertilizer caused an initial increase, then dramatic decrease, in percentage colonization by *Glomus deserticola* but had no effect on percentage colonization by indigenous Sandhills VAM. Mycorrhizal inoculated plants had a greater number of tillers, greater shoot weight, root weight, tissue P concentration and percentage P recovered, and a lower root/shoot ratio and P efficiency than noninoculated plants. Noninoculated sand bluestem had significantly lower shoot P concentration but greater P efficiency over all P levels than any other grass-VAM treatment combination. Phosphorus fertilizer and VAM effects were often complementary at P levels up to 16.2 to 21.6 mg P pot⁻¹, with no change or a decrease in plant responses at higher P levels. These 3 major Sandhills warm-season grasses were highly mycorrhizal dependent. Successful reestablishment of these on eroded sites in the Sandhills may be greatly improved if soil reinoculation with VAM occurred prior to revegetation.

Key Words: phosphorus recovery, phosphorus-use efficiency, revegetation, grass seedlings, tillering

The Nebraska Sandhills comprise the largest continuous expanse of Tallgrass Prairie in the Great Plains, covering about 52,000 km² in northcentral and western Nebraska (Seevers et al. 1975). Upland soils of the Sandhills are composed primarily of fine sand of high erosion potential and low organic matter content, water-holding capacity, and available nutrients. Available P may be the soil nutrient that has the greatest influence on the vegetative composition of range sites in the Sandhills (Burzlaff 1962).

Many areas of the Sandhills have experienced excessive erosion as a result of overgrazing and the abandonment of farming operations on center pivot irrigation sites (Kocher and Stubbendieck 1986). Vesicular-arbuscular mycorrhizae (VAM) are abundant in

undisturbed Sandhills soils, but rare or absent in actively eroding soils (Roder 1985, Reece et al. 1987). Reseeding these sites with native and introduced warm- and cool-season grasses has often been unsuccessful (King et al. 1989). Lack of success in seedling establishment in eroding Sandhills soils may be a result of very low levels of VAM fungi present in the soils at the time of seeding (Reece et al. 1987, Reeves et al. 1979).

Vesicular-arbuscular mycorrhizae readily colonize the roots of prairie grasses (Hetrick et al. 1988). Warm-season grasses display a high degree of dependence on mycorrhizae and are often unable to survive without them (Hetrick et al. 1988, 1989; Trappe 1981, Bethlenfalvay et al. 1984). Hetrick et al. (1989) reported that inoculation of sterilized prairie soils with mycorrhizae resulted in a 7 to 70 fold increase in big bluestem (*Andropogon gerardii* Vitman) seedling biomass. Plant response to mineral nutrients is often highly dependent on VAM. Hetrick et al. (1989) concluded that when mycorrhizae are present, N is the most limiting nutrient in most rangeland soils. However, in the absence of mycorrhizae, P is most limiting and no response to N is observed unless sufficient P is also applied.

Understanding the symbiotic relationship between VAM fungi and the native warm-season grasses of the Sandhills is essential for effective seeding of disturbed Sandhills soils. The purpose of this research was to examine relationships between VAM fungi and 3 native Sandhills warm-season grasses by (1) comparing the growth, P-uptake and P-use efficiency of these grasses with indigenous and introduced VAM fungi, and (2) evaluating the response at different P levels.

Materials and Methods

Inoculum Source and Treatments

Two greenhouse studies were conducted utilizing indigenous Sandhills VAM fungi from the Nebraska Sandhills, and the introduced VAM fungus *Glomus deserticola*. *Glomus deserticola* was chosen for comparison with indigenous Sandhills VAM because it is not native to the Sandhills but is found in more arid environments in the western U.S. Inoculum for the Sandhills VAM treatment was prepared by collecting large blocks of sod containing root material of native Sandhills grasses from the top 30 cm of a Valentine fine sand (fine sandy mixed mesic Typic Ustipsamment) at a rolling sands range site at the University of Nebraska's Gudmundsen Sandhills Laboratory, located in Grant County, 12 km northeast of Whitman, Neb. Soil at the collection site is low in mineral nutrients, especially in available P. Soil properties for the top 30 cm were: 0.81% OM, 0.04% total N (Kjeldahl), 5.7 mg kg⁻¹ available P (Bray and Kurtz #1), and a pH (1:1 soil-water ratio) of 6.45. Major components of the plant community at the collection site included sand bluestem [*Andropogon gerardii* var. *paucipilus*

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(Nash) Fern.), prairie sandreed [*Calamovilfa longifolia* (Hook) Scribn.], switchgrass (*Panicum virgatum* L.), little bluestem [*Schizachyrium scoparium* (Michx.) Nash.], and needleandthread (*Stipa comata* Trin. and Rupr.). Mycorrhizae native to the collection site and present in the Sandhills VAM inoculum included *Glomus fasciculatum*, *G. mosseae*, *G. macrocarpum*, *Sclerocystis dugesia*, and *Gigaspora* spp. Inoculum for the *G. deserticola* treatment was collected from sorghum (*Sorghum bicolor* L.) pot cultures previously inoculated with the fungus and maintained in a 20–27° C greenhouse.

Inoculum for both VAM treatments was prepared by removing the plant tops and thoroughly washing the roots to remove all soil and adhering spores, and refrigerating at 4° C prior to use. Half the root mass from both inoculum sources was combined and autoclaved at 100° C for 1-hour to sterilize to serve as a control treatment. Roots were cut into 2-cm pieces and 14 g (fresh weight) of root material was added to each pot in a layer 2.5 cm below the soil surface. Filtered washings (20 μ m sieve) from a mixed sample of all inocula was added to each pot to insure all pots developed a similar microflora except for mycorrhizal fungi (Linderman and Hendrix 1982). Coarse sand steamed for 1 hour to sterilize was used as the soil medium. Pots had a total volume of 1.9 L, and a depth of 25 cm.

Seeds of sand bluestem, prairie sandreed and switchgrass were surface sterilized by soaking in a 10% solution of commercial chlorine bleach for 3 minutes. Seeds were sown in each pot in excess of final density and covered with 1 cm of sterile sand. Immediately after seeding, each pot was watered thoroughly and allowed to drain freely. Seedlings emerged within 3 to 6 days and were thinned to 3 plants per pot after 1 week. Long Ashton nutrient solution (Hewitt 1966) containing all essential nutrients except P

was added to all pots weekly to insure that only P was on limited supply. Phosphorus level for the first study totaled 5.4 mg pot⁻¹ distributed over 12 applications at weekly intervals in 100-ml aliquots. Five P levels of 5.4, 10.8, 16.2, 21.6, and 27 mg pot⁻¹ were used for the second study and were applied in the same manner as in the first study. The lowest P level used in this study (5.4 mg pot⁻¹) was chosen because it represented an average P concentration found in the top 30 cm of Nebraska Sandhills rolling sands soils. Plants were watered as needed at 2- to 3-day intervals and allowed to drain freely. The photoperiod was extended to 16 hours with metal halide lamps that produced 55 W m⁻² and the greenhouse temperature was maintained between 20 and 27° C. Both experiments were 13 weeks long. The first greenhouse experiment was planted 8 October, and harvested on 6 January. The second greenhouse experiment was planted on 25 April and harvested on 24 July.

Data Collection and Analysis

At each harvest the number of tillers per pot was determined and shoots were separated from roots with the crown assigned to the shoots. Roots were washed free of soil and the shoots and half the root mass were dried separately at 70° C for 48 hours in a forced air oven and weighed. Shoots were ground in a Wiley mill to pass a 1-mm screen, dry ashed, and analyzed for total P content by the University of Nebraska-Lincoln Testing Laboratory. Phosphorus efficiency was calculated as g of shoot dry matter produced per mg P accumulated in the shoot dry weight (Elliott and Lauchli 1985, Marschner 1986).

A subsample of the fresh root mass was cleared in 10% KOH and differentially stained with 0.5% trypan blue in water (modification of Phillips and Hayman 1970). Percentage mycorrhizal coloniza-

Table 1. Percentage colonization, number of tillers per plant, shoot weight, root weight, and root/shoot ratio, of 3 Nebraska Sandhills warm-season grasses inoculated with *Glomus deserticola* or Sandhills VAM, and a noninoculated control during 2 greenhouse studies at P levels of 5.4 mg P pot⁻¹.

Mycorrhizal treatment	First study					Second study				
	Colonization	Tiller	Shoot weight	Root weight	Root/shoot ratio	Colonization	Tiller	Shoot weight	Root weight	Root/shoot ratio
	(%)		(No.) g dry matter pot ⁻¹			(%)		(No.) g dry matter pot ⁻¹		
Sand Bluestem										
Control	0	1.0	0.16	0.18	1.15	0	1.0	0.14	0.22	1.46
Sandhills	44	2.2	1.33	0.98	0.75	15	3.0	1.72	1.44	0.85
<i>G. deserticola</i>	35	1.8	1.79	1.61	0.90	43	2.7	2.73	3.24	1.20
Contrasts (P>F)										
Control vs Mycorrhizal	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.05
<i>G. deserticola</i> vs Sandhills	NS	NS	0.01	0.01	NS	0.05	NS	0.01	0.01	NS
Switchgrass										
Control	0	1.0	0.08	0.06	0.98	0	1.1	0.10	0.11	1.26
Sandhills	52	2.9	1.43	1.61	0.44	29	3.3	2.57	2.29	0.88
<i>G. deserticola</i>	44	2.8	2.58	1.72	0.69	27	3.7	3.21	3.46	1.08
Contrasts (P>F)										
Control vs Mycorrhizal	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	NS
<i>G. deserticola</i> vs Sandhills	NS	NS	0.01	0.01	NS	NS	NS	0.01	0.01	NS
Prairie Sandreed										
Control	0	1.0	0.16	0.15	1.01	0	1.1	0.31	0.32	0.89
Sandhills	35	1.2	0.61	0.70	1.20	24	1.4	1.52	1.52	1.00
<i>G. deserticola</i>	51	1.7	1.29	1.61	1.25	27	2.3	3.00	3.41	1.12
Contrasts (P>F)										
Control vs Mycorrhizal	0.01	0.01	0.01	0.01	NS	0.01	0.01	0.01	0.01	NS
<i>G. deserticola</i> vs Sandhills	0.01	0.01	0.01	0.01	NS	NS	0.01	0.01	0.01	NS

NS = Not significant at the 0.05 probability level.

tion was determined by examining 100 root intercepts of each sample at 100X magnification using the grid intersection method of Giovanetti and Mosse (1980). The 3 plants in each pot were combined for analysis and each pot was considered an experimental unit.

The first study involved a factorial arrangement of 3 grass species and 3 VAM treatments with 10 replications arranged in a completely randomized design. The second study had the additional factor of 5 P levels and 6 replications. A grass species by VAM treatment interaction was significant for most plant responses studied, so data were analyzed and presented separately for each grass. Significant mycorrhizal and P treatment main effects and interactions were analyzed using preplanned orthogonal contrasts (Steel and Torrie 1980). Contrasts for the mycorrhizal treatments compared the noninoculated control versus VAM inoculated treatments, and compared the 2 different VAM treatments. Plant responses over the 5 P levels were analyzed by regression analysis using orthogonal polynomials for significant linear, quadratic, and cubic responses (Steel and Torrie 1980).

Results and Discussion

Percentage Colonization

In the first trial, *G. deserticola* produced significantly greater colonization compared to Sandhills VAM in prairie sandreed, but there were no significant differences in percentage colonization between *G. deserticola* and Sandhills VAM in sand bluestem and switchgrass (Table 1). In the second trial, at 5.4 mg P pot⁻¹ *G. deserticola* produced greater percentage colonization than Sandhills VAM only for sand bluestem.

The VAM response to P was consistent over all 3 grasses for both VAM treatments. A significant mycorrhizal by P interactions for percentage colonization ($P < 0.05$) was detected as increased P levels produced a quadratic response in percentage to colonization by *G. deserticola*, but had no significant effect on percentage colonization by Sandhills VAM (Fig. 1). Percentage colonization by *G. deserticola* increased from the lowest P level and maximized at 8.9 mg P pot⁻¹, then decreased dramatically at higher P rates. The effect of increased P supply on the colonization rate varies markedly between VAM species (Marschner 1986). At the lowest P level the colonization rate by *G. deserticola* may have been inhibited by insufficient P for optimum growth of the fungus (Bolan et al. 1984), with maximum percentage colonization achieved at a P level that produced 85% of maximum shoot growth. The inhibitory effect of high P levels on VAM colonization has been documented (Jasper et al. 1979, Hayman 1983), but with Sandhills VAM the generally low level of colonization for all P levels may account for the lack of an inhibitory effect of high P levels on percentage colonization.

Tiller Number and Shoot Weight

At 5.4 mg P pot⁻¹, mycorrhizal inoculated plants had significantly greater tillering compared to noninoculated plants for all 3 grasses in both trials (Table 1). The degree to which plant tillering was stimulated varied with each grass-VAM species combination, resulting in a significant grass by VAM treatment interaction in both trials ($P < 0.01$). In the first trial, sand bluestem and switchgrass produced the most tillers when inoculated with Sandhills VAM, but prairie sandreed produced the most tillers with *G. deserticola*. Again, the second trial, sand bluestem produced more tillers when inoculated with Sandhills VAM, but prairie sandreed and switchgrass produced more tillers when inoculated with *G. deserticola*.

Plant tillering was stimulated by P as indicated by a significant quadratic response to increasing P levels (Fig. 2). The tillering response to P was similar for all 3 grasses and VAM treatments.

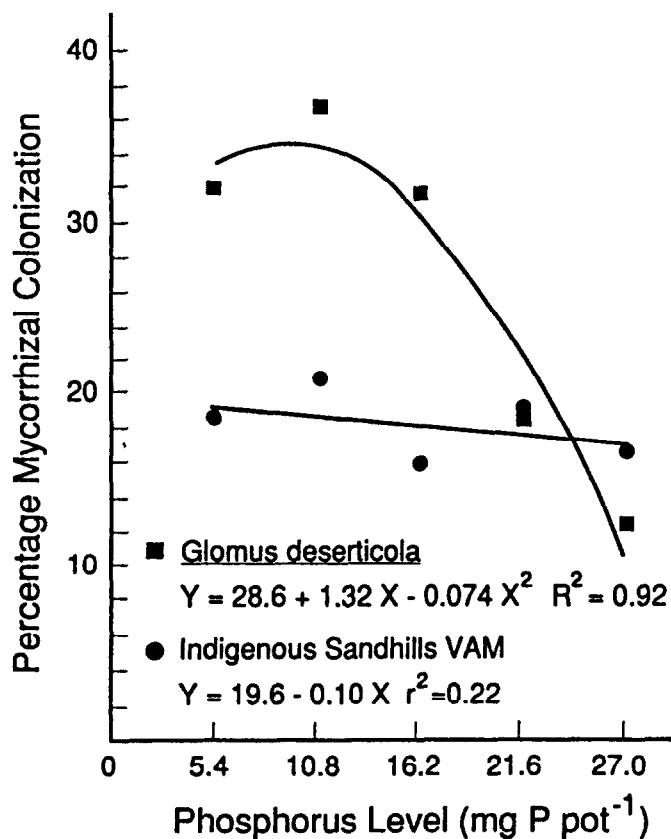


Fig. 1. Percentage colonization by *Glomus deserticola* and indigenous Sandhills VAM with increasing P levels (mg P pot⁻¹) averaged over Nebraska Sandhills warm-season grasses.

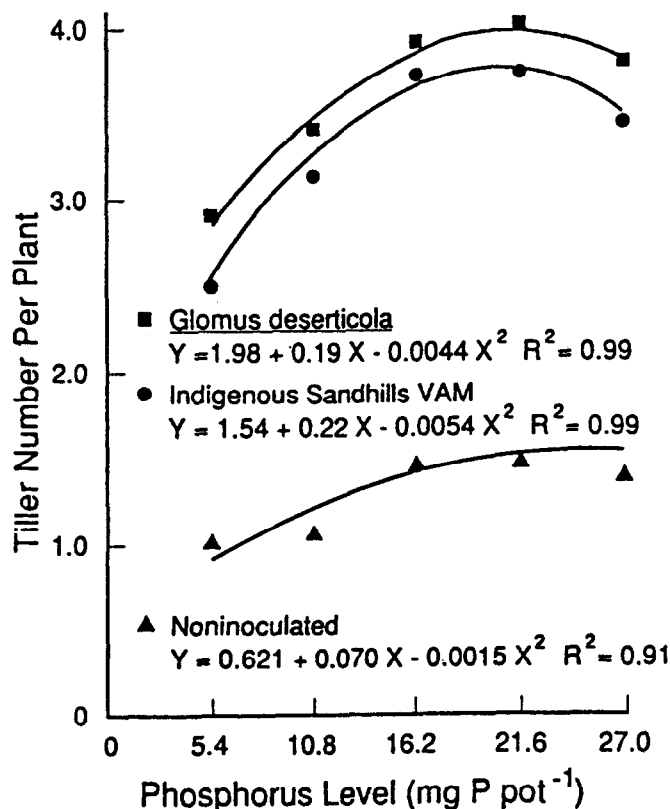


Fig. 2. Tiller numbers per plant with increasing P levels (mg P pot⁻¹) averaged over 3 Nebraska Sandhills warm-season grasses inoculated with *Glomus deserticola*, indigenous Sandhills VAM and a noninoculated control

Plants inoculated with VAM produced 2 to 3 times more tillers than noninoculated plants. Based upon the fitted regression equations, maximum tiller numbers were achieved at about 21.6 mg P pot⁻¹, with a decrease in tiller numbers at higher P levels (Fig. 2).

At the 5.4 mg P pot⁻¹ level, shoot weight of VAM-inoculated plants was significantly greater than shoot weight of noninoculated plants ($P < 0.01$), and *G. deserticola*-inoculated plants produced significantly greater shoot weights than Sandhills VAM-inoculated plants in both trials (Table 1). Different plant responses for tillering and shoot weights were confirmed by the low correlation between these 2 parameters ($r = 0.35$). Shoot weights of VAM-inoculated plants were not correlated with percentage colonization in either trial. Hayman (1983) reported there is no definite correlation between the amount of VAM colonization of the root and the enhancement of plant growth. Thus, the most infective fungi do not necessarily elicit the most positive responses.

A highly significant grass species by VAM treatment interaction occurred with shoot weight ($P < 0.01$) (Table 1). The interaction was caused by the magnitude of the response to the different VAM treatments. In both trials, *G. deserticola* increased shoot yields of all grasses over that of grasses inoculated with Sandhills VAM. The yield increase caused by *G. deserticola* was much greater for prairie sandreed than for sand bluestem and switchgrass.

A significant VAM treatment by P response for shoot weight was characterized by significantly different quadratic curves for noninoculated and VAM-inoculated plants (Fig. 3). The VAM response to P was similar for the 3 grasses. Shoot weights of the noninoculated grasses increased with increasing levels of P, with

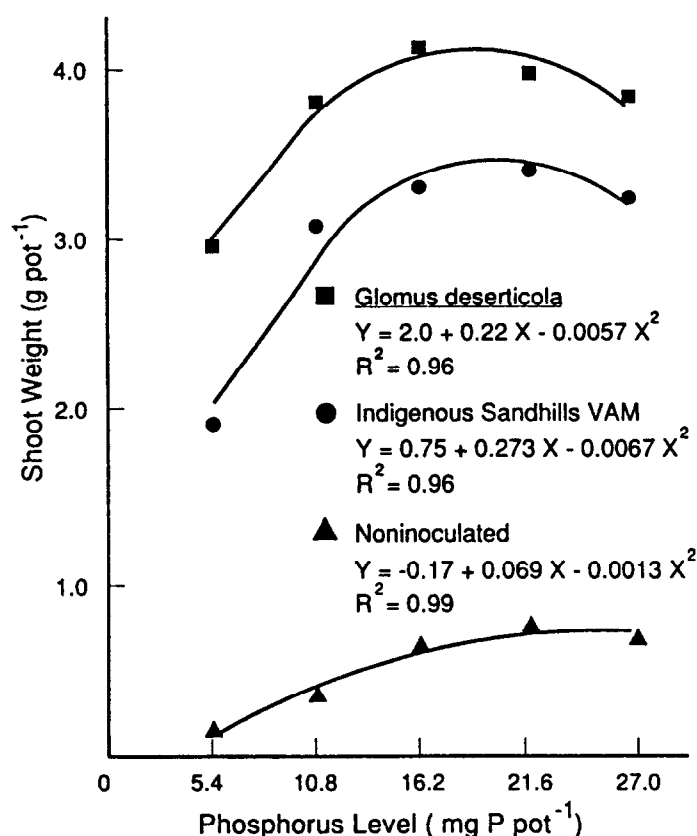


Fig. 3. Shoot weight (g pot⁻¹) with increasing P levels (mg P pot⁻¹) averaged over 3 Nebraska Sandhills warm-season grasses inoculated with *Glomus deserticola*, indigenous Sandhills VAM and a noninoculated control

maximum shoot weight at the highest P level. However, even at the highest P levels, shoot weight of noninoculated plants was less than half that of VAM-inoculated plants at the lowest P rate. Mycorrhizal-inoculated plants had large increases in shoot weight from 5.4 to 10.8 mg P pot⁻¹, followed by gradual increases and then a decline as P levels increased to 27 mg P pot⁻¹. Based upon fitted regression equations, shoot weights of *G. deserticola*-inoculated plants were greatest at 19.3 mg P pot⁻¹ with a decline in shoot weights at higher P levels; and shoot weights of Sandhills VAM-inoculated plants were greatest at 20.2 mg P pot⁻¹ with a decline in shoot weights at higher P levels.

Root Weight and Root/Shoot Ratio

Root weights of mycorrhizal-inoculated plants were greater than root weights of noninoculated plants (Table 1), and *G. deserticola*-inoculated plants had greater root weights than plants inoculated with Sandhills VAM for all 3 grasses in both trials. Root yields of prairie sandreed plants inoculated with *G. deserticola* were more than double those of plants inoculated with Sandhills VAM. Increasing P levels caused linear increases in root weight, with the response being similar over all grasses and mycorrhizal treatments.

Response of root/shoot ratio to the VAM treatments varied by species (Table 1). Root/shoot ratios of VAM-treated sand blue bluestem were less than ratios for noninoculated plants in both trials. Switchgrass root/shoot ratios were less when inoculated with VAM in trial 1. While not significant, a similar response occurred in trial 2. In prairie sandreed, the VAM treatments increased the root and shoot weights by a similar magnitude so the root/shoot ratios were not different from those of noninoculated plants. Other researchers have reported lower root/shoot ratios in mycorrhizal compared to nonmycorrhizal plants (Mosse and Hayman 1971, Hayman and Mosse 1972).

A significant quadratic response in root/shoot ratios occurred over P levels. The response was similar for all 3 grasses and mycorrhizal treatments and resulted from a sharp decline in root/shoot ratios between 5.4–16.2 mg P pot⁻¹, with no change in root/shoot ratios at higher P levels (data not shown).

Phosphorus Concentration, Recovery, and Efficiency

Mycorrhizal-inoculated plants had significantly greater shoot P concentrations over noninoculated plants in both trials ($P < 0.01$, Table 2). In general, the Sandhills VAM increased shoot P concentration more than *G. deserticola*. In the first trial, P concentration in the grass shoots followed a similar pattern as percentage colonization in which sand bluestem and switchgrass had the greatest shoot P concentration when inoculated with Sandhills VAM. Although prairie sandreed had the greatest shoot P concentration when inoculated with *G. deserticola*, it was not significantly greater than that for Sandhills VAM. In the second trial, percentage colonization and shoot P concentration followed an opposite pattern in which *G. deserticola* produced the greatest percentage colonization for all 3 grasses, but Sandhills VAM-inoculated plants produced the greatest shoot P concentration in sand bluestem and prairie sandreed.

A significant grass species by VAM treatment by P level interaction was observed for shoot P concentration ($P < 0.01$, Fig. 4). Shoot P concentrations of VAM-inoculated plants were similar for all 3 grasses over all P levels. Mycorrhizal-inoculated plants increased in shoot P concentration with increasing P rates from 5.4–21.6 mg P pot⁻¹, with maximum shoot P concentration at a P level of 23 mg P pot⁻¹, based upon fitted regression equations. However, noninoculated sand bluestem had significantly lower shoot P concentrations than noninoculated switchgrass and prairie sandreed, and responded differently to increasing P levels, which suggested that sand bluestem may have the lowest physiological P requirement. Sand bluestem showed a linear increase in

Table 2. Phosphorus concentration, percent recovery and phosphorus efficiency by shoots of 3 Nebraska Sandhills warm-season grasses inoculated with *Glomus deserticola* or Sandhills VAM or non-inoculated control for 2 greenhouse studies at P levels of 5.4 mg pot⁻¹.

Mycorrhizal treatment	First study			Second study		
	Concentration (mg P g ⁻¹ dry matter)	Recovery (%)	Efficiency (g dry matter mg P ⁻¹)	Concentration (mg P g ⁻¹ dry matter)	Recovery (%)	Efficiency (g dry matter mg P ⁻¹)
Sand Bluestem						
Control	40	1.3	2.54	40	1.2	2.50
Sandhills	100	26.2	1.01	84	26.6	1.20
<i>G. deserticola</i>	75	23.7	1.34	75	37.9	1.34
Contrasts (P>F)						
Control vs Mycorrhizal	0.01	0.01	0.01	0.01	0.01	0.01
<i>G. deserticola</i> vs Sandhills	0.01	NS	0.05	0.05	0.01	0.05
Switchgrass						
Control	52	0.6	2.04	48	0.9	2.14
Sandhills	102	25.8	0.99	76	36.0	1.34
<i>G. deserticola</i>	70	32.4	1.46	72	42.8	1.39
Contrasts (P>F)						
Control vs Mycorrhizal	0.01	0.01	0.01	0.01	0.01	0.01
<i>G. deserticola</i> vs Sandhills	0.01	NS	0.05	NS	NS	NS
Prairie Sandreed						
Control	66	17.7	1.79	53	2.5	1.99
Sandhills	98	12.7	1.04	78	21.8	1.30
<i>G. deserticola</i>	126	30.5	0.81	64	35.6	1.57
Contrasts (P>F)						
Control vs Mycorrhizal	0.01	0.01	0.01	0.01	0.01	0.01
<i>G. deserticola</i> vs Sandhills	NS	0.01	NS	0.01	0.01	NS

NS = Not significant at the 0.05 probability level.

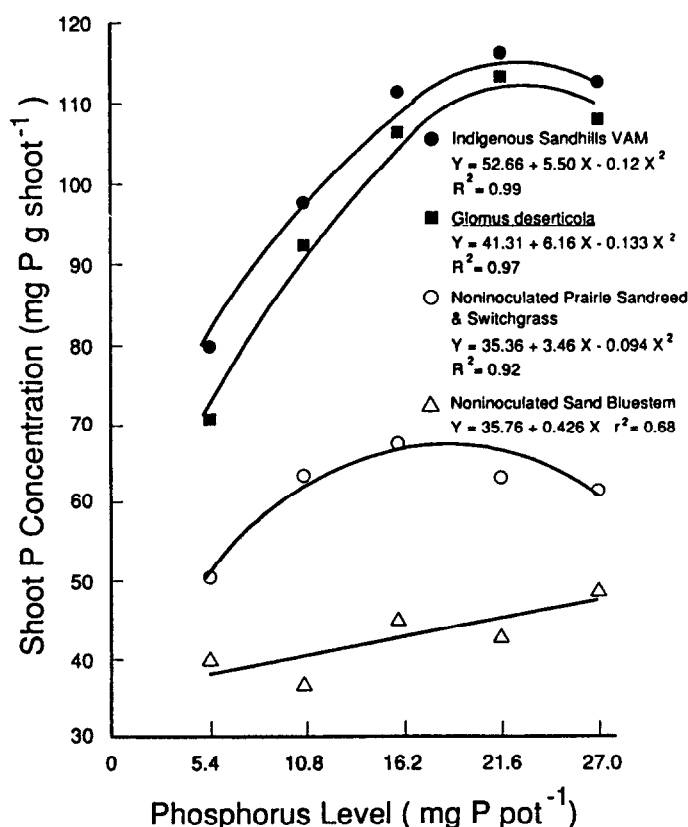


Fig. 4. Shoot P concentration (mg P g shoot⁻¹) with increasing P levels (mg P pot⁻¹) averaged over 3 Nebraska Sandhills warm-season grasses inoculated with *Glomus deserticola* or indigenous Sandhills VAM, and for noninoculated sand bluestem and an average of noninoculated prairie sandreed and switchgrass.

shoot P concentration with increasing P levels, but shoot P concentration of sand bluestem at the highest P level was similar to shoot P concentration of switchgrass and prairie sandreed at the lowest P level. Noninoculated switchgrass and prairie sandreed had similar quadratic responses to increasing P levels. Maximum shoot P concentration occurred at a P level of 21.6 mg P pot⁻¹. A 5-fold increase in the P fertilization rate failed to raise the shoot P concentration of noninoculated control plants to the same level achieved with VAM at the lowest P rate.

Mycorrhizal-inoculated plants recovered a much greater percentage P compared to noninoculated plants in both trials (Table 2). The *G. deserticola*-inoculated plants generally were more efficient in recovering applied P than Sandhills VAM-inoculated plants when averaged over all 3 grasses, but the total percentage P recovered depended on the grass species involved, resulting in significant grass species by VAM treatment interactions for both trials ($P < 0.01$). In the first trial *G. deserticola*-inoculated prairie sandreed plants recovered more P than Sandhills VAM-inoculated plants but differences in P recovery were not significantly different from the 2 VAM inocula for switchgrass and sand bluestem. A similar pattern was observed in the second trial, with mycorrhizal-inoculated plants recovering significantly more P than noninoculated plants, and *G. deserticola*-inoculated plants recovering more P than Sandhills VAM-inoculated plants for sand bluestem and prairie sandreed.

Increasing P levels resulted in a significant linear decrease in percentage P recovered by VAM-inoculated plants, but had no effect on percentage P recovered by noninoculated plants (Fig. 5). The response was similar for all 3 grasses. The rate of decline in percentage P recovered with increasing P levels was significantly different for the 2 VAM treatments in which percentage P recovered by *G. deserticola*-inoculated plants declined more rapidly than percentage P recovered by Sandhills VAM-inoculated plants.

Shoot P concentration and P efficiency (PE) varied inversely. The grass species-VAM treatment combinations with the highest shoot P concentrations had the lowest PE. Noninoculated plants

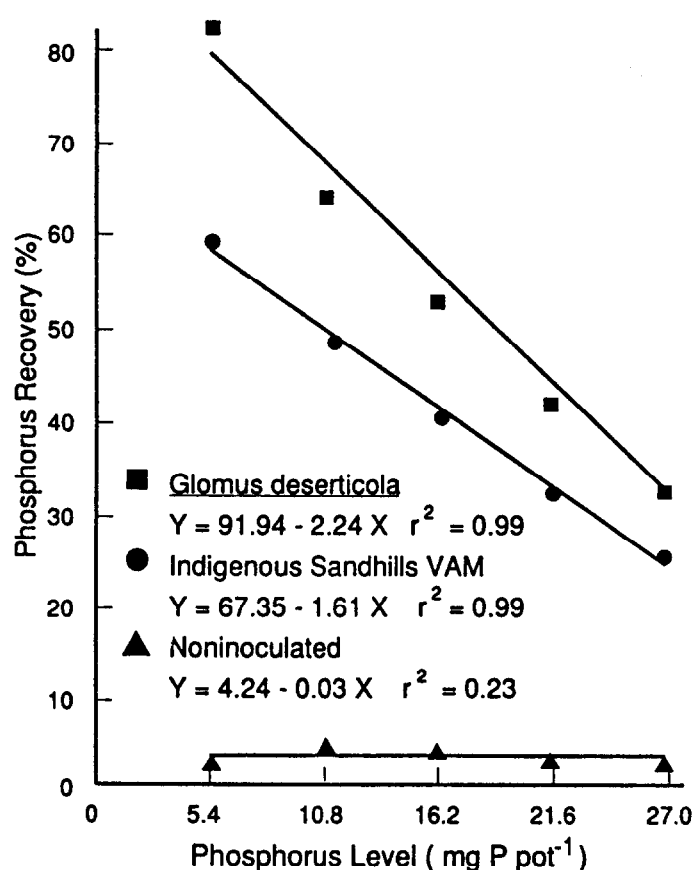


Fig. 5. Percentage P recovered with increasing P levels (mg P pot⁻¹) averaged over 3 Nebraska Sandhills warm-season grasses inoculated with *Glomus deserticola*, indigenous Sandhills VAM and a non-inoculated control.

had significantly greater PE than VAM inoculated plants, with sand bluestem being the most efficient of all 3 grasses (Table 2). In general, plants inoculated with *G. deserticola* had greater PE than plants inoculated with Sandhills VAM, but differences were significant only for sand bluestem in both trials and switchgrass in the first trial. In the second trial, differences in PE between *G. deserticola* and Sandhills VAM-inoculated plants were very small.

Increased P level resulted in a significant decrease in PE for all grasses and VAM treatments (Fig. 6). A significant VAM treatment by P level by grass species interaction occurred, resulting from a much greater PE in noninoculated sand bluestem than noninoculated prairie sandreed or switchgrass. Noninoculated sand bluestem showed a linear decline in PE with increasing P levels, but noninoculated prairie sandreed and switchgrass produced similar quadratic responses with a dramatic decline in PE up to 19 mg P pot⁻¹, then a small increase in PE at higher P levels. Elliott and Lauchli (1985) and Marschner (1986) reported that plants varied considerably among species and genotypes within species in PE, and possess both physiological and morphological characteristics that affect PE. Noninoculated sand bluestem had a greater PE than noninoculated prairie sandreed and switchgrass, or VAM-inoculated plants. Response of VAM-inoculated plants was similar for all 3 grasses over all P levels. Phosphorus efficiency of VAM-inoculated plants declined with increased P levels up to 16.2 mg P pot⁻¹, with no significant change in PE at higher P levels.

Results from this study suggest that 3 of the major climax grasses

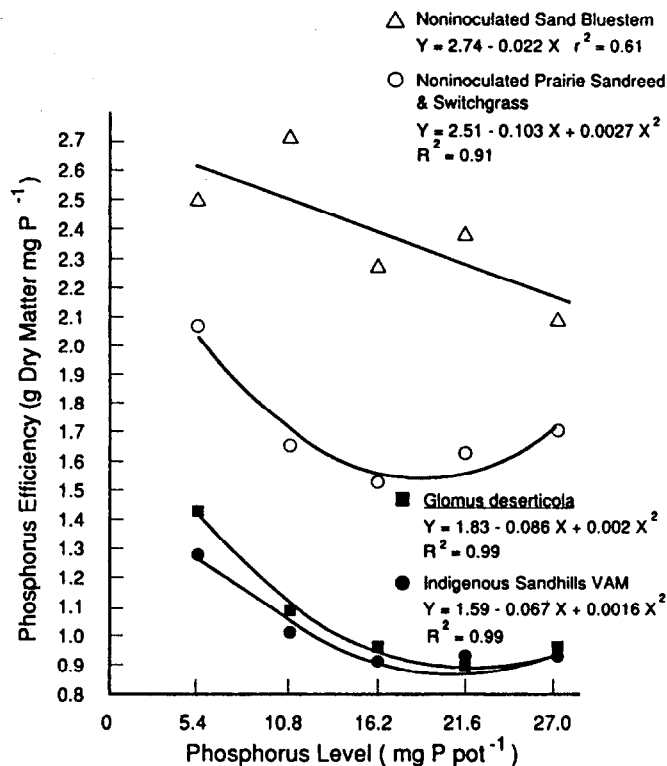


Fig. 6. Phosphorus efficiency (PE) (mg dry matter mg P⁻¹ accumulated) with increasing P levels (mg P pot⁻¹) averaged over 3 Nebraska Sandhills warm-season grasses inoculated with *Glomus deserticola*, indigenous Sandhills VAM, noninoculated sand bluestem and an average of non-inoculated prairie sandreed and switchgrass.

of the Sandhills are highly mycorrhizal dependent and the low density or absence of mycorrhizae in the eroding soils of blowouts and abandoned center pivots may inhibit the revegetation of these sites by these warm-season grasses. A 5-fold increase in P fertilizer did not stimulate as vigorous plant responses as inoculation with VAM.

At present, the technology for obtaining sufficient quantities of VAM inoculum, and for direct, widespread and economical reintroduction of mycorrhizae into rangeland and soils has not been developed. Therefore, alternative methods need to be developed for the reintroduction of VAM and revegetation of actively eroding Sandhills soils. One possibility could be to speed-up natural succession by growing a facultatively mycorrhizal annual cover crop on these sites that could help stabilize the eroding soil and serve as a host for reestablishment of mycorrhizal fungi and as a bridge for passage of mycorrhizae to perennial Sandhills warm-season grasses that could be seeded later.

Literature Cited

- Bethlenfalvay, G.F., S. Dakessian, and R.S. Pacovsky. 1984. Mycorrhizae in a southern California Desert: Ecological implications. Can. J. Bot. 62:519-524.
- Bolan, N.S., A.D. Robson, and N.J. Barrow. 1984. Increasing phosphorus supply can increase the colonization of plant roots by vesicular-arbuscular mycorrhizal fungi. Soil Biol. Biochem. 16:419-420.
- Burzlaff, D.F. 1962. A soil and vegetation inventory and analysis of three Nebraska Sandhills range sites. Univ. Nebr. Agr. Exp. Sta. Bull. 206. Univ. Nebraska, Lincoln.
- Elliott, G.C., and A. Lauchli. 1985. Phosphorus efficiency and phosphate-iron interaction in maize. Agron. J. 77:399-403.

- Giovannetti, M., and B. Mosse. 1980.** An evaluation of techniques for measuring vesicular-arbuscular mycorrhizal colonization in roots. *New Phytol.* 84:489-500.
- Hayman, D.S. 1983.** The physiology of vesicular-arbuscular endomycorrhizal symbiosis. *Can. J. Bot.* 61:944-963.
- Hayman, D.S., and B. Mosse. 1972.** Plant growth responses to vesicular-arbuscular mycorrhiza III. Increased uptake of labile P from soil. *New Phytol.* 71:41-47.
- Hetrick, B.A.D., D.G. Kitt, and G.T. Wilson. 1988.** Mycorrhizal dependence and growth habit of warm-season and cool-season tallgrass prairie plants. *Can. J. Bot.* 66:1376-1380.
- Hetrick, B.A.D., G.W.T. Wilson, and C.E. Owensby. 1989.** Influence of mycorrhizal fungi and fertilization on big bluestem seedling biomass. *J. Range Manage.* 42:213-216.
- Hewitt, E.J. 1966.** Sand and water culture methods used in the study of plant nutrition. *Tech. Commun.* 22 (2nd Ed.). Commonwealth Bureau of Horticulture and Plantation Crops. East Malling, Maidstone, Kent. Commonwealth Agr. Bur., Farnham Agr. Bur., Farnham Royal.
- Jasper, D.A., A.D. Robson, and L.K. Abbott. 1979.** Phosphorus and the formation of vesicular-arbuscular mycorrhizae. *Soil Biol. Biochem.* 11:501-505.
- King, M.A., S.S. Waller, L.E. Moser, and J.L. Stubbendieck. 1989.** Seedbed effects on grass establishment on abandoned Nebraska Sandhills cropland. *J. Range Manage.* 42:183-187.
- Kocher, E., and J. Stubbendieck. 1986.** Broadcasting grass seed to revegetate sandy soils. *J. Range Manage.* 39:555-557.
- Linderman, R.G., and J.W. Hendrix. 1982.** Evaluation of plant response to colonization by vesicular-arbuscular mycorrhizal fungi. A. Host variables, p. 69-76. *In:* N.C. Schenck (ed), *Methods and principles of mycorrhizal research*. The American Phytopathological Society, St. Paul, Minn.
- Marschner, H. 1986.** Mineral nutrition in higher plants. Academic Press Inc., London.
- Mosse, B., and D.S. Hayman. 1971.** Plant growth responses to vesicular-arbuscular mycorrhiza. II. In unsterilized field soils. *New Phytol.* 70:29-34.
- Phillips, J.M., and D.S. Hayman. 1970.** Improved procedures for clearing roots and staining parasitic and vesicular-arbuscular mycorrhizal fungi for rapid assessment of colonization. *Trans. Br. Mycol. Soc.* 55:158-161.
- Reece, P.E., D.H. Yocom, and M.G. Boosalis. 1987.** Role of mycorrhizae in revegetation of eroded Sandhills soils. p. 164. *In:* D.M. Sylvia, L.L. Hung, and J.H. Graham (eds.), *Mycorrhizae in the next decade*. Institute Food and Agr. Sci., Univ. Florida, Gainesville.
- Reeves, F.B., B. Wagner, T. Moorman, and J. Kiel. 1979.** The role of endomycorrhizae in revegetation practices in the semi-arid west. I. A comparison of incidence of mycorrhizae in severely disturbed vs. natural environments. *Amer. J. Bot.* 66:6-13.
- Roder, W. 1985.** Factors influencing grass seeding success on a sandy soil. M.S. Thesis, Univ., Nebraska, Lincoln.
- SeEVERS, P.M., D.T. Lewis, and J.S. Drew. 1975.** Use of ERTS-1 imagery to interpret the wind erosion hazard in Nebraska's Sandhills. *J. Soil and Water Conserv.* 30:181-183.
- Steel, R.G.D., and J.H. Torrie. 1980.** Principles and procedures of statistics. McGraw-Hill Co. N.Y.
- Trappe, J.M. 1981.** Mycorrhizae and productivity of arid and semiarid rangelands, p. 581-599. *In:* J.T. Manassah and E.J. Briskey (eds.), *Advances in food producing systems for arid and semiarid lands*. Academic Press, N.Y.

Annual medic establishment and the potential for stand persistence in southern Arizona

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Abstract

Few perennial legumes have been successfully introduced into western North American rangelands receiving less than 250 mm annual precipitation. Winter annual legumes in the genus *Medicago* (medics) are native to arid sites in North Africa and the Middle East and have been successfully introduced into arid and semiarid rangelands. The objective of this study was to evaluate the potential of establishing medics in areas of the southwestern U.S. receiving between 100 and 200 mm winter precipitation (November–May). Five medic accessions from 4 species (*M. laciniata* (L.) Miller, *M. littoralis* Rhode ex Loix. Delong., *M. polymorpha* L., *M. truncatula* Gaertn.) that could avoid drought were identified in a preliminary screening nursery in 1987–88. These accessions established and produced seed in 1989–90 in a field plot at Tucson, Ariz., with 125 mm winter precipitation. Less than 5% of all seed produced by these accessions germinated following summer precipitation. Plant re-establishment in the winter 1990–91 (181 mm precipitation) from pods produced in 1989–90 was observed for only 1 accession (*M. truncatula* 'Cyprus'). New plant re-establishment and seed production was observed in 1990–91 for all 5 accessions from seed produced in 1989–90 with supplemental irrigation (300 mm) in addition to precipitation. Failure to observe comparable establishment from seed produced without irrigation was attributed to the scarcity of germinable (permeable) seeds in the soil seed bank. Rapid maturing medics that exhibit breakdown of hardseededness by autumn appear to be well adapted to southern Arizona sites receiving as little as 110 mm winter precipitation. If such introductions are to be successful, initial seeding rates in excess of 115 pure live seeds/m² may be necessary to develop a large soil seed bank.

Key Words: hardseededness, legumes, precipitation, *Medicago*, soil seed bank, winter annuals

In western North America, most attempts to introduce legumes on rangelands have utilized alfalfa (*Medicago sativa* L. and *M. falcata* L.) and other perennial forbs (Rumbaugh 1983, Berdahl et al. 1986). Once introduced, these species have generally persisted only in areas receiving more than 250 mm annual precipitation (Rumbaugh 1982). There has been little success in identifying legumes able to establish and persist on more arid sites (Rumbaugh and Townsend 1985). Annual legumes may offer advantages in revegetation of arid rangelands where winter growth is possible. In addition to the N₂ fixation and forage quality attributes common to legumes, many annual legumes have characteristics that make them especially well adapted to arid environments. By growing when temperatures and evaporative rates are lower, water use efficiency may be higher than for warm-season species (Tadmor et al. 1972). Many annual legumes produce indehiscent fruits (pods)

that break down slowly and prevent moisture from rapidly reaching the seeds (Small et al. 1991). This trait, along with hardseededness mechanisms, allows a proportion of a seed population to survive extended drought and promotes germination only when soil moisture is high enough to support seedling growth. Once germinated, annual legumes exhibit rapid vegetative growth and maturity, and extreme developmental plasticity. This increases the chances for seed production under conditions of erratic and low precipitation (Kemp 1989).

Winter annuals in the genus *Medicago*, commonly referred to as medics, occur naturally in portions of North Africa, southern Europe, and the Middle East with mild, rainy winters, and alkaline soils (Lesins and Lesins 1979). Medics typically germinate in late autumn with the onset of winter rains, grow vegetatively until early spring, and flower and produce seed as rainfall ceases in late spring. Weather-resistant pods and hardseededness allow natural re-establishment in succeeding years even if winter rains are insufficient in a season to permit seed production (Crawford et al. 1989). Medics provide high quality forage in many regions with Mediterranean climates and more than 300 mm winter precipitation. This includes approximately 50 million ha of land in Australia where self-reseeding medic pastures are used in crop rotations (Crawford et al. 1989). Certain medics are native to sites in North Africa receiving an average of less than 250 mm winter precipitation (Francis 1981) and may provide significant forage in these regions (Gintzburger 1986). Medics from these regions have also been successfully introduced into other arid grazing lands. Using soil pitting, accessions from a variety of medic species were established at 2 sites in Western Australia that received less than 100 mm of precipitation (Gintzburger 1987). Similar approaches have been used successfully in southern Australia (Campbell 1980). Local strains of *Medicago polymorpha* L. established and produced seed in 1 season with 78 mm precipitation at a site in Israel's Negev Desert (Tadmor et al. 1968). In a 9-year study, Tadmor et al. (1971) observed that medics established and persisted (re-established from the soil seed bank) at a site in Israel with average seasonal water application of 235 mm. In another long-term study using native stands, the annual legumes *Medicago polymorpha* and *Trigonella arabica* Delile established without water catchment only when precipitation exceeded about 210 mm (Tadmor et al. 1974).

In the desert and semidesert shrublands and grasslands of the southwestern U.S. and northern Mexico, between 35 and 60% of the annual precipitation (about 100 to 250 mm) generally occurs between November and May (here referred to as "winter" (Jordan 1981). Vegetation in much of this region has been severely altered by grazing, irrigated agriculture, and urbanization, and revegetation efforts have been numerous. Emphasis has been placed almost exclusively on perennial grasses and shrubs (Cox et al. 1982). While the native flora of this region contains many winter and summer annuals (Shreve and Wiggins 1964), the potential of intro-

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Table 1. Total monthly precipitation (mm) and percent of 30-year mean monthly precipitation 500 m from the study site in Tucson, Arizona in the winter of 1989–90 (initial establishment), and summer of 1990, and winter of 1990–91 (re-establishment).

Years		Winter season								Summer season					
		Nov.	Dec.	Jan.	Feb.	Mar.	Apr.	May	Total	Jun.	Jul.	Aug.	Sep.	Oct.	Total
1989–90	(mm)	3.8	11.4	34.5	15.8	32.5	7.4	0.3	105.8	8.1	141.0	40.4	24.6	9.4	223.5
	(%) ¹	21	39	139	81	144	66	6	82	145	263	74	92	39	131
1990–91	(mm)	15.7	66.5	41.1	25.1	32.8	0	0.8	181.2	—	—	—	—	—	—
	(%) ¹	89	228	140	129	145	0	19	141	—	—	—	—	—	—

¹Percent of 30-year mean.

ducing exotic short-lived annuals into disturbed sites has not been investigated. Certain medics may be well adapted for this use given the similarity in winter temperature and precipitation regimes between areas where medics are native and low-elevation sites in the southwestern U.S. (Rudloff 1981). Early winter (November–December) precipitation is important for medic establishment since seedling growth is more rapid at the relatively higher temperatures in these months. In addition, seedlings that emerge with January or February precipitation may not be able to produce seed before spring drought (Crawford et al. 1989). Medics generally are not native in areas that receive significant summer precipitation. Summer rainfall, which is common in southern Arizona, may lead to the production of medic seedlings that are likely to be lost to desiccation before flowering, leading to depletion of the soil seed bank (Crawford et al. 1989).

The objective of this study was to evaluate the potential for establishing medics in rangelands or dryland pastures in the southwestern U.S. that receive between 100 and 200 mm precipitation between November and May in addition to summer precipitation. This involved: (1) selection of medic accessions preadapted to this arid environment; (2) evaluation of seedling establishment, growth and seed production in a field plot sown with nursery produced seed; and (3) assessment of plant re-establishment and seed production in the following seasons from the soil seed bank.

Materials and Methods

Medics able to establish in regions with less than 200 mm of winter precipitation would need to flower and rapidly produce viable seed to avoid spring drought. A screening nursery was established at Tucson, Ariz., in November 1987 to identify early flowering accessions that were able to produce seed under spring drought conditions. Rows 1 m long were sown with 50 pure live seeds (PLS) of 146 accessions from 20 *Medicago* species. Accessions were provided by the U.S. Regional Plant Introduction Station, Pullman, Wash., the South Australia Department of Agriculture Genetic Resource Centre, Adelaide, and by Walter Graves, University of California, Cooperative Extension, San Bernadino, Calif. The nursery was flood irrigated 4 times between 12 November and 17 March to insure establishment. Approximately 400 mm of water was applied in addition to 126 mm precipitation.

Five accessions from 4 species were selected from this nursery for use in an establishment and re-establishment trial under natural precipitation. Selection was based on: earliness of flowering, pod production, and apparent ability to tolerate drought as pods mature. Pods were collected from these accessions and seed removed. The 5 accessions were: (1) *M. littoralis* Rhode ex Lois. Delong. accession SA 21128, collected near Garian, Libya (Gorringe and Pullen 1984); (2) 'Cyprus', a cultivar of *M. truncatula* Gaertn. introduced into South Australia from Cyprus in the 1950s (Crawford et al. 1989); (3) Serena, a cultivar bred in South Australia using various sources of *M. polymorpha* (Crawford et al. 1989); (4) *M. laciniata* (L.) Miller accession PI 498891, collected in Lanza-

rote, Canary Islands; and (5) *M. laciniata* accession PI 498847 collected 145 km south of As-Salman, Iraq (Anon. 1985).

An establishment and re-establishment trial was conducted at the Campus Agriculture, Center, Tucson, Ariz., (elev. 757 m) from October 1989 through February 1991. Soil at the site is a Gila fine sandy loam (thermic torrifluvent) that was lightly tilled and free of vegetative residue. The effect of sowing a cool-season grass (Indian ricegrass, *Oryzopsis hymenoides* (R. & S.) Ricker. 'Nezpar') along with medics was also investigated in this study. Indian ricegrass established uniformly in all plots; however, the presence of grass did not have any significant ($P \leq 0.05$) effect on any aspect of medic performance. Therefore, all data reported are for plots sown with and without grass. The experimental design was a split-split-plot with 5 replicates. Two soil moisture treatments made up the main plot factor. Each of the moisture treatments was imposed within each replication. One main plot treatment received 300 mm of supplemental water from 15, 20-mm applications made with a hand-held sprinkler between 7 November and 9 April. The other moisture treatment received only precipitation. Presence or absence of Indian ricegrass represented the sub-plot factor, while the 5 medic accessions were the sub-sub-plot factor. Each sub-sub-plot contained 3 rows 0.75 m long.

Medic seeds were scarified using sandpaper and treated with commercial peat-base inoculant of *Rhizobium meliloti*. Seeds (43 PLS/row) of the 5 medic accessions (with or without 20 PLS of Indian ricegrass) were sown by hand on 29 October 1990 in each of the three 0.75-m rows in the plots (~ 115 PLS medics/m²). Seeds were sown in the bottom of 4-cm deep furrows made at 0.5-m intervals and lightly covered to a depth of approximately 1 cm.

Individual live medic plants in each row were counted every 7 days beginning with the first emergence on 11 November 1989 until early June 1990. Date of first flower within each plot was also recorded. In June 1990, dry medic shoots from the outside 2 rows of each plot were cut at ground level and bagged. Pods from plants in these rows were collected separately. Pods in the central row were buried 2 to 5 cm deep to estimate re-establishment. Shoots and pods from each plot were weighed after air drying and seed yield estimated based on mean seed number and weight from a sample of 20 threshed pods. Pods harvested by 10 June 1990 were threshed and germination of unscarified seeds evaluated in a laboratory test. A total of 50 seeds for each accession source were tested in 2 replicates. The experiment was performed using sealed petri dishes containing water-saturated filter paper at $25 \pm 1^\circ$ C under 12 hours light and 12 hours darkness. Percent germination was recorded after 30 days.

Re-establishment from medic pods or seeds buried in the central row of each plow was evaluated under natural summer and winter precipitation in July and August 1990, and February 1991. Live medic seedlings in each row were counted at each date. Daily precipitation and temperature data were taken from a weather station located approximately 500 m from the field site.

Hardseededness and pod disintegration were evaluated by sowing whole pods of each of the medic accessions in $70 \times 30 \times 10$ cm

Table 2. Seedling mortality, forage weight, and seed production of 5 medic accessions grown with 125 mm precipitation during the winter of 1989–90.

<i>Medicago</i> species	Accession	Premature seedling mortality ¹ (%)	Mature plants per row ² (no.)	Dry wt mature plants ² (g row ⁻¹)	Seeds per row (no.)	Seed yield ³ (kg ha ⁻¹)
<i>laciniata</i>	PI 498891	53.5 A ⁴	6.6 ± 1.1 B	3.4 ± 0.9 B	459 ± 139 C	15.9 C
<i>laciniata</i>	PI 498847	53.2 A	8.7 ± 1.2 B	11.1 ± 5.6 AB	1094 ± 562 A	52.5 B
<i>littoralis</i>	SA 21128	52.4 A	15.1 ± 3.4 A	20.6 ± 7.1 A	44.8 ± 141 B	29.9 B
<i>truncatula</i>	'Cyprus'	45.9 A	19.7 ± 1.3 A	16.4 ± 3.9 AB	461 ± 138 B	47.9 A
<i>polymorpha</i>	'Serena'	57.3 A	16.1 ± 2.3 A	8.0 ± 1.4 A	197 ± 42 C	20.5 B

¹Seedlings that died before 1 May 1990.

²After all plants had desiccated, 1 May 1990.

³Based on seed yield on area of each row (0.5 × 0.75 m).

⁴Means within a column followed by the same letter are not significantly different ($P \leq 0.05$) using Duncan's Multiple Range Test.

Duncan's Multiple Range Test.

metal flats filled with soil from the field site. These flats were placed outside an area where they would receive all precipitation and experience natural temperatures. Each of 10 flats contained 20 pods of the 5 accessions grown in 1989–90 in plots with precipitation only or precipitation and supplementary irrigation. One flat was initially irrigated using a hand sprinkler 1 July 1990 and the soil was kept moist through February 1991. Every 15 days, irrigation was initiated in another flat until all flats were irrigated (13 November). Seedlings were counted in each flat on 8 and 18 August, 10 October, 6 and 18 November, and 2 and 29 December 1990 and 2 February 1991.

Results

Precipitation during the winter season (November–May) of 1989–90 was 82% of the 30-year mean for the site (Table 1). Precipitation greater than 5 mm was received on 8 days while amounts less than 5 mm were received on 17 days. Rainfall received during November and December was significantly below long-term averages for the site, with precipitation occurring on only 3 days. Seedling emergence was observed in all accessions following a 3.4 mm rainfall on 21 November (Fig. 1). January precipitation was above normal and maximum seedling populations were observed for all accessions in early February. Premature mortality (before pod maturity) due most likely to drought was first noted in late February (Fig. 1), averaged 52.5% of emerged seedlings, and did not differ among accessions (Table 2). Serena flowered first (13 February) followed by PI 498847, SA 21128, and Cyprus (16 February), and PI 498891 (2 March). The number of plants and weight of herbage harvested at maturity varied widely among accessions and was not closely associated with the number of seeds produced (Table 2).

Even though June and July precipitation was above average, (Table 1) only field-buried pods of accession SA 21128 that had been produced without supplementary irrigation showed substantial seedling emergence in the summer (Table 3). These seedlings all desiccated before flowering but this represented a loss of less than 5% of the seed produced by this accession, the balance remaining in the soil seed bank. Additional moisture during the year of seed production had a dramatic effect on seed production and germination in the following summer. The mean number of seeds produced with supplemental irrigation was over 3 times greater (mean ± SE: 1982 ± 418 seeds/row) than that produced with natural precipitation (532 ± 149 seeds/row). In 4 of the 5 accessions essentially all seeds produced with irrigation that were germinable by mid-summer did germinate by late-August (Table 3). Over 50% of the permeable seed of the fifth accession also germinated in the summer. Immediately after harvest there was no difference in germination between seeds from the 2 moisture treatments (mean % germination ± SE: 5.9 ± 2.0% in seed from plots receiving precipitation only vs. 6.0 ± 1.4% in seed from irrigated plots).

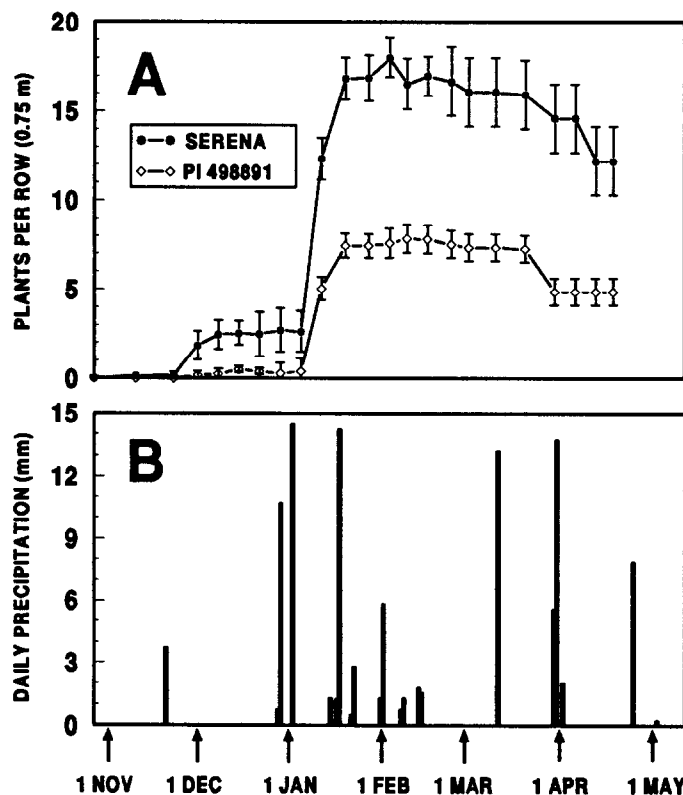


Fig. 1 Mean population of live plants (± SE) of *Medicago polymorpha* 'Serena' and *M. laciniata* PI 498891 with 125 mm precipitation at Tucson, Arizona between 29 October 1989 and 15 April 1990. These 2 accessions exhibited the highest and lowest maximum plant populations of the 5 accessions evaluated. Similar patterns of emergence and mortality were observed for other accessions. 1B. Daily precipitation recorded 500 m from the study site.

Mean seed weight did not differ between seeds from the 2 moisture treatments (2.68 ± 0.53 mg/seed with precipitation vs. 2.72 ± 0.69 mg/seed with irrigation). Supplemental moisture resulted in a slight increase in the mean number of seeds produced/pod (5.16 ± 1.01 irrigated vs. 3.82 ± 0.67 precipitation only).

During the winter season of 1990–91 precipitation was 141% of the long-term average (Table 1). Precipitation greater than 5 mm was received on 12 days while amounts less than 5 mm were received on 19 days. A total of 11 days in November and December received measurable precipitation. Nevertheless, only Cyprus seedlings emerged from buried pods in plots that had received only precipitation in 1989–90 (Table 3). As in the summer, seedling

Table 3. Seed permeability and seed re-establishment during the Summer and Winter seasons of 1990–91 from medic pods produced in the Spring of 1990 with 125 mm precipitation only or precipitation plus 300 mm irrigation.

Medicago species	Accession	Seed produced in 0.75-m row with precipitation only				Seed produced in 0.75-m row with precipitation + irrigation				
		Summer (Jul.–Oct.)		Winter (Nov.–Feb.)		Summer (Jul.–Oct.)		Winter (Nov.–Feb.)		
		No. seedlings		Permeable seeds ¹		No. seedlings		Permeable seeds		No. seedlings
		(% perm. seeds ¹)	(% of all seeds)	(% of all seeds)	(% of perm. seeds)	(% perm. seeds)	(% of all seeds)	(% of all seeds)	(% perm. seeds)	(% of all seeds)
<i>laciniata</i>	PI 498891	2.3 ± 0.9 (23.2)	(0.5)	41.9 ± 15.2 (9.1)	0	14.8 ± 4.4 (>100)	(1.9)	36.0 ± 19.2 (4.7)	6.5 ± 3.5 (8.1)	(0.9)
<i>laciniata</i>	PI 498847	1.9 ± 0.9 (38.1)	(0.2)	50.6 ± 25.1 (4.6)	0	10.3 ± 3.0 (>100)	(0.4)	93.6 ± 28.1 (3.4)	10.5 ± 5.5 (11.2)	(0.4)
<i>littoralis</i>	SA 21128	18.3 ± 6.6 (>100)	(4.1)	64.1 ± 15.4 (14.2)	0	24.6 ± 6.8 (>100)	(0.8)	507.6 ± 67.8 (17.1)	19.5 ± 10.9 (3.8)	(0.7)
<i>truncatula</i>	'Cyprus'	2.0 ± 1.1 (28.6)	(0.4)	59.6 ± 22.6 (12.9)	1.8 ± 1.2 (3.0)	11.9 ± 3.9 (>100)	(0.9)	158.0 ± 53.6 (11.8)	39.0 ± 12.4 (24.7)	(2.9)
<i>poly-morpha</i>	'Serena'	0.8 ± 0.5 (>100)	(0.4)	34.6 ± 7.4 (17.6)	0	0.2 ± 0.1 (>100)	(0.1)	413.3 ± 66.6 (20.0)	67.0 ± 21.8 (16.2)	(3.2)

¹Number of permeable seeds estimated from seedlings emerging from buried pods in soil-filled flats kept wet.

emergence in 1990–91 was considerably higher in plots that had received supplemental irrigation during 1989–90. This was observed even though the fraction of the seed population that was permeable did not differ significantly ($P \leq 0.05$) for individual accessions grown with or without supplementary irrigation (Table 3). For some accessions, mean seedling populations approached or exceeded those observed in the year of establishment (>20–30 plants/row). However, no more than 3.2% (mean = 1.6%) of all seeds resulted in established seedlings. Seeds were produced in the spring of 1991 by Cyprus plants in plots that received only precipitation in 1989–90 and by all accessions that had received supplemental irrigation in the year of establishment.

Discussion and Conclusions

Re-establishment of medic pastures is frequently unsuccessful if only seed produced in the previous season is available (Carter and Lake 1985). This was the case in this experiment as medics initially established and produced seed in a relatively dry winter but did not exhibit significant re-establishment in the following winter with above-average precipitation. Poor second-year establishment following favorable initial establishment was also observed in a medic establishment trial conducted in northern Utah (Rumbaugh and Johnson 1986).

Substantial re-establishment was observed in plots that received supplemental irrigation during seed development. Negligible re-establishment from plots that received only precipitation may have been due to poor seed quality associated with moisture stress during seed maturation. Andrew (1956) reported that *M. tribuloides* (*M. truncatula*) produced a lower percentage of mature seeds which had reduced hardseededness when plants were exposed to moisture stress following flowering. In this experiment, germination percentage initially following pod harvest (a measure of hardseededness and viability) did not differ between seeds from the 2 moisture treatments. Moreover, moisture stress during seed development did not reduce mean seed weight. This indicates that seed produced under the 2 moisture regimes did not differ physiologically.

Differences in re-establishment between the 2 moisture treatments may simply reflect the larger seed populations available in the irrigated plots. Mean seed yield produced with natural precipitation was considerably lower in all accessions than the 200 kg/ha considered necessary for re-establishment of medic pastures in Australia (Carter and Lake 1985). Assuming re-establishment

rates comparable to those seen with pods from irrigated plots (3.8–24.7% of permeable seeds), a mean of approximately 6.3 ± 1.5 seedlings would be expected to emerge in 1990–91 in plots that had received only precipitation in 1989–90. Given the very low number of permeable seeds available, and the variability in emergence and seed number among plots, lack of seedling emergence should be expected.

These data indicate that rapid maturing medics could be established in southern Arizona with less than 100 mm precipitation during the period between November and May. High seed populations following initial establishment would be necessary to increase the chances for successful re-establishment. Initial seeding rates in excess of 115 PLS/m² could increase the probability of successful re-establishment. Considering plant populations in the year of establishment, seed production, and seed loss due to summer germination, the 2 medic cultivars Cyprus and Serena appear to represent the type of medic that could be established in southern Arizona. SA 21128 produced high plant populations and relatively large plants but may have insufficient hardseededness in summer to persist in this environment. Accessions of *M. laciniata* had insufficient breakdown of hardseededness by the onset of winter rains to provide dependable re-establishment. Future screening of medic germplasm for this environment should focus on rapid breakdown of hardseededness in autumn in addition to early flowering and drought survival.

Literature Cited

- Andrew, W.D. 1956. Interaction of the moisture and nutritional regimes on hard seed production in barrel medic *Medicago tribuloides*. Aust. Pl. Nutr. Conf. 1:79–91.
- Anonymous. 1985. Plant inventory. USDA-ARS. Vol 19:404,406.
- Berdahl, J.D., A.C. Wilton, R.J. Lorenz, and A.B. Frank. 1986. Alfalfa survival and vigor in rangeland grazed by sheep. J. Range Manage. 39:59–62.
- Campbell, D.A. 1980. Pitting and sown pastures: The Nundora experience. Agr. Gaz. New South Wales 91:6–8.
- Carter, E.D., and A. Lake. 1985. Seed, seedling and species dynamics of grazed annual pastures in South Australia. Proc. XV Inter. Grassl. Cong. p. 654–656.
- Cox, J.R., H.L. Morton, T.N. Johnson, Jr., G.L. Jordan, S.C. Martin, and L.C. Fierro. 1982. Vegetation restoration in the Chihuahuan and Sonoran Deserts of North America. USDA-ARS, ARM-W-28.
- Crawford, E.J., A.W.H. Lake, and K.G. Boyce. 1989. Breeding annual *Medicago* species for semiarid conditions in southern Australia. Adv. Agron. 42:399–437.

- Francis, C.M. 1981. The distribution and ecology of annual *Medicago* species in North West Libya. Aust. Plant Introd. Rev. 13:3-14.
- Gintzburger, G. 1986. Seasonal variation in above-ground annual and perennial phytomass of an arid rangeland in Libya. J. Range Manage. 39:348-353.
- Gintzburger, G. 1987. The effect of soil pitting on establishment and growth of annual *Medicago* spp. on degraded rangeland in Western Australia. Aust. Rangel. J. 9:49-52.
- Gorringe, R.J., and R. Pullen. 1984. Aust. Plant Introd. Rev. 18:A-71.
- Jordan, G.L. 1981. Range seeding and brush management on Arizona rangelands. T81121. Coop. Exten. Serv. Univ. of Arizona, Tucson.
- Kemp, P.R. 1989. Seed banks and vegetation processes in deserts. p. 257-281. In: M.A. Leck, V.T. Parker, and R.L. Simpson (ed.) Ecology of soil seed banks, Academic Press, San Diego.
- Lesins, K., and I. Lesins. 1979. Genus *Medicago* (Leguminosae). A taxogenetic study. Junk, the Hague, Netherlands.
- Rudloff, W. 1981. World-Climates. Wissenschaftliche Verlag., Stuttgart.
- Rumbaugh, M.D. 1982. Reseeding by eight alfalfa populations in a semi-arid pasture. J. Range Manage. 35:84-86.
- Rumbaugh, M.D. 1983. Legumes—Their use in wildland plantings. p. 115-122. In: S.B. Monsen and N. Shaw (eds.). Managing Intermountain rangelands—Improvement of range and wildlife habitat. USDA-ARS General Tech. Rep. INT-157.
- Rumbaugh, M.D., and D.A. Johnson. 1986. Annual medics and related species as reseeding legumes for northern Utah pastures. J. Range Manage. 39:52-58.
- Rumbaugh, M.D., and C.E. Townsend. 1985. Range legume selection and breeding in North America. p. 29-39. In: J.R. Carlson and E.D. McArthur (eds.). Range plant improvement in western North America. Soc. Range Manage., Denver, Colo.
- Shreve, F., and I.L. Wiggins. 1964. Vegetation and flora of the Sonoran Desert. Stanford Univ. Press, Stanford, Calif.
- Small, E., M. Jomphre, and B. Brookes. 1991. *Medicago truncatula* f. *laxicycla* (Leguminosae), a new taxon with loose fruit coiling promoting rapid germination of the fruit-retained seeds. Pl. Syst. Evol. 174:37-46.
- Tadmor, N.H., M. Evenari, and J. Katznelson. 1968. Seeding annual and perennials in natural desert range. J. Range Manage. 21:330-331.
- Tadmor, N.H., L. Shanani, and M. Evenari. 1971. "Runoff farming" in the desert. V. Persistence and yields of annual range species. Agron. J. 63:91-95.
- Tadmor, N.H., M. Evenari, and L. Shanani. 1972. Primary productivity of plants as function of water use. p. 151-157. In: E.L. Rodin (ed.). Ecophysiological foundations of ecosystem productivity in arid zones. Nauka, Leningrad.
- Tadmor, N.H., E. Eyal, and R.W. Benjamin. 1974. Plant and sheep production on semiarid annual grassland in Israel. J. Range Manage. 27:427-432.

Changes in rangeland pricing method during the inflation-deflation price cycle

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Abstract

Utah rangeland real estate underwent an inflation-deflation price cycle from 1975 through 1988. A total of 166 Utah land sales were analyzed to determine whether factors affecting rangeland prices changed during the price cycle. Regression analysis was used to test changes in method of pricing rangeland between the inflation phase (1975-81) and deflation phase (1982-87). The effects on sale price of parcel size (acres or hectares) and number of deeded animal unit months (AUMs) differed between the 2 time periods. Size of parcel sold significantly affected land price in the first time period, but not in the second. Conversely, the number of deeded AUMs did not significantly influence land price in the first time period, but did in the second. Thus rangeland tended to be priced per acre (hectare) during the inflation phase of the price cycle and per AUM of carrying capacity during the deflation phase. These results indicate that rangeland owners should try to maintain or improve range condition and carrying capacity to preserve real estate values during deflationary times.

Key Words: range appraisal, price per animal unit, price per acre, carrying capacity value

Cropland is commonly priced on a unit price basis (dollars per acre or hectare). Potential buyers make subjective judgments about the quality of the cropland. This is normally done for each soil type in order to arrive at a total land value based on production potential. Rangeland has traditionally been priced in a similar manner with value expressed in terms of sustainable grazing capacity (typically as dollars per animal unit month). However, the rural real estate markets of the 1970's brought some changes in pricing practices.

Agricultural land prices peaked in the early 1980's and then fell dramatically (USDA 1988). During the inflation phase of the price cycle, one often heard the statement that a farm had "sold for more than it was worth" (Suter 1980). This statement refers to 2 different concepts of value: value-in-exchange and earnings value. Value-in-exchange is the concept upon which the market transfer of land is based. However, if revenue from inflated land cannot cover loan payments, earnings value is less than value-in-exchange (Fig. 1). Suter (1980) identified "basic farm value" as the valuation technique that capitalizes long term earnings potential into land value. This technique was rarely used during the inflation phase of the price cycle because it did not accurately estimate agricultural land prices.

Factors other than earnings may partially account for inflated farm and ranch prices. Figure 2 compares Utah real estate values (USDA 1984, 1988) and a proxy index for net ranch income, which

is the difference between the Beef Cattle Price Index and the Prices Paid Index (USDA 1987). Although earnings declined throughout the 1975-87 period, land values increased until 1982.

Each of the 3 standard appraisal techniques (cost approach, market data approach, and income approach) attempts to estimate real estate value in terms of current market price (Suter 1980). As noted above, the income capitalization approach often has not been a reliable estimator of farm and ranch values during the inflation phase of the price cycle. Only when appraisers have used a market-derived real capitalization rate (rather than a real borrowing or real opportunity cost rate) has the income approach been accurate. The correlation between earnings potential and sale price is critical for many buyers, especially if farm and ranch income is the only source of revenue. However, Godfrey and Andersen (1988) noted that over 95% of Utah farmers had some source of nonfarm income.

We hypothesized that prior to 1975 (the approximate beginning of the most recent inflation cycle) Utah ranch real estate was priced not in terms of dollars per acre (hectare), but on a dollars per animal unit month (AUM) basis. We further hypothesized that during the years of rapid land appreciation, Utah rangeland sold on a dollars per acre (hectare) basis. Nonfarm opportunities such as residential development, recreation potential, and mineral reserves may have influenced land values. Thus purchasers may have ignored differences in land productivity and based purchasing decisions solely on number of acres (hectares) rather than on livestock carrying capacity. Finally, we hypothesized that since the 1982 price decline (a decline in *both* per acre *and* per AUM prices), rangeland has once again sold by the AUM. If true, it would be extremely important for land owners to maintain (or improve) range condition and carrying capacity in order to preserve land values during the deflation phase of the land price cycle.

Methods

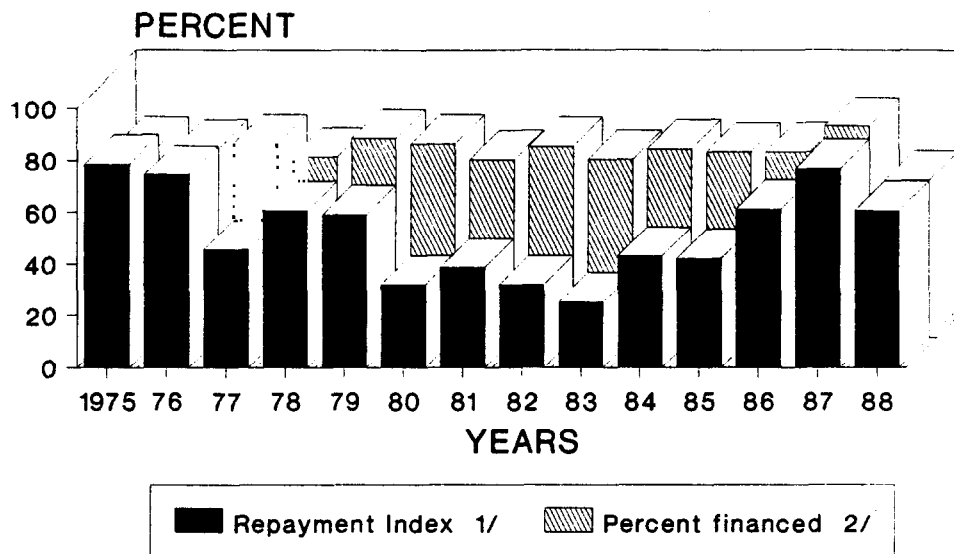
Rangeland prices during 1975-81 and 1982-87 were analyzed by regression equations for each of the 2 periods to test for any differences in pricing methods between the 2 periods. The relevant question was: on what basis has rangeland been priced in the market during the period 1975-87?

Comparing 166 individual Utah ranch sales over the period 1975-87 involved analysis of data collected by King (1981) along with those of this study. Combining data required that variables for the 2 studies be defined and measured in the same manner. For example, King's measure for carrying capacity was in brood cow units. These were converted to AUMs by multiplying brood cow units by 1.2 animal units yearlong/brood cow unit and then by 12 months (Workman and King 1982).

The peak land price month was identified as February 1982 (USDA 1984 and 1988) and the combined data were divided into 2

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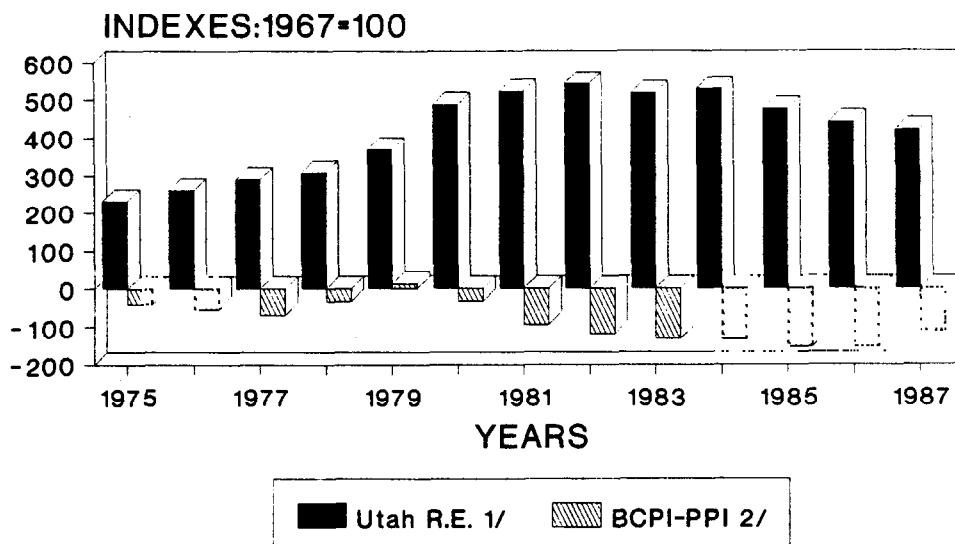
Repayment ability Vs. Percent financed Mountain States



1/ Gertel, personal communication
2/ Farm Real Estate Market Developments
2/ Agricultural Resources

Fig. 1. Repayment ability index (in percent) compared to the average percent financed on farm and ranch land in the mountain states for 1975-88.

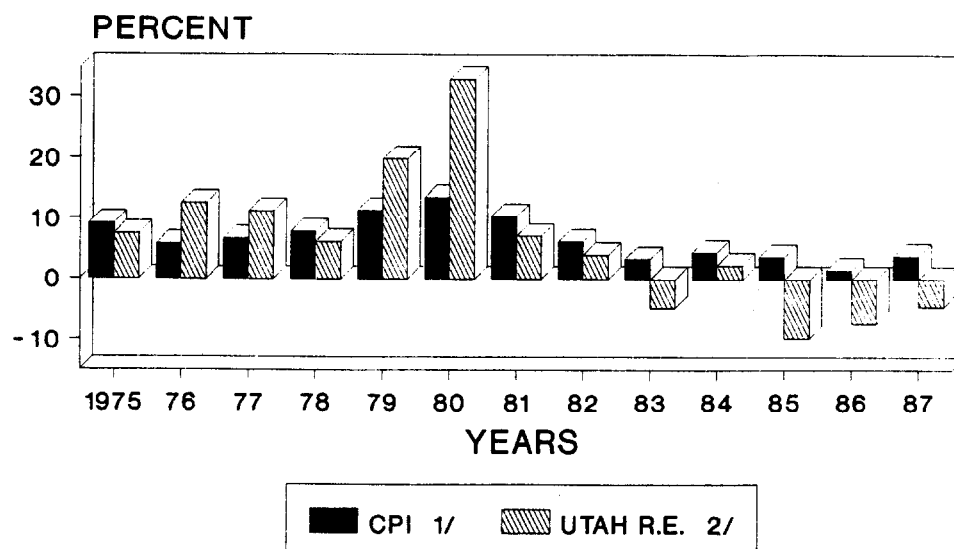
Comparison of Utah Real Estate Index to Proxy Index of Net Ranch Income



1/ Farm Real Estate Market Developments
1/ Agricultural Resources, 1985-1988
2/ Agricultural Prices, 1975-1987

Fig. 2. A comparison of the Utah real estate index (Utah R.E.) to a proxy index for net ranch income for 1975-87. The proxy index (BCPI-PPI) is the difference between the Beef Cattle Price Index (BCPI) and the Prices Paid Index (PPI).

Percent Change in CPI Index Compared to Change in Utah R.E. Index



1/ Statistical Abstract of the U.S.
 2/ Farm Real Estate Market Developments
 2/ Agricultural Resources, 1985-1988

Fig. 3. Percent change in the Consumer Price Index (CPI) compared to the change in the Utah Real Estate Index (Utah R.E.) for 1975-87.

time periods based on that month. Figure 3 shows the rise and fall of land values in Utah over the price cycle. The annual percentage change in dollars per acre (hectare) increased rapidly during 1979 and 1980. With the exception of 1984, annual changes during the period 1982 to 1987 were negative.

The regression model estimated total ranch sale price as a function of size of sale (Acres), deeded carrying capacity (Daums), leased carrying capacity (Laums), value of dwellings and buildings (Dwell), distance to town (Dist), index variables for sale size (S_1 and S_2), and an index variable for the stage of the land price cycle (P). The dependent variable was expressed as total ranch sale price in nominal dollars, following the methods of Torell and Fowler (1986). Carrying capacity variables (deeded and leased) were measured in animal unit months. Total value of dwellings and buildings included in land sales was expressed in dollars.

Distance values were measured as the shortest distance in miles along established roads from the ranch to a Utah town of at least 2,000 people (Andriot 1983). Measurements were made to the nearest mile using the Sigma Scan digitizing program (Jandel Scientific, Corte Madera, Calif.)

The index (dummy) variables for size of sale (<40, 40-640, and > 640 acres) followed the methods of Rowan and Workman (1992) and were represented as:

$$S_1 = 1 \text{ if size is } < 40 \text{ acres} \quad (1)$$

= 0 if size is 40-640 acres
 = -1 if size is > 640 acres

$$S_2 = 0 \text{ if size is } < 40 \text{ acres} \quad (2)$$

= 1 if size is 40-640 acres
 = -1 if size is > 640 acres.

Stage of the land price cycle was represented by a dummy variable (Gujarati 1988) for the 2 time periods (P). The period

January 1975 to February 1982 was assigned a value of $P = 0$ and the period March 1982 to December 1987 a value of $P = 1$. Each independent variable value was multiplied by the binary value for its respective time period. The regression model was defined as:

$$\hat{Y}_i = \alpha_0 + \alpha_1 P_i + \beta_1 X_{i1} + \beta_2 (P_i X_{i1}) + \beta_3 X_{i2} + \beta_4 (P_i X_{i2}) + \dots + \beta_{2k-1} X_{ik} + \beta_{2k} (P_i X_{ik}) + \epsilon_i \quad (3)$$

where

\hat{Y}_i = total ranch sale price,
 P_i = 1 if sale occurred after February, 1982,
 P_i = 0 if sale occurred February, 1982, or earlier,
 X_i = 1,2,...,k = independent variables:
 Acres = size of sale
 Daums = deeded carrying capacity
 Laums = leased carrying capacity
 Dwell = value of dwellings and buildings (\$),
 Dist = distance to a town of 2,000 population (miles),
 S_1 and S_2 = index variables for ranch sale size, and
 ϵ_i = error term.

This model identified α_1 as the differential intercept coefficient. If significant at a t-test probability <0.05, this coefficient indicates that the second time period Y-axis intercept differs significantly from that of the first time period. In the same manner, β_2 , β_4 , ..., β_{2k} are the differential slope coefficients which indicate whether or not the independent variable slope coefficients of the first time period differ from those of the second time period.

Regressions for both time periods were estimated from this 1 regression model (Gujarati 1988). They were:

First time period (1975-81)

$$\hat{Y}_i = \alpha_0 + \beta_1 X_{i1} + \dots + \beta_k X_{ik} + \epsilon_i \quad (4)$$

where $i = 1, 2, \dots, k$, and

Second time period (1982-87)

$$\hat{Y}_i = (\alpha_0 + \alpha_1) + (\beta_1 + \beta_2) X_1 + \dots (\beta_{2k-1} + \beta_{2k}) X_k + \epsilon_i \quad (5)$$

The combined coefficients of $(\beta_1 + \beta_2) \dots (\beta_{2k-1} + \beta_{2k})$ were tested for significance using the formula:

$$t = \frac{(\beta_1 + \beta_2)}{S_E(\beta_1 + \beta_2)} \quad (6)$$

where

$$S_E(\beta_1 + \beta_2) = \sqrt{V(\beta_1) + V(\beta_2) + 2 \text{Cov}(\beta_1, \beta_2)}$$

This t-value was derived differently than those for the other regression coefficients since it considered the sum of individual variances of β_1 and β_2 plus the covariance of these coefficients.

One additional test was used to determine whether ranches were priced differently before and after the peak price year: restricted and unrestricted regression models. Land sales were divided into the same 2 time periods described above. Next a "restricted" regression was estimated for the first time period (1975-81) based on the following independent variables: Acres, Dwell, Dist, and ranch size index variables (S_1 and S_2). Then an "unrestricted" regression was estimated for 1975-81 based on the same set of independent variables plus Daums and Laums. An F-test was then applied to test whether deeded and leased carrying capacity significantly affected total ranch sale price during the first time period (1975-81).

Similar "restricted" and "unrestricted" regressions were estimated to determine if size of sale (Acres) significantly affected total sale price during the second time period (1982-87). The restricted regression included the following variables: Dwell, Dist, ranch size index variables (S_1 and S_2), Daums, and Laums. The unrestricted regression included the same set of independent variables plus size of sale (Acres).

The formula for the F-ratio was:

$$F = \frac{(R^2_{UR} - R^2_R)/m}{(1 - R^2_{UR})/(N-k)} \quad (7)$$

where

- R^2_{UR} = R^2 value obtained from unrestricted regression
- R^2_R = R^2 value obtained from restricted regression
- m = number of linear restrictions
- k = number of parameters in unrestricted regression
- N = number of observations

Results and Discussion

Prices of Utah ranches sold before and after the peak year (1982) were compared to identify changes in the method of pricing rural real estate from 1975 through 1987. Applying Gujarati's (1988) dummy variable regression approach to the analysis of differences in land pricing method between the 2 phases of the land price cycle yielded a significant ANOVA difference ($P < 0.0001$) between the 2 phases in the coefficients for Acres and Daums. The 2 time-period equations were derived from the same regression model. If differential slope coefficients were statistically significant ($P < 0.05$), they were added to independent variable coefficients for the first time period, resulting in the equation for the second time period (Table 1). The combined coefficients were then re-tested using the t-test formulation described above. The 2 time period regressions were:

$$\begin{aligned} \text{First time period 1975-81: Total ranch price} = \\ 354,037.27^{**} + 102.75^{**}(\text{Acres}) + 4.06(\text{Daums}) + \\ (80,929.26) \quad (15.70) \quad (33.19) \\ 14.84(\text{Laums}) + 2.50^{**}(\text{Dwell}) - 2,832.25(\text{Dist}) + \\ (22.22) \quad (0.72) \quad (1,626.52) \end{aligned}$$

Table 1. Variables showing significance at the 0.05 and 0.01 levels in the 2 time period regression, 1975-81 and 1982-87.

Variables	First time period	Differential terms	Second time period
Intercept	+354,037.27** (80,929.26)	-140,077.53 (98,400.20)	+354,037.27** (80,929.26)
Number of Acres	102.75** (15.70)	-103.06** (20.87)	-0.31 (13.50)
Number of Daums	+4.06 (33.19)	+93.57* (46.45)	+97.63** (32.49)
Number of Laums	+14.84 (22.22)	+11.92 (39.58)	+14.84** (22.22)
Dollars of Dwellings	+2.50** (0.72)	+1.73 (1.41)	+2.50** (0.72)
Miles of Distance	-2,832.25 (1,626.52)	-715.30 (2,060.64)	-2,832.25 (1,626.52)
Dummy S_1	+58,809.11 (98,600.63)	-164,388.51 (118,005.69)	+58,809.11 (98,600.63)
Dummy S_2	-101,306.57 (69,640.61)	+61,207.51 (85,483.90)	-101,306.57 (69,640.61)

* = significance level 0.05 ** = significance level 0.01

$$58,809.11(S_1) - 101,306.57(S_2) \quad (8)$$

(98,600.63) (69,640.61)

* = significance level 0.05 ** = significance level 0.01

Numbers in parentheses below coefficients are standard errors of the coefficients.

$$\begin{aligned} \text{Second time period 1982-87: Total ranch price} = \\ 354,037.27^{**} - 0.31(\text{Acres}) + 97.63(\text{Daums}) + \\ (80,929.26) \quad (13.50) \quad (32.49) \\ 14.84(\text{Laums}) + 2.50^{**}(\text{Dwell}) - 2,832.25(\text{Dist}) + \\ (22.22) \quad (0.72) \quad (1,626.52) \\ 58,809.11(S_1) - 101,306.57(S_2) \quad (9) \\ (98,600.63) \quad (69,640.61) \end{aligned}$$

* = significance level 0.05 ** = significance level 0.01

These regressions indicate that number of acres sold was a significant predictor of total ranch sale price during 1975-81, but was no longer significant after 1981. Conversely, deeded carrying capacity was a significant predictor after 1981, but not before.

The final test of differences in the contribution of number of acres, number of deeded AUMs, and number of leased AUMs between the 2 time periods was conducted with restricted and unrestricted regressions. The first time period comparison (1975-81), which added Daums and Laums to a restricted model already containing number of acres, dwelling value, distance, and sale size index variables S_1 and S_2 , gave an insignificant F-value. It was concluded that (1) carrying capacity did not significantly affect total ranch sale price during the period 1975-81 and (2) during this period land was priced by the acre (hectare) rather than by the AUM of carrying capacity. Likewise, the second time period comparison (1982-87), which added acres to a restricted model already containing Daums, Laums, Dwell, Dist, S_1 , and S_2 , also gave an insignificant F-value. Thus, number of acres (hectares) did not significantly affect total ranch price during 1982-87 and during this period land was priced by the AUM rather than by the acre (hectare).

Scott (1983) reported that farmers and ranchers have traditionally purchased land if it requires revenue from no more than 2 acres (hectares) to make payments on 1 additional acre (hectare). How-

ever, this rule of thumb may be optimistic for Utah operators. Godfrey and Andersen (1988) noted that Utah farmers and ranchers are in a better than average debt/asset position, but that net income per dollar of assets is below the national average. Consequently, it is important for Utah land purchasers (and likely those in other states) to base purchase decisions on the relationship between land prices and net returns to that land.

Summary and Conclusions

The purpose of this study was to determine whether the method of pricing rangeland has changed during the recent inflation-deflation land price cycle. The key questions were (1) Was rangeland, which has traditionally been priced according to carrying capacity, priced instead by the acre (hectare) during the inflation phase of the price cycle (1975-81)? and (2) During the deflation phase (1982-87) has the method of pricing returned to a price per AUM basis? Statistical analysis of 166 Utah land sales indicates that the answer to both of these questions is yes. An important implication of our study is that rangeland owners should try to maintain or improve range condition and carrying capacity in order to preserve real estate values during deflationary times.

Literature Cited

- Andriot, J.J. 1983. Population abstract of the United States. Andriot Associates, McLean, Va.
- Godfrey, E.B., and J.C. Andersen. 1988. A synopsis of the financial status of Utah agriculture. *Utah Sci.* 49:77-82.
- Gertel, K. 1989. USDA Econ. Res. Serv., Washington, D.C. personal communication.
- Gujarati, D.N. 1988. Basic econometrics. McGraw-Hill Book Co., N.Y. p. 444.
- King, K.H. 1981. A determination of Utah ranch real estate values and an analysis of factors affecting these values. M.S. Thesis, Utah State Univ., Logan.
- Rowan, R.C., and J.P. Workman. 1992. Factors affecting Utah ranch prices. *J. Range Manage.* 45:263-266.
- Scott, J.T., Jr., 1983. Factors affecting land price decline. *Amer. J. Agr. Econ.* 65:796-800.
- Suter, R.C. 1980. The appraisal of farm real estate. The Interstate Printers and Publishers, Inc., Danville, Ill.
- Torell, L.A., and J.M. Fowler. 1986. A model for predicting trends of New Mexico grazing land values. *New Mexico State Univ. Agr. Exp. Sta. Res. Bull.* 723
- U.S. Department of Agriculture. 1975-87, various issues. Agricultural prices. Nat. Agr. Stat. Serv., Washington, D.C.
- U.S. Department of Agriculture. 1984. Farm real estate market developments. Econ. Res. Serv. Washington, D.C.
- U.S. Department of Agriculture. 1985-88, various issues. Agricultural resources. Econ. Res. Serv., Washington, D.C.
- U.S. Department of Commerce. 1975-87, various issues. Statistical abstract of the United States. National data book and guide to sources. Bureau of the Census, Washington, D.C.
- Workman, J.P., and K.H. King. 1982. Utah cattle ranch prices. *Utah Sci.* 43:78-81.

Impacts of big game on private land in southwestern Montana: Landowner perceptions

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Abstract

Increasing populations of big game animals are a problem for private landowners in some parts of western North America. Influence of big game costs, hunting-related income, noneconomic benefits, size of private land holding, and proportion of total income from agriculture upon landowner management goals as well as perception of damage to forage resources were studied in 1989-1990 using a mail survey of 858 randomly selected southwestern Montana landowners. They reported that elk (*Cervus canadensis*) populations increased, did not change, or decreased on 71%, 25%, or 4% of their private lands, respectively. Similar trends were reported for mule deer (*Odocoileus hemionus*), whitetail deer (*Odocoileus virginia*), and antelope (*Antilocapra americana*). More than 50% of the respondents thought that big game damaged forage and crop yields, while less than 2% of the respondents thought that big game was beneficial to forage and crop yields. Big game consumed a mean of 511 AUMs per private landowner, which contributed to the mean big game cost of \$6,353 per landowner. Respondents desiring fewer elk, deer, and antelope outnumbered those desiring more by a 4-to-1 margin. As costs of big game increased and as dependency on agricultural income for livelihood increased, respondents desired fewer big game animals and perceived the impact of big game to be more harmful to forage and crop yields. Landowner attitudes toward big game were not significantly affected by economic returns from big game. Although owners with larger land holdings were more likely to allow hunters access to hunt big game, owners of large- and of small-sized ranches generally regarded big game populations similarly. Results from this survey should be useful in forming natural resource policy.

Key Words: big game, landowner survey, private land, wildlife impact

Populations of many big game species are an interesting problem for many rural landowners in areas such as southwestern Montana. These animals are a public resource held for the public at large by individual states (Peek 1986). Yet some of the costs associated with the maintenance of these animals are borne by private landowners in the form of forage which would otherwise be consumed by privately held livestock, as well as damage to fences and other facilities.

Customary notions of rational action (Coleman 1990) hold that people, using the resources at their disposal, act in such a way as to

efficiently achieve their goals. This model can be applied to the case of big game and rural landowners. To the degree that big game interfere with management goals, some landowners are compensated by generating income from wildlife on private lands through both nonconsumptive use and hunting. However, game animals are under federal and state governmental regulation and control. There is no property right in wild animals until they are taken, confined, or domesticated in compliance with state and federal law (Kramer 1982). Although landowners do not own the game animals which inhabit their land, they do own the exclusive right to hunt them (subject to state laws) or to lease their property to another for similar purposes (Kramer 1982). The practice of charging access fees for hunting is more common in states that are predominately privately owned, rather than in states like Montana where federal land management agencies control much of the land (Thomas 1984).

Migratory patterns also limit the ability of landowners to respond to big game animals in a way which serves their interests. While these animals may spend considerable time on private land, and incur substantial costs while doing so, they often do so in seasons other than hunting season (Nielsen et al. 1986). In such cases, the production goals of landowners may be hindered by wildlife. From the perspective of the landowners, they bear some of the costs of producing these animals, but are unable to share in the benefits of their production (Krutilla 1967, Nielsen et al. 1986).

Landowners also consider noneconomic benefits of wildlife on their land (Applegate 1981). These include, but are not limited to, the enjoyment of observing big game animals and the satisfaction in knowing that they are there and using the habitat that was provided them.

In this paper, we examine the influence of big game costs, hunting related income, proportion of total income from agriculture, and noneconomic benefits upon landowner wildlife management goals as well as perception of damage to forage resources. Because costs and benefits associated with big game may be related to the size of land holding (Nielsen et al. 1986), size of operation is controlled in our analysis. Results are useful to: (1) improve communication and understanding between private landowners and sportsmen, (2) coordinate big game management on public and private land, and (3) provide data for natural resource policy makers.

Methods

The study area included 7 southwestern Montana counties, about 4.2 million ha (Fig. 1). Seventy-one percent of the land is range, pasture, or grazeable woodland (Headwaters RC&D 1974). Five percent is used for crop production. Forty percent of the total area is privately owned, slightly more than half is administered by

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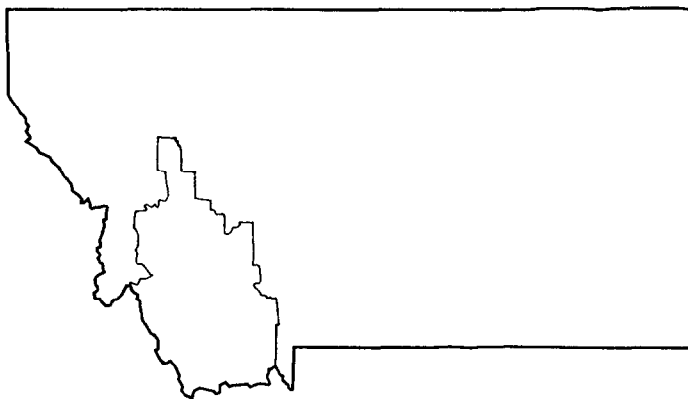


Fig. 1. The 7 counties in southwestern Montana included in the landowner-big game survey, 1989–1990.

the Forest Service (FS) and Bureau of Land Management (BLM), and the remainder is owned by the State of Montana. Livestock grazing on federal lands in the region is economically important (Lacey and Johnson 1990).

In December 1989, a self-administered, mail-back questionnaire (Dillman 1978) was mailed to 858 rural landowners. Individual landowners were randomly selected from an Agricultural Stabilization Conservation Service (ASCS) mailing list containing 1,959 names. The ASCS mailing list included all owners and operators of agricultural land, excluding subdivisions, in the study area. The percentage of landowners participating in the federal farm program is unknown. A second copy of the questionnaire was mailed in January, 1990, to landowners who failed to respond to the first mailing.

The questionnaire was designed to collect information from landowners on the impact of big game animals on their private land. Specific issues included the name of big game animals, benefits and costs associated with big game animals, and the compatibility of big game populations to private land management objectives. The possibility of landowners providing inflated estimates of big game numbers and of the length of time big game spent on private lands could not be evaluated. The analysis treated expense of repairing pasture fences damaged by big game, cost of labor and material (less the amount supplied by the State of Montana) to fence haystacks from big game, economic value of forage in haystacks that was consumed or ruined by big game, and value of private land forage consumed by big game as big game-related expenses incurred by landowners. It was assumed that forage consumed by big game could have been available for livestock. The probability of big game grazing reducing grain and hay yield and impairing long-term productivity of hay, range, and pasture lands were not included in the analysis. Likewise, forage consumed by big game was dependent on number of days that a specific type of land was used. No distinction was made between the value of forage harvested on different kinds of land.

Fees charged for grazing on federal lands are controversial. Price paid for forage to graze a mature cow of approximately 455 kg for 1 month on private land in Montana averaged \$11.00, \$0.35 less than the average price paid to graze a cow-calf pair for a month of grazing on private land (USDA 1990). To place a monetary value on forage harvested by big game from private land, the Montana rate was used. Ranchers paid \$1.86 to graze a mature cow weighing approximately 445 kg, either dry or with calf up to 6 months of age, on public land for 1 month. The traditional logic given for the lower fee on public land is the need to encourage good stewardship and private investment on public lands, higher nonfee costs of grazing on public lands, and the out-of-pocket expense

incurred when current leaseholders bought the grazing permit from original leaseholders (Torell and Doll 1991). Although the latter value (cost) has never been recognized by public land agencies, the Internal Revenue Service taxes the value of the estate when leases transfer. Regardless, many nonranchers and some ranchers who do not have a federal land grazing permit believe that the grazing fee is a subsidy and advocate reductions in federal land grazing (Hadley and Carrol 1986, National Wildlife Federation 1990, Gillis 1991).

An index of intangible benefits from big game was derived by summing the respondents' responses to the receipt of non-monetary benefits (enjoy having them, hunting by family, hunting by friends, and other). The index ranged from 0 to 4 for respondents receiving either none, or recognizing benefits in all 4 categories. The mean, median, and standard deviation of the index were 1.4, 1.0, and 1.2, respectively.

Frequency and bivariate analyses were used to summarize response to each question. Chi square analyses were used to compare the impact of big game damage among 6 types of land. Logistic and ordinary least squares regressions were used to examine influence of agricultural income, amount of land in private ownership, and big game populations on landowner satisfaction. Evidence of multicollinearity for logistic regression was evaluated by reviewing standard errors (Hosmer and Lemeshow 1989). No evidence of collinearity, as a result of a relationship between size of operation, proportion of income from agriculture, economic cost of big game, economic income from big game, or the index of intangible value of big game was found. Collinearity in the ordinary least squares regressions was checked by calculating and examining condition numbers (Kmenta 1986). A probability level of 5% (0.05) was used in all regression analyses. A study of non-respondents was not conducted, nor was nonresponse bias evaluated.

Results and Discussion

Characterization of Respondents

A total of 456 questionnaires containing useful information was returned (53% response rate). Respondents were fairly well-distributed among 7 size classes (Fig. 2). However, the U.S. Census

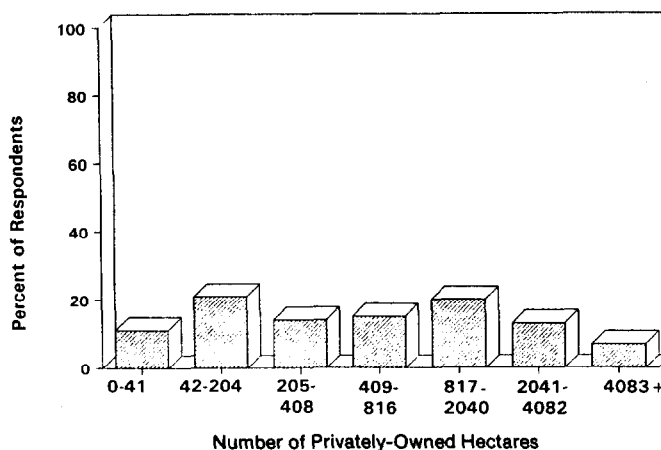


Fig. 2. Percent of respondents by land ownership patterns, landowner-big game survey, 1989–90.

of Agriculture (1987) reported a total of 1,657 ranches and farms in the study area, of which 46% and 40% are less than 202 ha and more than 409 ha, respectively. Therefore, in comparison to the Census, respondents to our survey represented a higher percentage of large landowners. In order to understand the consequences of this, size of operation was treated as a control variable in our analyses. Data

indicate size to be relatively unimportant.

Thirty-seven percent of the respondents had a BLM grazing lease, averaging 437 animal unit months (AUMs). Thirty-two percent of the respondents had FS grazing leases, averaging 1,082 AUMs. Thirty-three percent of the respondents controlled state-grazing leases, which averaged 687 ha in size.

Agricultural income represented a major part of the total income of most respondents (Fig. 3). Overall, the percent of total income from the farm/ranch enterprise averaged 75%.

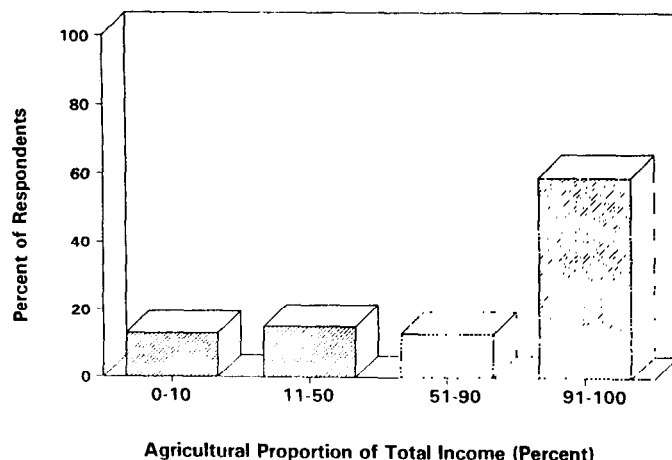


Fig. 3. Proportion of total income earned from agricultural production, as reported by respondents in landowner-big game survey, 1989-90.

Eight percent of the landowners had land enrolled in the Conservation Reserve Program (CRP). Although 70% of the CRP participants had fewer than 204 ha in the program, 80% of the CRP seedings were used by mule deer (*Odocoileus hemionus*), whitetail deer (*Odocoileus virginia*), and antelope (*Antilocapra americana*).

Estimated Economic Impact of Big Game Populations

Costs attributed to big game averaged \$6,467 per landowner (Table 1). The average cost attributed to forage consumption, hay stack damage, fencing haystacks, and repairing pasture fences damaged by big game are explained below. Landowner costs of providing information, assisting hunters, patrolling, repairing roads, and general nuisance are not included. The estimate also does not consider long-term resource damage resulting from the inability to control the time and degree of grazing by big game. These data support the contention that the interaction between livestock production and big game is often competitive (Smith 1961, Stoddard et al. 1975, Austin and Urness 1987, Nielsen and McBride 1989, Edge and Marcum 1990, Jackson 1991, Adkins 1991) and that big game management is an important economic

Table 1. Comparison of mean economic costs and returns from big game animals on private land in southwestern Montana, as derived from landowner-big game survey, 1989-1990.

Mean estimated impact/landowner	Quantity and Value	Dollars
Cost		
Forage consumption	511 AUMs @ \$11	5,616 ¹ (10561) ²
Damage to hay stack	7.5 tons @ \$60	450 (1040)
Fencing		
Haystack	Labor & Material	119 (451)
Pasture		
Labor	5.3 days @ \$32/day	169 (258)
Material	Posts and wire (unspecified)	113 (146)
Total Cost		6,467 (11227)
Economic Return		
Hunting Access Fees		114 (1025)
Total Return		114 (1025)
Net Estimated Cost		6353

¹Actual dollar value does not correspond to product of quantity and value due to rounding of AUMs to nearest integer.

²Standard deviation.

consideration on private land where big game animals compete with livestock for forage, trample forage and wet soils, and damage crops and fences (Pineo 1985, Gasson 1985, Nielsen et al. 1986, Lacey et al. 1988).

Big game were not uniformly distributed throughout the study area. Slightly over three-quarters of the respondents reported mule deer and whitetail deer use on private land (Table 2). Over half of the respondents reported elk (*Cervus canadensis*) use. Antelope, moose (*Alces alces*), and big horn sheep (*Ovis canadensis*) were reported on a smaller percentage of ranches. By evaluating numbers, animal size, and length of time spent on private lands, it was estimated that big game consumed 511 AUMs per landowner (Table 2). However, the median was 169, indicating that half of the respondents reported less than 169 AUMs of big game use. Elk and mule deer accounted for about 40% and 30% of the total AUMs used, respectively. Because of lower populations and less time spent on private land, consumption by antelope, bighorn sheep, and moose accounted for 13% of the total forage consumed by big game. The cost of forage consumption by big game averaged \$5,616 per landowner (Table 1).

Table 2. Occurrence of big game, big game uses and estimated forage consumption on privately owned acreage in southwestern Montana, as reported by landowners in 1989-1990 survey.

Kind of big game	Occurrence of big game on ranches (% of total)	Total big game use ¹	Mean big game use ²	Forage consumption equivalent ³	Mean animal unit months
Mule deer	78	395,568	869	.17	147
White-tailed deer	79	288,927	635	0.13	83
Elk	56	183,600	404	0.53	214
Antelope	46	221,360	487	0.10	49
Moose	30	7,911	17	0.87	15
Bighorn sheep	4	6,320	14	0.18	2
Total					511

¹ $\sum_{j=1}^{456} \sum_{k=1}^6 a_{jk} X_{jk}$; where j = respondents, k = big game species, a = number of big game and x = number of months on private land.

²Calculated by dividing total big game use by number of respondents.

³Forage demand of big game animals calculated on basis of body weight and expressed in relation to a cow, or 1 animal unit (Stoddard and Smith 1955, p. 192).

Forty-nine percent of the respondents reported big game damage to hay stacks. Of these, 65% experienced a loss between 1 and 10 tons. Since 7% reported damage on more than 51 tons, the loss per respondent averaged 7.5 tons (Table 1). Damage to haystacks was higher in areas with higher big game populations.

Thirty-six percent of respondents fenced their stackyards to reduce big game damage. Cost of labor and material averaged \$119 per respondent (Table 1). Forty-five percent of respondents reporting damage requested, and nearly all of these received, assistance (usually fencing materials) from Montana's Department of Fish, Wildlife, and Parks.

Sixty-four percent of respondents reported that big game damage to pasture fences resulted in additional management cost. Value of labor and materials to repair the fences averaged \$169 and \$113 per landowner, respectively (Table 1).

More than half (54%) of the respondents reported that big game impact other aspects of the ranching operation beyond damage to fences and haystacks. Hay and grain harvest schedules were altered on 35% and 24%, respectively, of the operations. Eighty-seven percent of the landowners reported their income was adversely affected by the "general inconvenience" of big game. Much of the latter costs were associated with providing assistance to hunters. These costs were not quantified in the present study.

Four percent of the respondents earned income from the big game on their private lands. Hunting rights were either leased to outfitters, or hunters were charged an access fee. Most of these landowners earned between \$1,000 and \$5,000 from their hunting enterprises, thus approaching the mean annual cost attributed to big game. A mean income of \$114 per landowner was calculated by dividing the total big game-generated income by 456 landowners. The net economic cost of big game per respondent was \$6,353 (estimated cost of \$6,467 less the \$114 income from marketing hunting opportunities) (Table 1).

The annual cost (\$6,353) of supporting big game on private land is not directly associated with the value of private land for big game hunting. Most of the respondents (77%) allowed big game hunters access to their privately owned lands. They reported a mean of 227 days of big game hunting on their private lands. Nineteen percent of the activity was described as trespass. By dividing \$6,353 (the cost of big game) by the mean number of hunting days (227), the estimated value for each day of big game hunting is \$28. However, the value of big game hunting is frequently reported to be much higher (Duffield 1988, Loomis and Cooper 1988, Brooks 1988). Landowners bear a cost while supporting an activity which provides benefits to the hunting public and community. This economic disparity may explain why many landowners feel that their contributions to big game hunting opportunities are seldom recognized and usually unrewarded (Nielsen et al. 1986, Lacey et al. 1988, Jordan and Workman 1989, McKetta and Bolan 1990, Rimbey et al. 1991). Some landowners deny hunter access to private lands

because they perceive the potential for intentional and unintentional vandalism (to roads, fences, and other facilities), spread of noxious weeds, and fear their susceptibility to liability claims. Some landowners desire compensation for providing wildlife habitat and recreational opportunities (Lacey et al. 1988, McKetta and Bolan 1990). The compensation issue is controversial because sportsmen recognize big game as a public resource (Hadley and Carroll 1986, Headwaters RC&D 1990).

Table 3. Regression of wildlife populations on importance of agricultural income and hectares of privately-owned land, as reported by respondents in landowner-big game survey, 1989-1990.

Big game animal	Number of respondents	Parameters ¹		
		Intercept	Income	Hectares
Mule deer	387	1.6 (31.1) ²	+0.16 (0.42)	+0.05*** (0.01)
Whitetail deer	387	2.45 (20.1)	+0.49 (0.27)	+0.01*** (0.00)
Elk	387	23.6 (50.9)	-0.26 (0.68)	+0.08*** (0.01)
Antelope	387	-3.3 (16.9)	-0.07 (0.23)	+0.02*** (0.00)
Moose	387	1.32 (6.43)	+0.04 (0.09)	+0.004*** (0.00)
Bighorn sheep ³				

¹*** Significant at the 0.05, 0.01, and 0.001 levels, respectively.

²Standard error

³Regression analyses inappropriate due to low number of respondents with big horn sheep.

Intangible Value of Big Game on Private Lands

Half of the respondents indicated that they "enjoyed having big game on their private lands." About three-fifths of the respondents reported hunting by family members, neighbors, and friends as benefits. Nearly one-third of the respondents reported that the big game animals on their private land did not provide them with any intangible (nonmonetary) benefits.

About 75% of the landowners either thought that the presence of big game did not influence or did not know if resale value of private land was influenced by big game. The likelihood of big game increasing or decreasing resale value was reported by 21% and 2% of respondents, respectively.

Factors Influencing Landowner Attitudes Toward Big Game

Respondents who had operated their ranch for more than 10 years were asked about trend of big game populations on their private lands. Over 70% of the respondents indicated that the numbers of whitetail deer and elk had increased on their land during this period (Fig. 4). Data from the Montana Department of

Table 4. Logistic regression of big game impact to haystacks (yes/no), fencing costs (yes/no), and hunting access on percent of income from agriculture, hectares of privately-owned land, and big game populations.¹

Item	Number of observations	Parameters ²			
		Intercept	Income	Hectare	Big game
Damaged haystack	366	1.73*** (0.309) ³	0.021*** (0.0039)	0.00008 (0.00005)	-0.0009*** (0.0002)
Fence hay	353	0.73** (0.273)	-0.0018 (0.0035)	0.00004 (0.00005)	-0.0003* (0.0001)
Hunting access	369	0.47 (0.25)	0.025*** (0.004)	-0.00005 (0.00008)	-0.0002 (0.0003)

¹Coefficients normalized with respect to "no". For example, the 0.021 income estimate for damaged haystacks indicates that as the percent income coming from agriculture increases, the probability of respondents indicating "yes" to the item increased relative to those reporting "no".

²***, **, * Significant at 0.05, 0.01, and 0.001 levels, respectively.

³Standard error.

Fish, Wildlife, and Parks (1986–1987a, 1986–1987b) also indicate that elk have increased 4- to 5-fold in many parts of southwestern Montana. Over half of the respondents also reported increases in mule deer and antelope (Fig. 4). From 3–10% of the respondents reported these big game populations declining. In contrast, big horn sheep and moose populations were thought to be fairly stable over the past 10 years (Fig. 4).

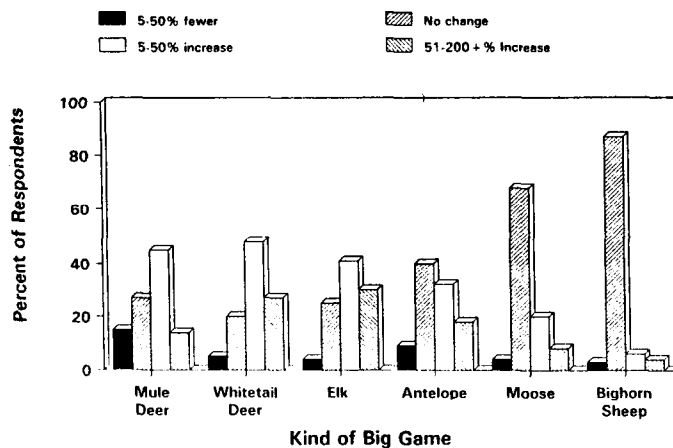


Fig. 4. Change in big game populations on private land in southwestern Montana, as reported by respondents in landowner-big game survey, 1989–90.

Big game populations reported by respondents were correlated with the size of private land holdings. More big game animals were reported by large, rather than by small landowners (Table 3). It is not known whether animal populations were influenced by habitat quality or simply reflect more animals on a larger land base. However, source of income was not significantly correlated with reports of game populations.

Labor required to repair fence was not related to size of operation or source of income. Additional labor was required on ranches with higher big game populations (Table 4). As relative importance of agricultural income and big game population increased, respondents were more likely to report higher fencing costs (Table 4). Size

of operation was not useful for explaining hay damage.

Percentage of landowners fencing hay stacks did not vary among land sizes nor with relative importance of agricultural incomes (Table 4). However, landowners reporting higher big game populations were more likely to fence hay stacks.

Respondents assessed the effect of big game on ranch management goals. About half of the respondents were satisfied with current populations (Fig. 5). However, from 32% to 44% of

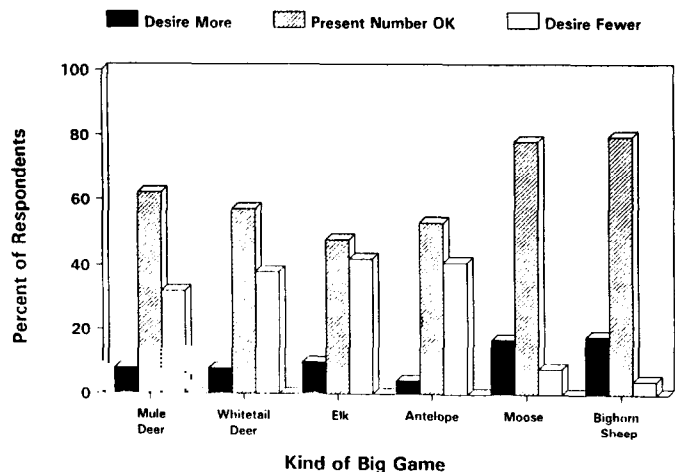


Fig. 5. Landowner satisfaction with big game populations on private land in southwestern Montana, as reported by respondents in landowner-big game survey, 1989–90.

respondents desired fewer elk, antelope, whitetail deer, and mule deer (Fig. 5). For all species except bighorn sheep and moose, the number of respondents desiring fewer big game animals on their land outnumbered those desiring more, by a 4- to 1-margin. In the case of bighorn sheep and moose, there were more respondents desiring population increases than decreases.

As dependency on agricultural income increased, landowners desired fewer mule deer, whitetail deer, and elk (Table 5). As the economic cost of big game increased, the probability also increased

Table 5. Logistic regression of big game satisfaction [(present number OK or desire more)/(desire fewer)] on percent of income from agriculture, hectares of privately owned land, estimated economic cost of big game, reported economic income from big game, and an index of intangible value of big game as reported by respondents in landowner-big game survey, 1989–1990¹.

Big game animal	Number of respondents	Parameter ²					
		Intercept	Income	Hectare	BG Cost	BG Income	BG Intangible
Mule deer	308	1.8*** (0.45) ³	-0.018*** (0.005)	0.0001 (0.00006)	-0.00005*** (0.00002)	-0.0001 (0.0001)	0.33** (0.12)
Whitetail deer	326	1.6*** (0.42)	-0.02*** (0.005)	0.00008 (0.00006)	-0.00005** (0.00002)	0.00009 (0.0001)	0.49*** (0.11)
Elk	236	2.1*** (0.52)	-0.022*** (0.006)	-0.00001 (0.00006)	-0.00004* (0.00002)	0.0000006 (0.0001)	0.21 (0.12)
Antelope	203	1.4** (0.50)	-0.008 (0.006)	-0.00008 (0.00006)	-0.00006** (0.00002)	0.00009 (0.0001)	0.17 (0.13)
Moose	149	14.1 (12.3)	-0.131 (0.12)	-0.0002 (0.0001)	0.00005 (0.00004)	0.0007 (0.0014)	0.50 (0.28)
Big Horn Sheep ⁴							

¹Coefficients are normalized with respect to the categories "desire fewer". Because few respondents indicated that they desired more big game, their responses were combined with those responding "present number OK" to provide a contrast with those desiring fewer big game. For example, the income estimate of -0.018 for mule deer indicates that as the percentage of income coming from agriculture increases, the probability of respondents indicating either "present number ok" or "desire more" mule deer decreased relative to those "desiring fewer" mule deer. Conversely, as the percentage of income coming from agriculture declines, the probability of respondents indicating "present number ok" or "desire more" mule deer increased relative to those "desiring fewer" mule deer.

²*, **, *** significant at the 0.05, 0.01, and 0.001 levels, respectively.

³Standard error.

⁴Regression analyses inappropriate due to low number of respondents "desiring fewer" big horn sheep.

that respondents would indicate that they desired fewer animals. After controlling for source of income and presence of big game, size of operation and income from big game did not influence the degree of satisfaction with any big game species (Table 5). For mule and whitetail deer, respondents were more likely to say they wanted more as the index of intangible benefits increased. The relationship was not significant for antelope, elk, or moose.

Most respondents indicated that big game either has no impact, or a negative impact on forage and crop yields from private land (Table 6). Reported damage was concentrated in alfalfa and grain fields, and was less in woodland. For each land type, as the landowner's dependence on agricultural income increased, landowners

Table 6. Impact of big game on forage and/or crop yield from private land, as described by respondents in landowner-big game survey, 1989–1990.

Kind of land or crop	Number of respondents	Kind of Impact		
		Beneficial	None	Harmful
Percent of respondents ¹				
Alfalfa	292	2 ^a	30 ^a	68 ^a
Grain	220	1 ^a	36 ^a	63 ^a
Pasture	341	2 ^a	44 ^b	54 ^b
Grass hay	296	1 ^a	45 ^b	53 ^b
Range	299	1 ^a	49 ^b	50 ^b
Woodland	202	5 ^b	78 ^c	17 ^c

¹Means within a column followed by the same letter are not significantly different at the 0.05 probability level, as determined using Chi square analyses.

were more likely to respond that big game had a harmful impact on forage and crop yields (Table 7). Landowners who were less dependent on agricultural income were more likely to report no impact or beneficial impact from big game (Table 7). As reported economic cost of big game increased, so did the likelihood that respondents would indicate that the big game was having a "harmful" effect on yields. Size of landholding and income from big game were not useful explanatory variables. As the index to intangible value of big game increased, respondents were more likely to indicate no impact or beneficial impact on pasture and grass hay (Table 7).

Probability of landowners allowing hunters access to private lands increased with the relative importance of agriculture in total income (Table 4). Access to private land was not influenced by size of land nor reported big game numbers.

Number of hunter days was positively associated with ranch size and with big game populations. Number of hunter days was not influenced by relative importance of agricultural income (Table 8).

In summary, big game adversely impact landowner income in Southwestern Montana. Number of respondents desiring fewer big game animals on their private land exceeded the number desiring more animals. Landowners exposed to high big game populations were less tolerant than those exposed to low big game numbers. Landowner attitudes toward big game were influenced by dependency on agricultural income and cost of big game. Income from big game and noneconomic values were less important as explanatory variables. Data indicate that landowners in areas such as

Table 7. Logistic regression of impact of big game on yield of private land [(no effect or beneficial effect on yield)/(harmful effect on yield)] on percent income from agriculture, hectares of privately-owned land, estimated economic cost of big game, reported economic income from big game, and an index of the intangible value of big game.¹

Kind of land	Number of respondents	Parameter ²					
		Intercept	Income	Hectare	BG Cost	BG Income	BG Intangible
Range	257	1.98*** (0.49) ³	-0.022*** (0.005)	0.00003 (0.00006)	-0.0001*** (0.00003)	-0.0016 (0.0002)	0.288 (0.132)
Pasture	292	1.45*** (0.39)	-0.018*** (0.005)	-0.00002 (0.00006)	-0.0002*** (0.00003)	0.0002 (0.0002)	0.38** (0.13)
Grass hay	263	1.33*** (0.44)	-0.018*** (0.005)	0.000004 (0.00006)	-0.00009*** (0.00003)	0.0002 (0.0001)	0.31** (0.12)
Alfalfa	256	1.10** (0.40)	-0.018*** (0.005)	0.00002 (0.00007)	-0.00009** (0.00003)	0.0002 (0.0001)	-0.070 (0.13)
Grain	189	1.9*** (0.55)	-0.03*** (0.006)	0.00006 (0.00007)	-0.00006* (0.00003)	-0.00007 (0.0003)	-0.08 (0.14)
Woodland	179	3.15*** (0.78)	-0.019* (0.008)	0.00006 (0.00008)	-0.00002 (0.00002)	0.0003 (0.0006)	-0.072 (0.178)

¹Coefficients normalized with respect to the category "effect on yield is harmful". For example, the -0.022 income estimate for range indicates that as the percentage of income coming from agriculture increases, the probability of responses indicating "no effect" or "beneficial" effect on yield decreased relative to those reporting "effect on yield is harmful". Because few respondents indicated effect on yield is "beneficial", these responses were combined with those responding "no effect" on yield to provide a contrast with those reporting harmful effects on yield.

²*, **, ***Significant at the 0.05, 0.01, and 0.001 levels, respectively.

³Standard error.

Table 8. Regression of number of hunter days, increased labor, and fencing materials cost on percent of income from agriculture, hectares of privately owned land, and big game populations.

Item	Number of respondents	Parameters ¹			
		Intercept	Income	Hectare	Big game
Hunter days	368	8.96 (21.64) ²	0.189 (0.0289)	0.018*** (0.004)	0.054*** (0.013)
Increased labor days/yr	245	1.12*** (0.104)	0.0005 (0.0012)	-0.000009 (0.00001)	0.0002*** (0.00005)
Fencing & materials costs (days/yr)	247	1.24*** (0.175)	0.005** (0.002)	0.00004 (0.00002)	0.0003*** (0.00008)

¹*, **, ***Significant at the 0.05, 0.01, and 0.001 levels, respectively.

²Standard error.

southwestern Montana may be expected to request a reduction in number of big game animals on private lands.

Literature Cited

- Adkins, R.J. 1991.** An analysis of game damage and game damage complaints in Montana. MS Thesis, Montana State Univ., Bozeman.
- Applegate, J.E. 1981.** Landowner's behavior in dealing with wildlife values. In: Robert T. Dumke, George V. Burger, and James R. March (eds.). *Wildlife management on private lands*, Wisconsin Dep. of Natur. Resourc. Madison, Wisc.
- Austin, D.D., and P.J. Urness. 1987.** Consumption of fresh alfalfa hay by mule deer and elk. *Great Basin Natur.* 47:100-102.
- Brooks, R. 1988.** The net economic value of deer hunting in Montana. Montana Dep. Fish, Wildl. and Parks. Helena, Mont.
- Coleman, J.S. 1990.** Foundations of social theory. Harvard Univ. Press, Cambridge.
- Dillman, D.A. 1978.** Mail and telephone surveys: the total design method. John Wiley and Sons. N.Y.
- Duffield, J. 1988.** The net economic value of elk hunting in Montana. Mont. Dep. Fish, Wildl. and Parks. Helena, Mont.
- Edge, W.D., and C.L. Marcum. 1990.** Elk and cattle on public lands: a new look at an old conflict. *Western Wildlands Summer*: 12-15.
- Gasson, W. 1985.** Capitalizing on Wyoming's game and fish resources. In: J. Powell (ed.) *Holistic Ranch Management Workshop Proc.* Wyo. Agr. Ext. Serv. May 28-30, 1985. Casper, Wyo.
- Gillis, A.M. 1991.** Should cows chew cheatgrass on common lands? *Bio. Sci.* 41:668-675.
- Hadley, K., and N. Carroll. 1986.** From conflict to cooperation. Proc. Montana Landowner's and Sportsmen's conference. April 18-20, 1986. Fairmont Hot Springs, Mont.
- Headwaters, RC&D. 1974.** Headwaters RC&D project plan Montana. Assisted by USDA, Soil Conserv. Serv. and other Cooperating Agencies. Butte, Mont.
- Headwaters, RC&D. 1990.** People working together in the 1990's. A landowners/sportsmen conference to identify common ground. 28 April 1990. Butte, Mont.
- Hosmer, D.W., Jr., and S. Lemeshow. 1989.** Applied logistic regression, John Wiley & Sons.
- Jackson, J.M. 1991.** Impact of elk in Catron County, New Mexico. *Rangelands* 13:287-290.
- Jordan, L.A., and J.P. Workman. 1989.** Economics and management of fee hunting for deer and elk in Utah. *Wildl. Soc. Bull.* 17:482-487.
- Kmenta, J. 1986.** Elements of econometrics. Macmillan Publ. Co., Inc. N.Y.
- Kramer, B.M. 1982.** Legal aspects of use and development of wildlife resources on private lands: Colorado, Kansas, New Mexico, Oklahoma, Texas. *Great Plains Agr. Counc. Wildl. Resour. Comm.* Great Plains Agr. Counc. Pub. 103.
- Krutilla, J.V. 1967.** Some environmental effects of economic development. *Daedalus* 96:1058-1070.
- Lacey, J.R., and J.B. Johnson. 1990.** Livestock grazing on federal lands: a boon to Montana's economy. *Western Wildlands*. 16:23-26.
- Lacey, J.R., S.B. Laursen, J.C. Gilchrist, R.M. Brownson, J. Anzick, and S. Doggett. 1988.** Economic and social implications of managing wildlife on private land in Montana. *Northwest Sci.* 62:1-9.
- Loomis, J., and J. Cooper. 1988.** The net economic value of antelope hunting in Montana. Mont. Dep. of Fish Wildl. and Parks. Helena, Mont.
- McKetta, C., and N. Bolon. 1990.** Idaho landowners' attitudes on hunting and hunters. *Idaho Forest Wildl. and Range Exp. Sta. Focus* 15:3-4.
- Montana Department of Fish, Wildlife, and Parks. 1986-1987a.** Big game survey and inventory, Region 2. Job Prog. Rep., Survey and Inventory Proj. Job I-2 Elk and Deer.
- Montana Department of Fish, Wildlife, and Parks. 1986-1987b.** Big game survey and inventory, Region 3. Job Prog. Rep., Survey and Inventory Proj. Job I-3 (Segment-B) Elk.
- National Wildlife Federation, Montana Wildlife Federation, and Skyline Sportsmen. 1990.** Appeal of the 1990 Plan of Use for the Upper Ruby Cattle and Horse Allotment. Notice of Appeal, Statement of reasons, and request for stay. United States Forest Service, Sheridan Ranger District are appellees.
- Nielsen, D.B., and K. McBride. 1989.** Losses on private land due to big-game animals. *Utah Agr. Exp. Sta., Utah Sci. Summer*.
- Nielsen, D.B., F.J. Wagstaff, and D. Lytle. 1986.** Big-game animals on private range. *Rangelands* 8:36-38.
- Peck, J.M. 1986.** A review of wildlife management. Prentice-Hall. Englewood Cliffs. N.J.
- Pineo, D. 1985.** Wildlife and recreation management on private lands—A guide for Washington. Produced in cooperation with: Washington Farm Bureau, Washington Dairymen-Federation, Washington Cattlemen's Assoc., Washington Forest Protect. Assoc., and Washington Farm Forest. Assoc.
- Rimbey, N.R., R.L. Gardner, and P.E. Patterson. 1991.** Wildlife depredation policy development. *Rangelands* 13:272-275.
- Smith, D.R. 1961.** Competition between cattle and game on elk winter range. *Univ. of Wyo., Agr. Exp. Sta. Bull.* 377.
- Stoddard, L.A., A.D. Smith, and T.W. Box. 1975.** Range management. Third Ed. McGraw-Hill Book Co., N.Y.
- Stoddard, L.A., and A.D. Smith. 1955.** Range management. Second Ed. McGraw-Hill Book Co. N.Y.
- Thomas, J.W. 1984.** Fee-hunting on the public's lands?—An appraisal. *Trans. North Amer. Wildl. Natur. Resour. Conf.* 49:455-467.
- Torell, L.A., and J.P. Doll. 1991.** Public land policy and the value of grazing permits. *W.J. Agr. Econ.* 16:174-184.
- U.S. Census of Agriculture. 1987.** Montana State and County Data. Geographic Area Services. Vol. 1, Part 26. Washington, D.C.
- USDA National Agricultural Statistics Service. 1990.** Agricultural prices: 1989 Summary. *Agr. Stat. Board. Washington, D.C.* p. B-60.

Fee hunting in the Texas Trans Pecos area: A descriptive and economic analysis

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Abstract

Previous studies of fee hunting have focused only on fee-hunting ranches with little consideration given to ranches that choose not to operate fee-hunting enterprises. Our study compares fee-hunting with non-fee-hunting ranches. The most important reasons given for engaging in fee hunting were increased income, trespass control, and prevention of nuisance requests for free hunts. The most important reason offered for choosing not to have fee hunting was to keep the ranch available for hunting by family and friends. The potential exists for a large expansion of private land fee hunting by current non-fee-hunting ranches. Ranchers with fee hunting were more likely to manage the grazing resources, wildlife population, and wildlife habitat than non-fee-hunting ranchers. The typical hunting enterprise in the Texas Trans Pecos area provided a total annual net revenue of about \$7,900. Average annual net grazing returns per livestock animal unit were smaller on fee-hunting ranches but fee-hunting revenue offset the difference. The fee-hunting enterprises also reduced risk by providing a second source of cash returns.

Key Words: range recreation, mule deer, net returns, multiple use.

Wildlife populations are dependent upon the land manager's decisions to provide and maintain suitable habitat. In states with mostly private lands, these populations are dependent upon the private landowner (Dill et al. 1983). Clawson (1978) contends that unless owner-managers perceive some direct economic benefit, they will have little incentive to protect and support wildlife.

As ranching enterprises are adapting to survive in today's economic environment, managers must evaluate all ranch resources, including wildlife populations. These wildlife resources must be economically competitive with other ranch production possibilities for the landowner to commit time, labor, and money to wildlife enhancement (White 1986).

Ranchers examining their options for diversifying are considering the alternative of supplying a quality place to hunt (White 1986). Fee hunting has occurred in Texas since the passage of stringent trespass laws in 1925 (Thomas 1966), which transferred control of access to game to the landowner.

Sargent et al. (1958) gave a general idea of the types of hunting lease arrangements used in Texas and the range of resulting income. They called for research to determine the landowners' actual costs of habitat improvement and preservation, crop losses, and reduced livestock production resulting from increased emphasis on wildlife production. They argued that these costs must be related to expected income from hunting leases and expected satisfactions obtained from the enjoyment of wildlife.

Comprehensive economic studies concerning fee hunting are few (Steinbach 1988). Most of the literature has dealt with fee hunting's

history and rationale (Jordan and Workman 1989). Ramsey (1965) and Forrest (1968) conducted research on white-tailed deer (*Odocoileus virginianus*) fee hunting in Texas. Recent research findings have been reported for Utah (Jordan and Workman 1989); Texas (Steinbach 1988, Glover and Conner 1988, Butler 1990, Butler 1991, Cohen 1991); New Mexico (Morgan 1988); California (Fitzhugh 1988); Oregon (Butler 1990); and Montana (Lacey et al. 1988).

Most of the previous research has described and analyzed the fee hunting situation based on examination of only those ranches with fee hunting. With the exception of Butler (1990) and Butler and Workman (1991), ranches with and without fee-hunting enterprises have not been compared.

Objectives

The objectives of the study were: to describe current fee-hunting enterprises in the Trans Pecos area; to make comparative economic analyses of ranches with and without fee-hunting enterprises; and to make comparative analyses of range management for livestock and wildlife on ranches with and without fee-hunting enterprises.

Methods

The Study Area

The study was conducted in the Trans Pecos area of Texas (Fig. 1). Over 90% of this area is rangeland (SCS Pecos Area Staff 1985).



Fig. 1. The study area, the Texas Trans Pecos.

The Trans Pecos includes 4 vegetative zones: Desert Shrub, Desert Grassland, Mixed Prairie, and Mountain Savannah. Elevation ranges from 671 to 2,667 meters and topography varies from nearly level areas along the Rio Grande and Pecos River flood plains to very steep mountain slopes. Climate is variable depending mostly on elevation. Annual precipitation ranges from about 15 centimeters along the Rio Grande to 46 centimeters in the higher mountains.

Data Collection

Data were collected by mail questionnaires and personal interviews. All data were pertinent to calendar year 1988. Mail questionnaires were sent to all ranch owners and/or operators with at least 200 hectares of rangeland. The mail survey was conducted in early 1989, utilizing the Total Design Method (Dillman 1978). The original mailing lists contained 445 names and addresses. Removal of nondeliverable surveys, deceased individuals, nonrangeland owners, and individuals who had sold their ranches reduced the total population of interest to 347.

Seven percent of these 347 individuals responded by stating that they chose not to complete the questionnaire while 130 (38%) completed questionnaires were received, resulting in a 45% total response rate.

Telephone surveys were conducted to check for nonresponse bias. Ten percent of the nonrespondents (20 individuals) were randomly selected for the follow up survey. A series of key questions from the original mail questionnaires were asked. Based upon an analysis of mean and median values it was concluded that there were no significant differences between the mail questionnaire respondents and nonrespondents. Therefore study results can be considered as representative of the total population.

Personal interviews were conducted in the summer of 1989. Respondents to the mail questionnaire were categorized based upon their responses. Ranchers that indicated they had a huntable population of mule deer (*Odocoileus hemionus*) and a fee-hunting enterprise were placed in the "with fee hunting" group while those with a huntable population of mule deer and no fee hunting were placed in the "without fee hunting" group. Ranchers without a huntable population of mule deer were excluded from the study in order to compare only those ranchers that actually had the option to have mule deer fee hunting.

Ranchers from the with and without fee hunting groups were then randomly selected for personal interviews. Useable interviews were conducted with 25 ranchers with fee hunting and 19 ranchers without fee hunting.

Interview data were used to develop typical enterprise budgets for the 2 groups. These budgets consisted of revenues and costs from the grazing enterprise and hunting enterprise. Grazing revenue consisted of livestock sales and grazing leases. Hunting revenue consisted of only mule deer hunting fees. Only variable costs were considered in the budgets since the decision to operate an enterprise in the short-run is driven by the ability to at least cover variable costs. Owner/manager labor and land costs (mortgage and lease payments) were treated as fixed costs. Net return above variable costs per livestock animal unit was used as a standard of comparison between the 2 groups.

Data Analysis

Frequencies, means, *t*-tests, and likelihood ratio Chi-square (G^2) statistics were computed with the SAS System statistical package. For nominal categorical data the G^2 likelihood ratio test was used to determine if a relationship existed between the variables. Odds ratios (95% confidence interval) were calculated to determine the magnitude and direction of the relationships.

The *t*-test was used (95% confidence level) to test for differences of means of normally distributed data. Data were tested for nor-

mality utilizing the SAS univariate procedure. When a data distribution failed the normality test, data were normalized through log transformation prior to conducting the *t*-test.

Results and Discussion

Mail Questionnaires

Data from the mail questionnaires were used to describe the current fee-hunting enterprises. The average ranch size in the Texas Trans Pecos area was 12,747 hectares (range: 291 to 89,126 ha), consisting of 6,969 hectares (range: 145 to 71,706 ha) owned land, 4,402 hectares (range: 0 to 81,947 ha) leased private land, and 739 hectares (range: 0 to 56,716 ha) leased state land.

Respondents were asked to identify the various sources of ranch income and to indicate the percentage that each source contributed to total gross income. Eighty-five percent of the ranches reported income from livestock sales. Crop sales contributed income to only 4% while mineral lease revenue was reported by 28% of the ranches. Sixteen percent of the ranches received income from leasing rangeland to other ranchers. Fee hunting was a source of income for 52% of the ranches.

The fee-hunting enterprises did not totally exclude non-paying hunters. Sixty-eight percent of ranchers with fee hunting and 84% without allowed family and friends to hunt on their lands. Non-family and nonfriends were allowed to hunt for free on 24% of the ranches without fee hunting. Twenty percent of the ranches with fee hunting reserved some areas of the ranch and/or some period of the hunting season for these nonpaying hunters.

Reasons Ranchers Chose Not to Have Fee Hunting

Nonfee hunting ranchers gave a variety of reasons for choosing not to have fee hunting (Table 1). Most (56%) preferred to keep

Table 1. Reasons given by non-fee-hunting ranchers for choosing not to have a fee-hunting enterprise (%).

	(%)
Prefer to keep ranch for family & friends	56
Don't want hunters on the ranch	29
Ranch is not big enough for fee hunting	27
Not enough wildlife	26
Miscellaneous reasons	15
Added income wouldn't offset the costs	13
Don't need the additional income	7
Tried to, but couldn't get hunters	0
n = 55	

their ranch for hunting by their family and friends. About one-fourth of the ranchers stated that their ranch was too small for a fee-hunting enterprise. This may be a false perception as all ranches in the survey included at least 200 ha. Twenty-six percent reported that they did not have enough wildlife for fee hunting. The mule deer density in the study area varied from 67 to 1,235 ha per deer (Clark 1988). There are some fee-hunting ranches in the low deer density areas, but their fee-hunting enterprises are small.

The number of ranchers with fee hunting may increase in the future since 9% of non-fee-hunting ranchers said they planned to start a fee-hunting enterprise. Thirty-two percent without fee hunting were uncertain about initiating fee hunting in the future.

Reasons Some Ranchers Chose to Have Fee Hunting

Fee-hunting ranchers were asked to indicate the importance of 4 reasons for choosing to have fee-hunting enterprises: to increase income, for trespass control, to prevent hunters from asking for

Table 2. Percentages of ranchers with fee hunting that reported various levels of importance of reasons for having fee hunting.

Reason	Very important	Somewhat important	Not important	Does not apply
	------(%)-----			
Increase income	69	26	2	3
Trespass control	18	34	6	42
Offset forage loss	5	20	25	50
Prevent hunters from asking for free hunts n = 65	32	34	14	20

permission to hunt for free, and to offset of livestock forage to wildlife. As shown in Table 2, increased income was the most important reason followed by prevention of hunters from asking for free hunts.

Description of Fee-Hunting Enterprises

The average fee-hunting ranch was larger than the average non-fee-hunting ranch (Table 3). *T*-tests were conducted after log transformation of the data to achieve normality. The average number of hectares for ranches with hunting enterprises was statistically larger than ranches without hunting enterprises ($t = -4.14$, $df = 127$, $p = 0.0001$). When ranch size was calculated as average total

Table 3. Comparison of ranch characteristics between ranches with and without fee-hunting enterprises in the Texas Trans Pecos.

		with	without
Total hectares	mean	16,525	8,768
	n	66	63
Owned hectares	mean	9,183	4,675
	n	66	63
Total private hectares	mean	13,558	8,677
	n	66	63
Total livestock animal units	mean	440	245
	n	61	58

animal units, the ranches with hunting enterprises were almost twice as large as those without.

Huntability Populations and Fee Hunting

Ranchers were asked to identify the wildlife species present in huntable numbers on their ranch. They were also asked to identify the species for which hunters paid access fees to hunt on their lands.

Table 4. Percentages of Texas Trans Pecos ranchers with huntable populations and with fee hunting by species.

Species	Ranches with huntable population	Ranches with fee hunting
	------(%)-----	
Mule deer	79	35
Antelope	28	17
Javelina	59	12
Scaled quail	89	21
Mourning dove	78	13
Coyote	81	8
Aoudad sheep n = 130	18	9

Table 4 shows the percentages of ranches with huntable populations and the percentage of ranches with fee hunting for each species. Seventy-nine percent of the ranches reported a huntable

population of mule deer. Thirty-five percent had fee hunting for mule deer. Thus some ranchers were not utilizing huntable populations of wildlife on their ranches and these wildlife represent a potential for expansion of fee hunting.

It cannot be assumed that all ranchers with huntable populations could have a fee-hunting enterprise for all these species. Some ranches include animals such as javelina (*Tayassu tajacu*), scaled quail (*Callipepla squamata*), and coyote in their fee-hunting package for mule deer or antelope hunters. Some constraints are beyond their control. Due to variable rainfall in the Texas Trans Pecos region the numbers of mourning dove (*Zenaidura macroura*) or scaled quail may be too variable over time to plan a fee-hunting operation. State game regulations may constrain ranchers' abilities to have a fee-hunting enterprise. For example, antelope permits are allocated to the landowners based upon an annual aerial census and ranchers cannot be sure how many permits, if any, they will receive.

Fee Hunting Gross Revenues

Ranchers with fee hunting were asked to list the prices they charged for the right to hunt various wildlife species. Ranchers charged for hunting rights by the day, week, or hunting season and by the animal for exotic species.

Charging by the season was the most common pricing method for mule deer hunting as 84% of mule deer fee-hunting ranchers priced their hunts by the season. The average season price was \$828 per hunter. The minimum price quoted was \$300 and the maximum was \$1,500. Other ranchers charged by the week for mule deer hunting with an average of \$1,183 per hunter per week. The weekly rates varied from a minimum of \$1,000 to a maximum of \$1,500. The average gross revenue per ranch for mule deer hunting was \$11,245, ranging from \$1,500 to \$37,500.

Twenty-one ranchers charged fees for antelope hunting rights. The average fee was \$570 per hunter per season with a range from \$100 to \$1,000. Fifteen ranchers received fees for scaled quail hunting. Several reported that quail hunting was included as a bonus in their mule deer hunting rights. The average price for one day's quail hunting was \$34 with a range of \$25 to \$50 per day. Season quail hunting rights averaged \$118 per person with a range of \$38 to \$300. Some ranchers sold mourning dove hunting rights that allowed hunting in any area of the ranch while others specified stock watering ponds or windmill sites. Hunting rights for mourning dove varied from \$13 per hunter day to \$1,000 per windmill per season. The average price for a season of dove hunting was \$233 per person.

The aoudad or Barbary sheep (*Ammotragus lervia*) is an introduced exotic that free ranges in many of the mountain ranges of the Texas Trans Pecos. There is no specified hunting season for these sheep and they are considered the private property of landowners where they live. Aoudad sheep hunting is generally sold on a per animal basis. The average price received for an aoudad sheep hunt was \$881 per animal with a range of \$250 to \$2,000.

Facilities and Services

There was a variety of services and facilities provided to hunters. No statistically significant relationship was found between the fee charged and number or combination of facilities and services for mule deer hunting. Table 5 shows the percentage of ranches with mule deer fee-hunting enterprises that provided specific facilities and/or services.

Wildlife Management

Ranchers were asked to list specific wildlife management practices utilized on their ranches. The results are listed in with and without fee-hunting categories (Table 6).

Table 5. Percentages of Texas Trans Pecos ranchers with mule deer fee hunting that provided specific hunter facilities and/or services.

Facility or service provided to hunters	(%)
Guides	36
Meals	20
Horses	16
Vehicles	27
Water	69
Lodging	78
Firewood	53
Firing range	31
Campsite with electricity	22
Campsite without electricity	22
Care of trophy	4
Field dress game	18
Miscellaneous	22
n = 45	

The practices of adjusting livestock numbers, adjusting livestock grazing pattern, and adjusting wildlife harvest were clustered together and the responses of ranchers with and without fee hunting were tested with a likelihood ratio chi-square. A relationship existed ($G^2 = 7.8$, $p = 0.005$) and the data indicated that ranchers with fee hunting were almost 3 times more likely to adjust livestock numbers, adjust livestock grazing patterns, or adjust the wildlife harvest than were ranchers without fee hunting (odds ratio = 2.7, 95% confidence interval = 1.3 to 5.6).

Personal Interviews

Ranch Income Statements

Ranchers were asked to provide revenues and costs from their grazing and hunting enterprises. The grazing enterprise was defined as the ranch-owned cattle operation and/or ranch grazing lease. The hunting enterprise consisted of only mule deer hunting. Revenues from separate enterprises were easily identified; however, enterprise costs were often difficult to identify. While some ranchers had a very detailed accounting system, others had not considered costs on an enterprise basis. In all interviews, ranchers successfully separated costs by enterprise.

The FEEDSTORIS micro-computer program for ranch planning (Tanaka et al. 1987) was used as a guideline to develop ranch income statements from rancher questionnaires. Individual ranch income statements were used to develop typical ranch income statements for ranchers with and ranchers without mule deer fee hunting.

Grazing Enterprise Revenue

As the average fee-hunting ranch was larger than the average ranch without fee hunting, it was not surprising that the average total grazing revenue was larger on fee-hunting ranches (\$87,988) than non-fee-hunting ranches (\$77,693). Based on these data, it was initially hypothesized that there was no difference between the 2 ranch groups in net grazing return above variable cost per livestock animal unit.

Ranches with fee hunting typically ran 300 to 350 animal units of livestock while those without ran 225 to 300 animal units. Average cash operating costs were \$36,927 and \$25,274, respectively, on ranches with and without fee hunting. Grazing enterprise net returns above variable costs were about the same, \$51,061 and \$52,419, respectively, for ranches with and without fee hunting.

Average net grazing returns above variable costs per livestock animal unit were \$150 and \$197, respectively, on ranches with and

Table 6. Wildlife management practices reported by ranchers with and without fee hunting in the Texas Trans Pecos.

	with fee hunting	without fee hunting
	----- (%) -----	
Supplemental feeding	33	40
Adjust livestock numbers	34	22
Adjust livestock grazing pattern	34	27
Provide water	72	68
Adjust wildlife harvest	60	27
Food plots*	5	8
Brush management**	25	16
Range seeding**	12	21
	n = 67	n = 63

*Plots not available to livestock.

**Done with consideration for wildlife.

without mule deer fee hunting. There was a significant difference ($t = 1.65$, $df = 42$, $p = 0.11$) in net grazing return above variable costs per livestock animal unit between the 2 groups. One possible explanation of the higher income per livestock animal unit on non-fee-hunting ranches is that these ranchers have less diversification and concentrate more of their attention, labor, and management on their grazing enterprise.

Hunting Enterprise Revenue

Interviews reported an average mule deer hunting revenue per fee-hunting ranch of \$11,283, confirming the \$11,245 average value resulting from questionnaire data. Typical fees were \$750 to \$1,250 per hunter and the typical hunting enterprise had 8 to 14 fee-paying hunters per mule deer season.

Typical hunting variable costs totaled about \$3,400 per ranch per year. Included were: fuel; insurance (personal liability and casualty on property such as hunting cabins); hired labor (guides, cooks, and general labor during the year); repairs (maintenance on cabins, water developments, gates, fences, and roads); supplies (tires, batteries, and groceries for hunter meals); utilities (cabins, camper-trailer hook-ups, and pumping costs for wildlife water); and other expenses (mail, telephone, predator control, trespass control, attorney and accountant fees, supplemental feed, hunting preserve license, and professional dues).

Average net return above variable costs was \$7,893 per ranch. Fee-hunting ranches had a typical population of 60 to 70 animal units of mule deer. The average net hunting return above variable costs per mule deer animal unit was \$116.

Since fee-hunting ranches had a lower average net grazing return per animal unit of livestock, it was important to determine if added hunting revenue would offset the lower returns per animal unit from a possibly less efficient livestock operation should a rancher consider adding a hunting enterprise. Adding net hunting revenue to the net grazing revenue, total net return above variable costs was \$58,954, and average net return per livestock animal unit was calculated at \$183. This was not significantly different ($t = 0.60$, $df = 41$, $p = 0.55$) from the \$197 per livestock animal unit on the ranches without fee hunting.

If adding fee-hunting could potentially reduce the net return per livestock animal unit, why add a hunting enterprise? One reason is to diversify as a precaution against risk (Castle et al. 1987). The fee-hunting enterprise is a more reliable source of income than the livestock enterprise since a hunting enterprise has an income level that has remained constant (or even increased) over time (Butler 1991).

A second source of cash income is another benefit from fee hunting. Mule deer fee hunting contributed about 10% of total grazing and hunting receipts on fee-hunting ranches. Hunting fees usually consisted of 2 cash payments. The first payment was usu-

ally one-half the total fee and was generally collected 6 to 8 months prior to the hunt. The remainder was usually collected during the hunt. In-kind services, such as road repair work, were sometimes substituted for cash payments. Often the in-kind services had a higher value than the hunting fees.

Two ranchers interviewed had negative net returns above variable costs for their hunting enterprises. Both were relatively new enterprises that provided guides, vehicles, meals, and lodging, and had extensive and expensive deer censuses conducted by consulting wildlife biologists.

Ranchers without fee hunting typically had 30 to 40 animal units of mule deer. Given an average potential net return above variable costs of \$116 per mule deer animal unit, they were foregoing an average increased potential net return of \$3,000 to \$4,000. The fact that fee-hunting ranchers, on the average, earned lower net grazing returns per livestock animal unit does not diminish their hunting returns. It only indicates that, on the average, net grazing returns per animal unit could potentially be increased on the fee-hunting ranches.

Conclusions

The potential for more ranchers to begin a fee-hunting enterprise exists. Since the desire to increase income is the most important reason reported for having a fee-hunting enterprise, it is important that ranchers with fee hunting and those considering the addition of a fee-hunting enterprise evaluate potential management and economic impacts this additional enterprise might have upon existing ranch enterprises.

On the average, fee-hunting ranchers received less net grazing returns per livestock animal unit than did non-fee-hunting ranchers. Ranchers with fee hunting should recognize that the potential for less efficiency in their grazing enterprise exists. Ranchers without fee hunting should recognize that there is a potential to increase total net revenue. Both groups of ranchers should take the necessary management actions to maintain efficiency in all enterprises.

Literature Cited

- Butler, L.D. 1990. Fee hunting on private rangelands in the Texas Trans Pecos and central Oregon. Ph.D. Diss. Utah State Univ., Logan.
- Butler, L.D. 1991. White-tailed deer hunting leases: hunter costs and rancher revenues. *Rangelands* 13:20-22.

- Butler, L.D., and J.P. Workman. 1991. A comparison of fee hunting on private rangelands in central Oregon and the Texas Trans Pecos. p. 269-276. In: L.A. Renecker and R.J. Hudson (eds.) *Wildlife Production: Conservation and Sustainable Development*. Univ. Alaska, Fairbanks.
- Castle, E.N., M.H. Becker, and A.G. Nelson. 1987. Farm business management: the decision-making process. Macmillan, N.Y.
- Clark, T.L. 1988. Performance report. Big game harvest regulations-mule deer. Texas Parks and Wildlife Department, Austin, Tex.
- Clawson, M. 1978. Economic choices of rangeland uses. p. 32-33. In: D.N. Hyder (ed.). *Proc. First Int. Rangeland Congr., Soc. Range Manage.* Denver, Colo.
- Cohen, W.E. 1991. Economic strategies for white-tailed deer and livestock production on central and south Texas ranches. Ph.D. Diss. Utah State Univ., Logan.
- Dill, T.O., J. Menghini, S.S. Waller, and R. Case. 1983. Fee hunting for Nebraska big game: a possibility. *Rangelands* 5:24-27.
- Dillman, D.A. 1978. Mail and telephone surveys: the total design method. John Wiley and Sons, N.Y.
- Fitzhugh, E.L. 1988. What can we learn from California's ranching for wildlife program?, p. 69-73. In: D. Rollins (ed.), *Proc. recreation on rangelands: promise, problems, projections*. Soc. Range Manage. Corpus Christi, Tex.
- Forrest, N.K. 1968. Effects of commercialized deer hunting arrangements on ranch organization, management, costs, and income—the Llano basin of Texas. M.S. Thesis. Texas A&M Univ., College Station.
- Glover, M.K., and J.R. Conner. 1988. A model for selecting optimal combinations of livestock and deer lease-hunting enterprises. *Wildl. Soc. Bull.* 16:158-163.
- Jordan, L.A., and J.P. Workman. 1989. Deer and elk fee hunting opportunities in Utah: economics and management. *Wildl. Soc. Bull.* 17:482-487.
- Lacey, J.R., S.B. Laursen, J.C. Gilchrist, R.M. Brownson, J. Anzick, and S. Doggett. 1988. Economic and social implications of managing wildlife on private land in Montana. *Northwest Sci.* 62:1-9.
- Morgan, C.H. 1988. Fee hunting on private ranches in New Mexico. M.S. Thesis. New Mexico State Univ., Las Cruces, N.M.
- Ramsey, C.W. 1965. Potential economic returns from deer as compared with livestock in the Edwards Plateau region of Texas. *J. Range Manage.* 18:247-250.
- Sargent, F.O., C.C. Boykin, O.C. Wallmo, and E.H. Cooper. 1958. Land for hunters—a survey of hunting leases. *Texas Game and Fish*. September. p. 22-23, 29.
- SCS Pecos Area Staff. 1985. Pecos area resource data. USDA. Soil Conserv. Serv., Pecos, Tex.
- Steinbach, D.W. 1988. Economics and operational characteristics of recreational leasing in the Edwards Plateau and Rio Grande Plains of Texas. Ph.D. Diss., Texas A&M Univ., College Station.
- Tanaka, J.A., L.A. Torell, and J.P. Workman. 1987. FEEDSTORIS: a micro-computer program for ranch planning. *Rangelands* 9:51-55.
- Thomas, J.W. 1966. Deer and the commercialized hunting system in Texas, p. 42-54. In: *Proc. The white-tailed deer: its problems and potentials*. Texas A&M Univ., College Station.
- White, R.J. 1986. Big game ranching in the United States. *Wild Sheep and Goat Int.* Mesilla, N.M.

Biological and physical factors influencing *Acacia constricta* and *Prosopis velutina* establishment in the Sonoran Desert

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Abstract

Over the past century woody plants have increased in abundance on sites formerly occupied by grasslands in the Sonoran Desert. Woody plant invasion has been associated with a multitude of biological and physical factors. This study was conducted to determine temperature, soil, fire, rodent, and livestock effects on the germination and establishment of whitethorn acacia (*Acacia constricta* Benth.) and velvet mesquite (*Prosopis velutina* (Woot.) Sarg.). Optimum germination temperatures for both shrubs ranged from 26 to 31° C, and seedling emergence was greatest from seed sown at 1 to 2 cm depths in sandy loam soil. Merriams kangaroo rats (*Dipodomys merriami*) fed seeds in the laboratory removed seed coats and planted embryos at 2 to 4 cm depths in a sandy loam soil. Prescribed fire killed 100% of seed placed on the soil surface but had no measurable effect on the germination of seed planted at 2 cm. After passage by sheep, about 6% of the *A. constricta* and 13% of the *P. velutina* seeds germinated while after passage by cattle, only 1% of the *A. constricta* and 3% of the *P. velutina* seed germinated. Embryo planting by rodents may improve survival efficiencies for these leguminous shrub seedlings, but seed consumption and passage by sheep and cattle appear to adversely affect seed germination. *Dipodomys merriami*, rather than domestic livestock, may be responsible for the spread of these shrubs in the Sonoran Desert.

Key Words: whitethorn acacia, velvet mesquite, germination, soil texture, planting depth, fire, rodents, cattle, sheep semidesert grasslands, North America

Vegetation changes concern both livestock producers who depend on perennial grass forage and land managers who are responsible for natural resource conservation. Of particular concern is the increasing abundance of native shrubs, such as whitethorn acacia (*Acacia constricta* Benth.) and velvet mesquite [*Prosopis velutina* (Woot.) Sarg.]. Since the 1800's these shrubs have spread from riparian lowlands to adjacent semidesert grasslands (Hastings and Turner 1972, Humphrey 1958). Shrub invasion is followed by a reduction in perennial grass densities and accelerated soil erosion (Cooke and Reeves 1976).

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Griffith (1901), Bryan (1925), Humphrey (1958), and York and Dick-Peddie (1969) documented the shrub invasion and Cable and Tschirley (1961) and Morton et al. (1990) developed technology to chemically and mechanically reduce *A. constricta* and *P. velutina* populations. There are, however, few studies which identify climatic, edaphic, and biological factors that cause vegetation change (Glendening and Paulsen 1955) and address interactions (Brown and Heske 1990). Yet, the development of basic information which identifies processes and process interactions, and their incorporation in management practices, may be the only acceptable approach for reducing shrub populations.

This paper addresses the effects of temperature, soil texture and planting depth, rodents, fire, and livestock on the germination and seedling growth of *A. constricta* and *P. velutina*. Historical observations of range management practices are included, and a discussion centers around strategies to limit shrub invasion through rodent and livestock management.

Materials and Methods

Between August and October 1983 and 1984, seed from *A. constricta* and *P. velutina* trees were collected along the Pantano Wash and the Santa Cruz River in southeastern Arizona. Elevations range from 965–1,000 m and slopes are 2–6%. Soils are recent alluvium, weathered from granitic rocks, and are moderately acid (pH = 6.2–6.9); depths range from 1–5 m (Richardson et al. 1979). Average annual precipitation is 400 mm and has ranged from 140–665 mm in the past 80 years (Sellers and Hill 1974, National Oceanic and Atmospheric Administration 1982).

Seed from the 1983 and 1984 collections were retained as separate lots. *A. constricta* seed were cleaned by hand and *P. velutina* seed were processed through a mechanical thresher. Mean seed weights were 20 ± 3 g for *A. constricta* lots and 45 ± 5 g for *P. velutina* lots. Chipped, distorted, and light seed were omitted and only apparently undamaged seed used in the study.

Temperature Effects on Germination

A hard pericarp surrounds the endosperm of the seed of these leguminous shrubs, and pericarp scarification is necessary to insure germination (Glendening and Paulsen 1955). Twenty scarified seed of either *A. constricta* or *P. velutina* were placed on 2 layers of Whatman's #3 filter paper in a plastic petri dish. Approximately 10 ml of distilled water was added and seed germinated at 16, 21, 26, 31, or 38° C on a thermogradient plate (Larsen 1962). These temperatures were selected because soil temperature at 2 cm usually varies from 16–38° C in spring, summer, and fall at 2 locations in the Chihuahuan and 3 locations in the Sonoran deserts (unpublished data, USDA-ARS, Tucson, Ariz.). Germination was considered complete when the seed radicle was 2 cm in length.

Dishes were arranged in a stratified randomized complete block design because temperature was constant across the thermogradient plate. One petri dish was used for each species at each temperature and the experiment was repeated 4 times for seed collected in 1983 and in 1984. Total germination was determined by accumulating the number of germinated seed over the 14-day period. Germination values for the respective shrubs were compared at the same rate for 5 temperatures with analysis of variance (ANOVA). When F-values were significant ($P \leq 0.05$), data were tested for normality and variance homogeneity, and means were separated with a Tukey's HSD test (Sokal and Rohlf 1981).

Planting Depth and Soil Textural Effects on Emergence and Seedling Growth

Soil with textural characteristics common to desert regions of the Southwestern United States and Northern Mexico (Hendricks 1985) were collected in southeastern Arizona. These soils were Pima silty clay loam, Sonoita silty clay loam, and Comoro sandy loam. The soils are classified as thermic Typic Torrifluvents. The clay content of both the Pima and Sonoita silty clay loam soil varies from 20–30%; however, the clay fraction of the Pima is 60% montmorillonite and the Sonoita is 80% kaolinite (Cox and Martin-R. 1984).

Soils were collected from 0–15 cm depths, screened to 1 cm to remove rocks, thoroughly mixed, and added to 15- by 15-cm tapered plastic pots to 6, 10, 12, 13, and 14 cm depths above the pot base. Ten pure live seed (scarified) of 1 shrub species were sown on the surface in each pot. Soils were added to 14 cm depths in all pots; thus, seed were planted at 0, 1, 2, 4, and 8 cm depths below the soil surface. Pots were sub-irrigated with distilled water continuously for 7 days, and thereafter for 8 hours on days 14 and 21. Sub-irrigation was used to minimize soil disturbance and seed movement, and surface soil was moist for at least 72 hours after each sub-irrigation.

Glasshouse relative humidity varied from 58–80%, and temperatures ranged from 23–32° C. Day length was 12 hours and no supplemental light was added. Twenty-five percent Hoagland's solution was added with distilled water on days 7, 14, and 21.

Seedling emergence was considered complete when the first leaf was 2 cm above the soil surface, and the seedling radicle had penetrated the soil surface in pots where seed were surface sown. Emergence was measured 28 days after the experiment began.

Pots were arranged in a randomized complete block design, with a 3 by 2 by 5 factorial arrangement. There were 6 blocks and each contained 3 soils, 2 species, and 5 planting depths. The experiment was repeated twice: once for seed collected in 1983 and once for seed collected in 1984. Seed seedlings did not emerge from all depths in the 3 soils, data were arcsin transformed and subjected to ANOVA with unequal sample numbers. When F-values were significant ($P \leq 0.05$), means were separated with Tukey's HSD Test (Sokal and Rohlf 1981).

Kangaroo Rat Predation Effects on Germination

In August 1985 and September 1986, 6 *D. merriami* males were trapped in a creosotebush (*Larrea tridentata* (DC.) Cov.)-*A. constricta*-*P. velutina* community on the Santa Rita Experimental Range in southeastern Arizona. Animals were transferred to a holding room and a series of feeding trials conducted to determine (1) the quantity of seed coat removal and its effect on seed germination and (2) depths in soil where seed are most frequently planted.

Six *D. merriami* were individually housed in 40- by 40- by 15-cm wire cages. Cages were fitted with 39- by 39- by 10-cm pans, and pans filled with sandy loam soil. On 5 consecutive nights, 200 seed of either *A. constricta* or *P. velutina* were provided to each rat. Each morning soil was removed from the pan at 1-cm intervals and

passed through a 5-mm sieve. Seed were separated, with a dissecting scope, in 3 categories: (1) no seed coat removal, (2) partial seed coat removal (teeth marks), and (3) complete seed coat removal. Remaining unburied seeds were counted.

A. constricta or *P. velutina* seed from the 3 categories were placed on 2 layers of Whatman's #3 paper in a plastic petri dish and germinated in distilled water at 30° C for 7 days. Hard seed were scarified on day 8 and observed for an additional 6 days. Germination was considered complete when the seed radicle was 2 cm in length.

The experiment was repeated twice for seed collection in 1983 and in 1984. Total germination was determined by accumulating the number of germinating seed in each category over the 14-day period. The 6 *D. merriami* were considered blocks and the 4 feeding trials were replications. Germination values for the respective shrubs were compared at the 3 levels of seed coat removal with ANOVA. When F-values were significant ($P \leq 0.05$) means were separated with a Tukey's HSD Test (Sokal and Rohlf 1981).

Fire Effects on Germination

A 6-ha stand of dense, shrub-free Lehmann lovegrass (*Eragrostis lehmanniana*) on the Santa Rita Experimental Range was fenced to exclude livestock. Twelve 15- by 15-m plots were established in January 1984, with 4 plots in each of 3 blocks. One plot in each block was randomly selected and *E. lehmanniana* was harvested at the soil surface in ten 25- by 25-cm quadrats. Above-ground biomass was separated into either standing (live and dead) or litter components. Biomass was weighed, dried at 105° C for 48 hours and reweighed. Dry weight (fuel load) and fine fuel water content (expressed as a percent of dry weight) were averaged by plot.

Prior to burning, 1 plot in each block was randomly selected, and all plots were divided into 4 equal quarters. In each quarter a 25- by 25-cm frame was centered over a randomly selected point. At the 4 corners, 25 seed of either *A. constricta* or *P. velutina* were either sown on the soil surface or hand planted at about 2 cm beneath the soil surface. In each plot there were a total of 100 seeds of each species at 2 soil depths. Seed were planted or sown 5–10 min. before the burn. Three plots were burned with a headfire in mid-June 1984, 1985, 1986, and 1987; no plot was reburned during the experiment.

Mid-June burning was selected because soils are dry, most above-ground biomass is dead, and daytime temperatures exceed 38° C (Cox 1988). During this period lightning frequency peaks (Osborn 1983) and naturally occurring fires would be most common (Hastings and Turner 1972). The time from fire ignition to total herbage removal was recorded by plot. During each burn, wind speed and air temperature were recorded at 10-s intervals. Near each random point, 2 thermocouples were installed to measure soil temperature extremes at 0 and 2 cm depths.

Following the burn, seed of either *A. constricta* or *P. velutina* collected at the 2 depths were placed on 2 layers of Whatman's #3 paper in a plastic petri dish, and germinated in distilled water at 30° C for 7 days. Hard seed were scarified on day 8 and observed for an additional 6 days. Again germination was considered complete when the seed radicle was 2 cm in length.

The experiment was repeated twice for seed collected in 1983 (1984 and 1985 burns) and twice for seed collected in 1984 (1986 and 1987 burns). Total germination was determined by accumulating the number of germinating seed at each depth over the 14-day period. Experimental design was a randomized complete block with 3 replications in 4 years. Because seed on the soil surface failed to germinate, germination of seed planted at about 2 cm was subjected to ANOVA. When F-values were significant ($P \leq 0.05$) means were separated with Tukey's HSD Test (Sokal and Rohlf 1981).

Livestock Digestion Effects on Germination

Four Suffolk ewes (*Ovis aries*) and 4 Angus cows (*Bos taurus*) were fed in individual feeding cages at the University of Arizona-Campbell Avenue Farm. Animals had free access to water, and each group was fed a predetermined daily quantity of alfalfa cubes; ewes received 3.2 kg/day and cows received 6.4 kg/day. During four 15-day feeding trials, animals were fed either *A. constricta* or *P. velutina* seed. On day 1, each ewe received 1,000 *A. constricta* seed, and on day 30, 3 of the same ewes and a substitute received 1,000 *P. velutina* seed. On day 15, each cow received 2,500 *P. velutina* seed, and on day 45, the same cows received 2,500 *A. constricta* seed. Seed were mixed in a 1.5 kg ration of milo and sorghum (50:50).

Fecal collections were made twice daily for 13 days. After each collection ewe feces were soaked for 15 min. in tap water and massaged with a wooden spatula, and cow feces were soaked for 30 min. and liquefied in a commercial dough blender. Fluid was passed through 10-, 5-, and 2-mm sieves, and sieve surfaces sprayed with tap water to remove herbage fragments. Seed were separated, with a dissecting scope, in 3 categories: (1) no seed coat removal, (2) partial seed coat removal (chips), and (3) complete seed coat removal. Some seed in each category had imbibed (swollen) moisture during passage through livestock, hence, a second separation of each category.

Hard and imbibed *A. constricta* and *P. velutina* seed from the 3 categories were placed on 2 layers of Whatman's #3 paper in a plastic petri dish and germinated under the conditions described in the 2 preceding studies. Germination was considered complete when the seed radical was 2 cm in length.

The experiment was conducted in 1985. The 1983 and 1984 seed lots, for the respective species, were combined to obtain adequate seed numbers. Experimental design was a split-plot, animals were main plots, and shrub seed were subplots. Data were subjected to repeated measures ANOVA. When F-values were significant ($P \leq 0.05$) means were separated with a Tukey's HSD Test (Sokal and Rohlf 1981).

Results and Discussion

Temperature Effects on Germination

Germination characteristics of *A. constricta* and *P. velutina* seed collected in 1983 and 1984 were similar ($P \leq 0.05$) and the data were pooled (data not shown). For both species seed germination began on day 5 at 26 and 31° C, and on day 9 at 21 and 38° C. Seed of *A. constricta* began to germinate on day 12 at 16° C and at the same temperature seed of *P. velutina* failed to germinate in 14 days (Table 1).

Table 1. Mean germination (%) of *Acacia constricta* (A.C.) and *Prosopis velutina* (P.V.) seed at 5 constant temperatures during a 14-day germination trial.

Species	Day	Temperature (° C)				
		16	21	26	31	38
		----- % -----				
A.C.	7	0	0	89a ¹	85a	0
P.V.		0	0	81a	95a	0
A.C.	14	27d	54c	98a	97a	34d
P.V.		0	77b	95a	95a	15d

¹Means within days are not significantly different ($P \leq 0.05$) when followed by the same letter.

Optimal germination of *A. constricta* and *P. velutina* occurred at 26 and 31° C, and percentages declined above or below these temperatures (Table 1). Cotyledons and radicles of seed germination at 21, 26, and 31° C appeared healthy and normal. At 16° C, 10% of the *A. constricta* seedlings had either chlorotic or wrinkled

cotyledons, whereas, *P. velutina* failed to germinate after 14 days. At 38° C, 15% of *A. constricta* and 65% of *P. velutina* seedlings had stunted radicles and black abscesses on the root tip.

Germination of *A. constricta* across all temperatures suggests that this species may be adapted over a wider environmental gradient than *P. velutina*. Thus, *A. constricta* might be expected to germinate following both cool- and warm-season moisture. Cox et al. (1987) suggested *P. velutina* would most likely germinate in August when rainfall amounts peak and surface soil temperatures decline from 40 to 30° C. These germination characteristics may partially explain the abundance of *A. constricta* at higher elevations (1,000–1,500 m) and *P. velutina* at lower elevations (900–1,000 m) in the Sonoran desert (Hastings and Turner 1972).

Planting Depth and Soil Textural Effects on Emergence and Seedling Growth

There were no differences ($P \leq 0.05$) in *A. constricta* and *P. velutina* seed lots, thus data were pooled (data not shown). Germination of seed placed on the soil surface averaged 95–100% for the 2 species. Radicles of *A. constricta* seedlings grew horizontally and they died within 48 hours because radicles were unable to penetrate soil surfaces. About 20% of the *P. velutina* radicles penetrated the soil surfaces. *P. velutina* radicles, however, elevated the endosperm above the surface soil, and seedlings with exposed radicles died in 3 to 6 days.

A. constricta seedlings failed to emerge when planted at any depth in the silty clay loam (Sonoita and Pima) soils, and emergence was less than that of *P. velutina* at all depths in the sandy loam (Comoro) soil (Table 2). Radicles of *A. constricta* seedlings

Table 2. Mean emergence (%) of *Acacia constricta* (A.C.) and *Prosopis velutina* (P.V.) seed sown at 5 depths in 3 soils after a 14-day trial.

Planting depths (cm)	Soils and species					
	Comoro		Sonoita		Pima	
	A.C.	P.V.	A.C.	P.V.	A.C.	P.V.
0	0	0	0	0	0	0
1	77b ¹	100a	0	37d	0	67bc
2	57c	95a	0	42d	0	57c
4	10	50cd	0	0	0	0
8	0	15e	0	0	0	0

¹Means followed by the same letter in both columns and rows are not significantly different ($P \leq 0.05$).

excavated from Sonoita and Pima soils were similar in amount and distribution to those from Comoro soils. In the coarse-textured Comoro soil, the epicotyl and cotyledons of emerging *A. constricta* seedlings were protected by the seed coat. Excavations of the 2 fine-textured soils, and the greater depths in Comoro soil, suggest that the seed coat was shed before emergence. Seed coat removal before emergence appears to damage either the epicotyl or hypocotyl and limit *A. constricta* survival. In contrast, seed coat removal before emergence did not appear to effect *P. velutina* seedlings.

Emergence of *A. constricta* and *P. velutina* seedlings peaked between days 6 and 9, and seedling recruitment appeared complete by day 11. However, additional *P. velutina* seedlings emerged when sub-irrigation was discontinued. As soils dried, 3 cm deep cracks appeared in Pima soil. Between days 13 and 15, seedlings from *P. velutina* seed planted at 1 and 2 cm depths emerged from cracks. *A. constricta* seedlings apparently died before the Pima soil cracked because additional seedlings failed to emerge after day 11.

Kangaroo Rat Predation Effects on Germination

Seed collection years, 1983 and 1984, had no apparent effects ($P \leq 0.05$) on the germination characteristics of *A. constricta* and *P.*

Table 3. Effects of seed coat removal by *Dipodomys merriami* on mean germination (%) of *Acacia constricta* (A.C.) and *Prosopis velutina* (P.V.). Numbers in parentheses indicate total germination after further scarification.

Species	Seed coat removal		
	None	Partial	Complete
	----- % -----		
A.C.	59b (95)	92a ¹	95a
P.V.	35c (98)	57b (78)	85a

¹Means followed by the same letter in both columns and rows are not significantly different ($P \leq 0.05$).

velutina seed fed to *D. merriami* in 1985 and 1986. As the amount of seed coat removal increased, germination increased (Table 3) and the time to germination decreased. Embryo's with all seed coat removed began to germinate in 24 hours and germination was complete in 60 hours. Nonscarified seed germinated between days 4 and 6.

During nights 1 and 2, *D. merriami* consumed 30% (60 ± 43) of the 200 seed and planted 10% (22 ± 8). The trend reversed on nights, 3, 4, and 5, and for health concerns we provided fleshy fruits on night 5. Approximately 87% of the lighter *A. constricta* seed were excavated between 2 and 3 cm depths, while 84% of the heavier *P. velutina* seed were found between 3 and 4 cm depths (Table 4). Our most important finding is that 90% of the seed

Table 4. Mean quantities of *Acacia constricta* (A.C.) and *Prosopis velutina* (P.V.) seed buried at several soil depths by *Dipodomys merriami* in laboratory feeding trials.

Depth (cm)	Species	
	A.C.	P.V.
	----- number -----	
1	3	5
2	80	20
3	50	100
4	10	58
5	5	3
6-10	2	2

buried between 2 and 4 cm had the seed coat completely removed and 85 to 95% of the seed germinated in less than 60 hours (Table 3).

Fire Effects on Germination

Aboveground phytomass (standing biomass plus litter) was highly variable among burned plots during the 4-year study, and varied from 3,890 to 5,600 kg/ha (Table 5). Fuel water content varied from 10 to 55%, and the forage in each plot was consumed by the fire in about 44 sec. Ambient air temperatures were 1 to 4° C cooler than temperatures at 2 cm depths during the fire. During the

12 fires, peak surface temperatures near the seed varied from 380° C in June 1987 to 535° C in June 1986.

Fire blackened the seed coat of surface sown *A. constricta* and *P. velutina* seed, and after scarification seed failed to germinate. In contrast, the fire had no effect on germination of seed sown at 2 cm soil depths; mean germination of nonscarified *A. constricta* and *P. velutina* was 55 and 33%, respectively. Germination of scarified seed exceeded 90% for both species.

Livestock Digestion Effects on Germination

Total recovery of ingested *A. constricta* and *P. velutina* seed varied from 2-8% in sheep and from 28-31% in cattle (Table 6). There was considerable variability in seed passage, but in most instances viable seed were passed before day 8 by sheep and day 9 by cattle. Seed passage by sheep and cattle primarily occurred on days 1 through 5, a pattern previously reported by Burton and Andrews (1984) and Simao Neto et al. (1987).

Seed passage in this study is about half that reported by Fisher (1947) and Glendening and Paulsen (1955). Differences may occur because we removed seed damaged by insects and light seed before feeding. Differences may also be attributed to livestock class, and passage rates of caged versus free roaming livestock.

The recovery of the smaller *A. constricta* seed from sheep was about 3 times greater than the larger *P. velutina* seed: 82 versus 25, respectively. Since *A. constricta* (770 seed) recovery from cattle was similar to *P. velutina* (703), seed size had no apparent effect on passage. The greater seed recovery from cattle compared to sheep is related to initial mastication and rumination. We observed that cattle chewed on seed of both shrubs for 15 to 45 sec. before swallowing. Sheep, in contrast, chewed on smaller *A. constricta* seed for 2 to 3 min. and larger *P. velutina* seed for 5 to 7 min. The passage of ingesta from the rumen is determined by particle size reduction during mastication and rumination (Van Soest 1982). Since the reticulo-omasal orifice in cattle is larger than in sheep, more large particles can be expected to pass from cattle. Hence, the passage of more seed by cattle (Table 6).

Of the 1,000 seed of each species ingested by sheep, only 56 (5.6%) *A. constricta* and 13 (1.3%) *P. velutina* seed germinated after passage through the digestive tract (Table 6). Of the 2,500 seed ingested, only 48 (1.9%) *A. constricta* and 81 (3.2%) *P. velutina* seed germinated after passage through cows. Considering the number of viable seed that survive after passage through cows and sheep, there is no reason to believe that germination is enhanced by domestic livestock (Janzen 1981).

D. merriami may remove hard seed from livestock dung (Reynolds 1958). Although most seed are eaten, the few remaining seed, with the seed coat removed, might be planted at 1 to 4 cm. This process may aid in the establishment of recruitment trees which serve as collection and dispersal centers (Brown and Archer 1987, 1989).

Conclusions

Prior to 1890, *A. constricta* and *P. velutina* commonly occurred

Table 5. Fuel characteristics, burn time, and environmental conditions during burning in summers of 1984, 1985, 1986, and 1987.

Year	Fuel load		Fuel water content	Burn time	Wind speed	Temperature (Max) during burn		
	Standing	Litter				Soil		
						Air	Surface	2cm
	----- kg/ha -----		(%)	(sec)	(km/h)	----- ° C -----		
1984	1,890	2,000	10	35	10	30	480 ± 50	33
1985	4,350	1,205	30	45	10	29	410 ± 90	31
1986	3,950	900	25	45	10	31	470 ± 65	32
1987	3,600	1,600	55	50	10	29	390 ± 110	33

Table 6. Number of imbibed and hard *Acacia constricta* (A.C.) and *Prosopis velutina* (P.V.) seed excreted from sheep (1,000 seed fed/animal) and cows (2,500 seed fed/animal). Values in parentheses represent the actual number of germinating seed.

Time after feeding seed	Sheep				Cattle			
	Imbibed seed		Hard seed		Imbibed seed		Hard seed	
	A.C.	P.V.	A.C.	P.V.	A.C.	P.V.	A.C.	P.V.
Days	----- No. -----							
1	1 (1)	2 (1)	9 (8)	0	10 (2)	4 (3)	6 (5)	1 (1)
2	4 (2)	3 (0)	23 (18)	2 (2)	310 (25)	378 (26)	15 (8)	11 (10)
3	4 (3)	1 (1)	10 (9)	4 (3)	250 (20)	124 (9)	25 (10)	22 (20)
4	2 (1)	0	10 (8)	4 (3)	86 (6)	72 (4)	16 (7)	6 (5)
5	0	0	7 (6)	3 (2)	20 (2)	17 (2)	9 (8)	15 (14)
6	0	0	3 (3)	4 (3)	5 (1)	8 (2)	5 (4)	9 (9)
7	0	0	4 (2)	2 (0)	0	4 (0)	9 (4)	4 (4)
8	0	0	3 (2)	0	0	4 (0)	4 (2)	13 (11)
9	0	0	2 (0)	0	0	3 (0)	0	6 (6)
10	0	0	0	0	0	1 (0)	0	1 (1)
11	0	0	0	0	0	0	0	0
12	0	0	0	0	0	0	0	0
13	0	0	0	0	0	0	0	0
Total	11 (7)	6 (2)	71 (56)	19 (13)	681 (56)	615 (46)	89 (48)	88 (81)

as trees and their populations were largely limited to riparian zones (Hastings and Turner 1972). Seed dispersal occurred when floods carried seed downstream but frequent fires killed seed and seedlings. Humphrey (1958) was one of the few ecologists who regarded fire as an evolutionary force in semiarid Sonoran desert grasslands. Recent interpretations of historical evidence (Bahre 1985) supports Humphrey's hypotheses that large, destructive wildfires burned at 2-3-year intervals between 1859 and 1890. These uncontrolled fires were feared by federal land managers, who promoted overgrazing by domestic livestock as a tool to reduce timber losses by wildfire (Leopold 1924).

Our data suggest that increases in *A. constricta* and *P. velutina* during the past century may be attributed to a multiplicity of natural and man-made causes. The failure of surface sown seed to germinate after fire indicates wildfire was important in limiting shrub distribution on former semidesert grasslands. The inability of surface sown seed to establish on bare soil (Table 2), the destruction of most viable seed during mastication and passage through cattle and sheep (Table 6), and seed coat removal and seed embryo planting by *D. merriami* at depths ideal for seedling emergence in soil (Tables 2 and 3) do not support the commonly accepted belief that domestic livestock were solely responsible for shrub invasion. It could be argued that domestic livestock act primarily as seed predators, while *D. merriami* functions as predator, distributor, and planter. The identification of *D. merriami* as a primary mechanism responsible for maintaining and increasing shrub populations (Reynolds 1958) is supported by studies which show similar shrub populations in grazed and nongrazed enclosures where kangaroo rat populations were uncontrolled (Smith and Schmutz 1975, Wright 1982).

Little work has been done to identify the processes responsible for brush invasion. Research has emphasized chemical and mechanical treatments designed to reduce shrub populations and seed introduced grasses to restore grassland productivity (Stoddart et al. 1975). Today, the use of herbicides is less acceptable, mechanical treatments are costly, and there is more concern over introduced grasses. Hence there is a growing need to develop environmentally sound strategies to limit shrub encroachment on existing grasslands.

Because the studies presented in this paper were conducted in the laboratory, greenhouse, and at a few field locations, the results may not be applicable over vast areas. The results do suggest that frequent fires before 1890 probably controlled the abundance of

both species. The abundance of cattle and sheep after 1890, and their preference for pods of leguminous species, contributed to seed spread from lowland to upland areas (Glendening and Paulsen 1955). As livestock numbers increased, perennial forage crop declined and rat densities increased on semidesert grasslands (Norris 1950). Rats harvested seed from trees and livestock dung piles and stored seed at ideal planting depths. The number of germinating leguminous seed is directly related to rat populations, and rats may transport seed 30 m from the seed source (Reynolds 1958).

Cattle and sheep grazing is known to affect plant species composition, but vegetation change has yet to be detected when their numbers are reduced or excluded (Barns 1982, Stoddart et al. 1975). In contrast, kangaroo rat exclusion may be followed by perennial grass establishment, plant litter accumulation, soil disturbance reductions, and the return of a semidesert grassland ecosystem (Brown and Heske 1990). Selective foraging by kangaroo rats for large seed (Brown et al. 1979) and their apparent inability to locate all seed caches (Glendening and Paulsen 1955) appear to be, at least partially, responsible for the shrub invasion in semidesert grasslands.

Literature Cited

- Bahre, C.J. 1985. Wildfire in southeastern Arizona between 1859 and 1890. *J. Desert Plants* 7:190-194.
- Barns, D.L. 1982. Management strategies for the utilization of Southern African savanna. In: Huntley, B.J., and B.H. Walker (eds.), *Ecology of tropical savannas*. Springer-Verlag, N.Y.
- Brown, J.H., O.J. Reichman, and D.W. Davidson. 1979. Granivory in desert ecosystems. *Annu. Rev. Ecol. Syst.* 10:201-227.
- Brown, J.H., and E.J. Heske. 1990. Control of a desert-grassland transition by a keystone rodent guild. *Science* 250:1705-1707.
- Brown, J.R., and S. Archer. 1989. Woody plant invasion of grasslands: establishment of honey mesquite (*Prosopis glandulosa* var. *glandulosa*) on sites differing in herbaceous biomass and grazing history. *Oecologia* 80:19-26.
- Brown, J.R., and S. Archer. 1987. Woody seed dispersal and gap formation in a North American subtropical savanna woodland: the role of domestic herbivores. *Vegetatio* 73:73-80.
- Bryan, K. 1925. Date of channel trenching (arroyo cutting) in the arid Southwest. *Sci.* 62:338-344.
- Burton, G.W., and J.S. Andrews. 1948. Recovery and viability of seeds of certain southern grasses and lespedeza passed through the bovine digestive tract. *J. Agr. Res.* 76:95-103.
- Cable, D.R., and F.H. Tschirley. 1961. Responses of native and introduced grasses following aerial spraying of velvet mesquite in southern Arizona. *J. Range Manage.* 14:155-159.

- Cooke, R.U., and R.W. Reeves. 1976. Arroyos and environmental change in the American Southwest. Clarendon Press, Oxford.
- Cox, J.R. 1988. Seasonal burning and mowing impacts on *Sporobolus wrightii* grasslands. J. Range Manage. 41:12-15.
- Cox, J.R., R.D. Madrigal, and G.W. Frasier. 1987. Survival of perennial grass transplants in the Sonoran Desert of the Southwestern U.S.A. Arid Soil Res. Rehab. 1:77-87.
- Cox, J.R., and M.H. Martin-R. 1984. Effects of planting depth and soil texture on the emergence of four lovegrasses. J. Range Manage. 37:204-205.
- Fisher, C.E. 1947. Present information on the mesquite problem. Texas Agr. Exp. Sta. Pro. Rep. 1056.
- Glendening, G.E., and H.A. Paulsen. 1955. Reproduction and establishment of velvet mesquite as related to invasion of semidesert grasslands. USDA-Forest Serv., Tech. Bull. 1127. Gov. Print. Off., Washington, D.C.
- Griffith, D. 1901. Range improvement in Arizona. USDA-Bureau of Plant Industry, Bull. 67. Gov. Print. Off., Washington, D.C.
- Hastings, J.R., and R.A. Turner. 1972. The changing mile. Univ. of Arizona Press, Tucson.
- Hendricks, D.M. 1985. Arizona soils. Univ. of Arizona Press, Tucson.
- Humphrey, R.R. 1958. The desert grassland. Bot. Rev. 24:193-253.
- Janzen, D.H. 1981. *Enterolobium cyclocarpum* seed passage rate and survival in horses, Costa Rica pleistocene seed dispersal agents. Ecology 62:593-601.
- Larsen, A.L. 1962. Two-way thermogradient plate for seed germination research. USDA-ARS., ARS 51-41.
- Leopold, A. 1924. Grass, brush, timber and fire in southern Arizona. J. Forest. 22:1-10.
- Morton, H.L., F.A. Ibarra-F., M.H. Martin-R., and J.R. Cox. 1990. Creosotebush control and forage production in the Chihuahuan and Sonoran Deserts. J. Range Manage. 43:43-48.
- National Oceanic and Atmospheric Administration. 1982. Climatology of the United States (Arizona). Asheville, N.C.
- Norris, J.J. 1950. Effect of rodents, rabbits and cattle on two vegetation types in a semidesert rangeland. New Mexico Agr. Exp. Sta. Bull. 353.
- Osborn, H.B. 1983. Precipitation characteristics affecting hydrologic response of Southwestern rangelands. USDA-ARS. ARM-34.
- Reynolds, H.G. 1958. The ecology of Merriam kangaroo rat (*Dipodomys merriami* Mearns) on grazing lands of southern Arizona. Ecological Mono. 28:11-127.
- Richardson, M.L., G.D. Clemmons, and J.C. Walker. 1979. Soil survey of Santa Cruz and parts of Cochise and Pima Counties, Arizona. National Cooperative Soil Survey, USDA, Soil Conserv. Serv., Forest Serv., and Arizona Agr. Exp. Sta., Washington, D.C.
- Sellers, W.D., and R.H. Hill. 1974. Arizona climate (1931-1972). Univ. of Arizona Press, Tucson.
- Simao Neto, M., R.M. Jones, and D. Ratcliff. 1987. Recovery of pasture seed ingested by ruminants. I. Seed of six tropical pasture species fed to cattle, sheep and goats. Aust. J. Exp. Agr. 27:239-246.
- Smith, D.A., and E.M. Schmutz. 1975. Vegetative changes on protected versus grazed desert grassland ranges. J. Range Manage. 28:453-458.
- Sokal, R.R., and F.J. Rohlf. 1981. Biometry. Freeman and Co., San Francisco.
- Stoddard, L.A., A.D. Smith, and T.W. Box. 1975. Range Management. McGraw-Hill, San Francisco.
- Van Soest, P.J. 1982. Nutritional ecology of the ruminant. O and B Books, Portland.
- Wright, R.A. 1982. Aspects of desertification in *Prosopis* dunelands of southern New Mexico, U.S.A. J. Arid Environ. 5:277-284.
- York, J.E., and W.A. Dick-Peddie. 1969. Vegetation changes in southern New Mexico during the past 100 years. In: W. G. McGinnies and B.J. Goldman (eds.), Arid lands in perspective. Univ. of Arizona Press, Tucson.

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Private forest landowner's perceptions of forest grazing in Washington state

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Abstract

Nonindustrial private forest landowners (NIPF) control 21.4% of Washington's commercial forestland, much of which produces forage. Resident NIPF owners in 3 regions in the state were surveyed to determine their perceptions of forest grazing. Thirty-nine percent of the respondents grazed livestock on forestland they leased or owned, and grazing was perceived by practitioners to contribute significantly to household income. Nonincome-related motivations for owning and managing land were also significant: passing land on to children, keeping it 'natural', conservation, aesthetics, and as a current or future homesite. In western Washington, some forest grazing occurred year round while in eastern Washington it was all seasonal. Cow/calf pairs were the most commonly grazed livestock. The median size forestland parcel owned by forest grazers was 47 ha versus 24 ha for nongrazers. Leasing additional land increased the likelihood of forest grazing. Significant opportunities exist to improve both the condition and productivity of forested ranges. Achieving this requires a clear understanding of landowner's objectives and beliefs. Data are needed to evaluate landowner's perceptions that forest grazing has both economic and amenity benefits.

Key Words: woodland grazing, silvopastoral, motivations, survey, NIPF

Washington has the lowest proportion of publicly owned land and the highest proportion of privately owned land of the 11 western states. One-third of Washington is rangeland, half of which is forested. Approximately 21% (1.5 millions ha) of the state's commercial forestland is owned by nonindustrial private forest landowners (NIPF) (Hanley and Baumgartner 1990). Non-industrial private forest landowners are those who do not own facilities to process the wood they produce. Recent studies (Wetton 1988, Blatner et al. 1991, Lawrence and Hardesty 1992, Lawrence et al. 1992) suggest that a number of Washington NIPF owners practice forest grazing, and that as much as 65% of this land is not in acceptable condition (Harris and Chaney 1984). Current lack of knowledge about NIPF's forest grazing practices and motives constrain efforts to assist them in utilizing their resources more effectively.

Grazing in the Pacific Northwest has been documented since the 1840s (Coville 1898, Jardine and Anderson 1919, Pearson 1923, Ingram 1931, Reid 1947, Colwell 1954, Sharrow and Leininger 1983, Harris and Chaney 1984). Early forestland grazing in the Pacific Northwest region followed timber harvest to reduce

unwanted vegetation and the fire hazard this vegetation posed to forest regrowth (Judd 1911, Reid et al. 1938, Byington 1990). As a result, much of the forest land suffered from overgrazing (Hess 1966, Harris and Chaney 1984).

The degradation testifying to this history may have fostered the assumption that livestock grazing and forestry are incompatible (Daniel and Ensminger 1945), a perception that lingers unexamined in many quarters today. If, as some anticipate, public and government concern about fish and wildlife habitat, threatened and endangered species, and water quality results in reduction or elimination of timber harvest and livestock grazing on some public lands in this region, private forest lands may be subjected to increasingly intensive use. There is an urgent need for information on which to base technical recommendations for sound management of these lands in the future.

NIPF owners control over 58% of all commercial forestland in the US, and consequently, have been studied extensively. Activities such as livestock grazing have been addressed only indirectly, however, and generally from a timber management perspective (Weatherland et al. 1982, Bliss and Martin 1989). Our study examined how Washington NIPF owners perceive forest grazing as a means of meeting their land management objectives.

Objectives of this study were to:

- 1) determine the extent that forest grazing is used by Washington's NIPF owners,
- 2) develop insight into NIPF owner's motivations for forest grazing,
- 3) describe forest grazing practices of NIPF owners in Washington, and
- 4) determine the information needs of NIPF owners who practice forest grazing.

Methodology

Sample Frame

Using the sample frame of a concurrent study of NIPF agroforestry practices (Lawrence and Hardesty 1992, Lawrence et al. 1992), our survey was confined to 3 areas, each consisting of 4 counties where forestry and agriculture were primary land uses (Fig. 1). Geographic location, socioeconomic conditions, and agricultural and horticultural crops produced were considered to insure maximum diversity.

Area 1, in northcentral and northeastern Washington, is predominantly rural. This area is characterized by 400 to 650 mm annual precipitation, hot (16–24° C) dry summers and cold (–6–0° C) winters. There are 100 frost-free days in the mountains and 140 to 200 frost-free days in the valleys (Franklin and Dyrness 1973, Harris and Chaney 1984). Ranching, forestry, and agriculture (tree fruits, grain, and hay) are the dominant land uses. Forests are dominated by ponderosa pine (*Pinus ponderosa* Dougl.), Douglas

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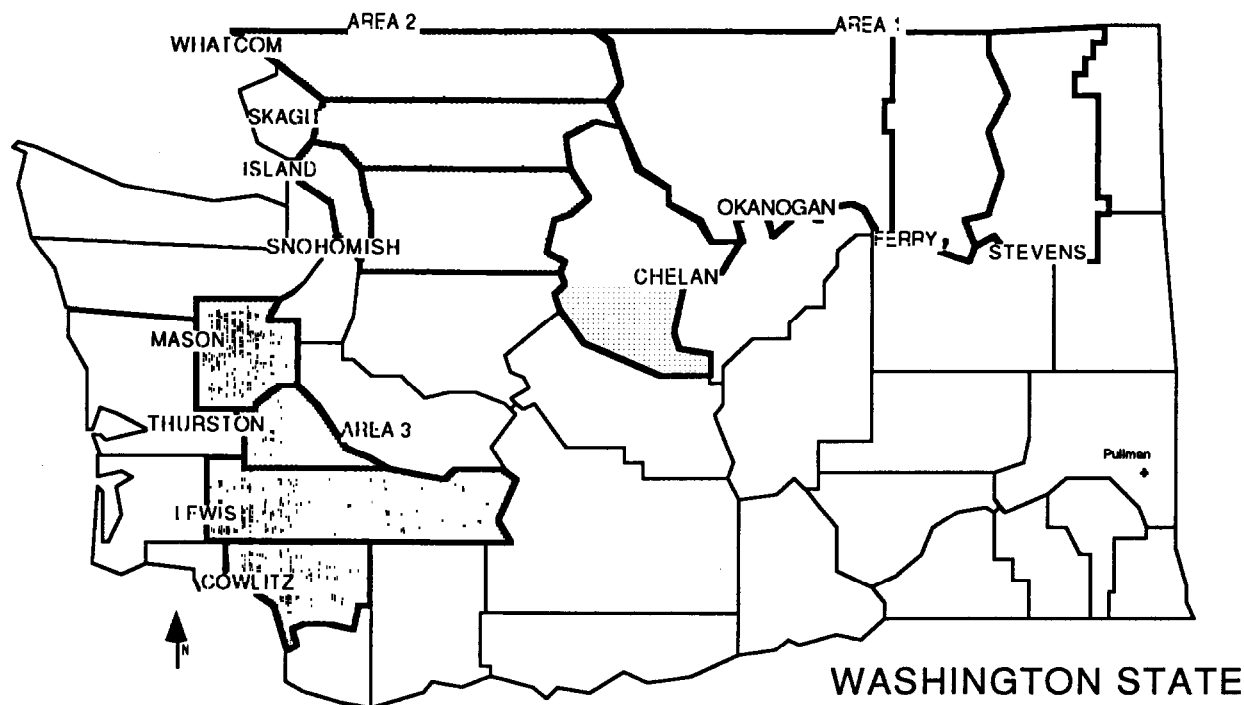


Fig. 1. Areas sampled for Washington State nonindustrial private forest landowner forest grazing practices survey.

fir (*Pseudotsuga menziesii* (Mirbel) Franco.), lodgepole pine (*Pinus contorta* Dougl.), western larch (*Larix occidentalis* Nutt.), and Engelmann spruce (*Picea engelmannii* Parry) mixed with true firs (*Abies* spp. Mill) (Franklin and Dyrness 1973).

Areas 2 and 3 are located in northwest and southwest Washington, respectively. Both are a mix of highly urbanized and rural environments. Important land uses in area 2 include forestry, growing vegetable seeds, potatoes, and bulbs. In area 3, forestry, mixed farming, and dairy farming are important. The climate of western Washington is relatively mild without the extremes of area 1. Temperatures in the coastal lowlands average from 7 ° C in January to 17° C in July. The frost-free season ranges from 200 to 240 days. Average rainfall in areas 2 and 3 ranges from 1,000 to 2,000 mm annually (Harris and Chaney 1984).

The forests of western Washington are some of the world's most productive. Douglas fir forests are the most extensive. Other common forest types include western hemlock (*Tsuga heterophylla* (Raf.) Sarg.), true firs, and mountain hemlock (*Tsuga mertensiana* (Bong.) Carr.) intermixed with western red cedar (*Thuja plicata* Donn.), red alder (*Alnus rubra* Bong.), and bigleaf maple (*Acer macrophyllum* Pursh) (Franklin and Dyrness 1973). Generally, the dense, humid westside forests are not suited for grazing except as transitory range.

Survey Tool

Data were collected using a mail survey based on Dillman's Total Design Methodology (Dillman 1978). A previously defined population of NIPF landowners randomly selected from county tax records (Blatner et al. 1991) was used to generate mailing lists. Only those forest landowners residing in a county where they owned forest land were surveyed, and we requested that the person most directly responsible for management of the land complete the questionnaire. The survey was pretested and 3 mailings and 3 follow-ups were sent to encourage response. The mailings included a 10-page questionnaire of 45 questions, cover letter, and pre-paid return envelope. The questionnaire was initially mailed to 680 NIPF owners in August, 1990, and responses were accepted through November, 1990. No survey of nonrespondents was

undertaken.

The questionnaire was developed after an earlier, open-ended questionnaire had defined the issues addressed herein (Lawrence and Hardesty 1992). Forest grazing was one of several agroforestry practices included in the study and although it is the only practice reported here, some of the questions addressed a broader range of agroforestry practices. The questionnaire was divided into 5 sections:

1. land: hectares owned and leased from others, location, and tenure;
2. forest grazing practices;
3. reasons for practicing agroforestry, and the potential advantages or disadvantages of agroforestry;
4. reasons for owning land, and the contribution of 12 different products or activities to household income; and
5. socioeconomic characteristics and agroforestry information needs.

Table 1. Percentage distribution of current occupations of forest grazers (FG) and non-forest grazers (NG) within geographic regions.

Occupation	N=	Overall		FG		NG	
		FG 113	NG 138	East 65	West 48	East 56	West 82
		----- % -----					
Rancher/farmer		34.8	11.8	46.8	19.7	17.4	7.6
Retired		26.1	34.2	20.8	32.8	37.0	
Self-employed other		15.9	15.5	15.6	16.4	15.9	15.2
Full-time employee		11.6	22.4	7.8	16.4	23.2	21.7
Part-time employee		5.8	4.3	3.9	8.2	2.9	5.4
Professional		2.9	6.8	1.3	4.9	4.3	8.7
Full-time home-maker		2.2	3.1	2.6	1.6	2.9	3.3
Unemployed		0.7	0.6	1.3	0.0	1.4	0.0
Full-time student		0.0	1.2	0.0	1.4	1.1	
		100	100	100	100	100	100

Table 2. Total area owned and leased and median parcel sizes controlled by forest grazers (FG) and non-forest grazers (NG), by land type and geographic region.

Ownership	Total size				Median parcel size					
	FG		NG		Overall		FG		NG	
	(ha)	(n)	(ha)	(n)	FG	NG	East	West	East	West
Land owned	35,896	(114)	9,659	(143)	100	32	223	64	49	19
Land leased	39,374	(36)	468	(11)	166	40	239	36	65	40
Forestland owned	15,138		7,184		47	24	121	41	32	16
Non-forested owned ¹	20,758		2,475		—	—	—	—	—	—
Forestland leased	12,626	(25)	373	(6)	162	49	202	31	81	26
Non-forested leased ¹	26,748		95		—	—	—	—	—	—
All land (leased & owned)	75,270		10,127							

¹hectares for non-forestland leased and owned were derived by subtraction.

Statistical Analysis

Frequency data were obtained for each response. The Chi square equality of means test at $\alpha = 0.05$ was used to detect statistically significant differences. When necessary, responses from areas 2 and 3 were combined to meet minimum cell requirements, and referred to collectively as western Washington. Median values were reported for livestock and land ownership data to circumvent the influence of extreme values. Respondents were classified as forest grazers or nonforest grazers. Nonforest grazers may or may not have livestock, but do not use forested lands for grazing. Regional differences (east versus west) in responses were also examined.

Results and Discussion

Sixty-four percent of the sample population returned useable questionnaires and 39% of these grazed livestock they owned on forestland they leased or owned. An additional 13% had practiced forest grazing, but no longer do so due to advancing age or other personal factors. Our results should be qualified with a reminder that only resident NIPF owners were surveyed, hence these results may not apply equally to NIPF owners living more distant from the land they own, and as a result perhaps less able to manage their land.

Demographic Characteristics

Our respondents differed little from respondents in other state or

Table 3. Percentage distribution of forest grazers (FG) and non-forest grazers (NG) selecting a particular reason for owning land, and selecting particular contributors to total household income, by geographic region.

Reasons	Overall			FG			NG	
	FG	NG	P ¹	East	West	P ¹	East	West
	----- % -----	----- % -----		----- % -----	----- % -----		----- % -----	----- % -----
To pass on land to children	82.4	77.8	0.408	81.3	83.7	0.757	77.1	78.3
Income from timber	79.3	66.7	0.042	88.2	68.3	0.019	63.0	69.0
Income from livestock	78.7	22.8	0.001	88.7	65.9	0.007	26.2	20.3
For conservation	77.8	67.0	0.107	82.2	72.2	0.282	61.9	70.5
To keep it natural	75.3	75.5	0.975	80.0	69.2	0.243	85.1	68.3
A place to live/future building site	74.4	71.8	0.699	79.5	68.4	0.250	68.2	74.6
A place to retire	73.9	72.3	0.807	71.4	76.9	0.560	79.5	66.7
Income from farming	64.1	32.7	0.001	75.0	50.0	0.013	43.2	25.0
A place to fish or hunt	53.6	50.9	0.719	55.6	51.3	0.696	53.5	49.2
The land has always been in the family	45.2	51.8	0.396	51.2	39.5	0.282	86.7	44.1
Investment or tax shelter	41.7	49.0	0.313	34.1	50.0	0.140	41.9	54.1
Income from leasing out grazing rights	21.0	9.1	0.024	33.3	5.6	0.002	12.2	6.9
Eventual commercial development	18.9	27.1	0.214	12.5	26.5	0.126	22.0	30.9
Income from recreation	8.8	19.0	0.052	16.7	0.0	0.008	9.8	25.4
Income from Christmas trees	7.7	18.0	0.046	4.9	10.8	0.326	4.9	27.1
Income Sources								
Sale of livestock	78.4	19.4	0.001	83.3	71.4	0.150	22.7	16.9
Timber sales	73.0	65.9	0.251	83.1	58.5	0.007	61.7	68.4
Off-farm work	48.2	26.5	0.003	54.3	40.5	0.211	24.4	28.1
Income from grain farming	29.5	5.9	0.001	46.9	7.7	0.001	11.6	1.7
Leasing out grazing rights	20.2	5.9	0.003	27.1	12.2	0.081	7.1	5.1
Leasing land other than grazing or CRP land	17.2	7.9	0.052	25.0	7.7	0.034	7.1	8.5
CRP land payment	11.3	3.1	0.033	20.9	0.0	0.003	2.5	3.6
Selling fruit	10.6	12.0	0.763	14.9	5.3	0.151	21.4	5.2
Income from special forest products	9.6	15.7	0.224	10.9	8.1	0.671	7.3	21.3
Income from farming vegetables	8.4	9.9	0.733	10.9	5.4	0.373	14.3	6.8
Christmas trees	8.3	16.2	0.112	8.5	8.1	0.950	4.9	24.1
Selling nuts	6.1	4.0	0.516	8.9	2.7	0.244	2.4	5.2

¹Response of forest grazers and non-grazers overall and response of forest grazers by region. Chi square test for equality of means ($\alpha=0.05$).

regional NIPF studies (Baumgartner 1980, Weatherhead et al. 1982, Wetton 1988, Blatner et al. 1991, Force and Lee 1991), permitting us to omit a survey of nonrespondents. Respondents who grazed livestock on forestland were predominantly white (95.0%) and male (89.5%) with an average age of 58 years. The only notable difference between forest grazers and nonforest grazers was that over twice as large a percentage of nonforest grazers had done graduate study. Average gross annual household income for forest grazers and nonforest grazers were \$53,372 and \$42,615, respectively.

Forest grazers most often identified themselves occupationally as a rancher/farmer or retired. Nonforest grazers were most often full time employees or retired. Eastern Washington had a higher proportion of farmer/ranchers and fewer retirees among those who practice forest grazing (Table 1).

Forest grazers owned and leased more land and larger parcels than nonforest grazers (Table 2). Deeded and leased lands made up 53 and 47%, respectively, of all land managed by respondents. Forest grazers control 78.8% of all deeded land and 98.8% of the leased acreage managed by survey respondents. Forty-nine percent of the deeded land was forested as was 33% of the leased land. Dependence on leased land was most evident among eastern Washington forest grazers.

Motivations for Owning Land

A landowner's reasons for owning land can be expected to

influence land management strategies. Our respondents were similar to those in other NIPF studies regarding reasons for owning land (Baumgartner 1980, Weatherhead et al. 1982, Wetton 1988, Blatner et al. 1991, Force and Lee 1991). 'To pass on land to children' was the forest grazer's most frequently stated reason followed by 'income from timber' and 'income from livestock', 'for conservation', and 'to keep it natural' (Table 3). For non-forest grazers, the most frequently cited reasons for owning land were 'to pass on land to children', 'to keep it natural', 'a place to retire', 'a place to live/future building site' and 'for conservation'. Some reasons were cited significantly more often by forest grazers: 'income from timber' ($p \leq 0.042$), livestock ($p \leq 0.001$), farming ($p \leq 0.001$), and leasing out grazing rights ($p \leq 0.024$). These data suggest that forest grazers' land use is more financially motivated than nonforested grazers', an idea supported by employment and income data. Only 'income from Christmas trees' ($p \leq 0.046$) was more important to nonforest grazers than to forest grazers. The most frequent reason eastside nongrazers owned land was the 'land has always been in the family'. The importance of nonincome generating motives for owning land should not be underestimated.

Eastside forest grazers cited income from timber ($p \leq 0.019$), livestock ($p \leq 0.007$), farming ($p \leq 0.013$), leasing out grazing rights ($p \leq 0.002$) and income from recreation ($p \leq 0.008$) more often than their westside counterparts, perhaps because they control more land and are more likely to be supported by it. Further, farming, livestock, and forestry, as well as off-farm work may all be needed

Table 4. Percentage distribution of forest grazers (FG) and non-forest grazers (NG) indicating choices of potential advantages and disadvantages of agroforestry, as well as reasons others might not consider practicing agroforestry, by geographic region.

	Overall			FG			NG	
	FG	NG	P ¹	East	West	P ¹	East	West
Advantages	----- % -----			----- % -----			----- % -----	
Integration of conservation and production goals	78.6	65.9	0.080	73.7	84.4	0.278	67.6	64.7
Aesthetically pleasing	77.3	76.1	0.850	66.7	88.9	0.022	84.2	70.4
Increases land unit income	76.7	69.9	0.292	75.0	78.9	0.667	73.2	67.7
Diversifies productive and economic base	75.7	59.6	0.028	70.0	82.4	0.217	59.5	59.6
Increases biological diversity	73.5	58.8	0.058	65.8	83.3	0.103	70.3	50.0
Improves soil fertility and productivity	72.0	77.4	0.420	69.0	75.8	0.520	86.8	70.9
Water conservation	71.6	69.1	0.723	75.0	66.7	0.414	84.6	58.2
Soil erosion control	69.5	62.9	0.352	76.1	61.1	0.144	68.3	58.9
Reduced need for pesticides/-fertilizer/fuel	61.4	53.9	0.343	56.8	66.7	0.395	60.0	50.0
Microclimate improvement	53.7	52.8	0.909	51.4	56.3	0.693	75.0	37.7
Disadvantages								
Financial assistance lacking for unknown practice	61.8	60.7	0.891	63.9	59.4	0.702	58.3	62.3
Livestock damage trees and crops	58.0	69.9	0.103	446.5	71.1	0.026	64.9	73.2
Establishment cost high	56.2	72.0	0.033	64.1	47.1	0.143	68.4	74.5
Trees get in the way of farm equipment	42.3	40.7	0.844	40.5	0.760	41.2	40.4	
Increase in rodent/insect/weed pests	39.4	47.0	0.353	45.7	32.3	0.264	57.6	40.0
Complicated/difficult to manage	35.9	53.5	0.033	37.1	34.5	0.825	55.9	51.9
Livestock compact soil	31.9	39.6	0.306	11.1	52.8	0.001	40.0	39.6
Reasons								
Lack of technical assistance/education support	62.7	63.4	0.917	63.4	61.8	0.882	73.7	56.4
Not practical in commercial situations	53.4	57.1	0.634	44.7	62.9	0.121	63.9	52.7
No access to livestock or livestock facilities	49.3	59.8	0.188	50.0	4.4	0.893	57.6	61.1
Lack of scientific research	48.0	53.2	0.503	41.0	55.6	0.208	56.8	50.9
Bias against combining grazing and forestry	38.7	33.3	0.484	27.5	51.4	0.034	33.3	33.3
Bias of peers against unfamiliar practice	28.6	37.1	0.259	28.9	28.1	0.938	42.4	33.9

¹Response of forest grazers and non-forest grazers overall and response of forest grazers by region. Chi square test for equality of means ($\alpha=0.05$).

Table 5. Comparison of median and total land area and herd size characteristics of forest grazers who do and do not lease land.

Leased land	All land owned	All land leased ¹	Forest land owned	Forestland leased ²	Forestland grazed	Head livestock ³	Years practiced
Yes (n=37)	----- (ha) -----						
Total	16,318	39,374	5,119	12,626	18,520	7,088	1,206
Median	219	166	51	146	138	65	26
No (n=77)							
Total	19,574	na	10,000	na	9,194	3,267	1,726
Median	87	na	32	na	26	20	20

¹Includes all land leased: forestland, pasture, agricultural land, etc.

²Includes land leased by one respondent for harvest of special forest products.

³Includes all livestock.

to reduce the risk of land based enterprises in the more extreme and variable climate of eastern Washington.

Respondents were asked to indicate important sources of household income (Table 3). Forest grazers most often chose 'sale of livestock', 'timber sales', 'off-farm work', 'income from grain farming', and 'leasing out grazing rights', while nonforest grazers chose 'timber sales', 'off-farm work', 'sale of livestock', 'Christmas trees', and 'income from special forest products' (evergreen boughs and understory plants with culinary or ornamental value). Significantly more forest grazers chose 'sale of livestock' ($p \leq 0.001$), 'off-farm work', ($p \leq 0.003$) 'income from grain farming' ($p \leq 0.001$), 'leasing out grazing rights' ($p \leq 0.003$), and 'CRP (Conservation Reserve Program) payments' ($p \leq 0.033$). These differences are consistent with the reasons forest grazers and nonforest grazers most often cited for owning land, suggesting that forest grazers place greater emphasis on nontimber resources.

Eastern and western Washington forest grazers did not differ in their 3 most frequent income sources: 'sale of livestock', 'timber sales', 'off-farm work'. However, on the eastside 'income from grain farming' replaced 'leasing out grazing rights' as the fourth most frequent choice. For westside forest grazers, 'Christmas trees' and 'income from special forest products' were the fifth most often chosen income sources, compared to 'leasing out grazing rights' in the east. Significantly more eastern forest grazers chose 'timber sales' ($p \leq 0.007$), 'income from grain farming' ($p \leq 0.001$), 'leasing out grazing rights other than CRP land' ($p \leq 0.034$) and 'CRP payments' ($p \leq 0.003$).

Eastern Washington forest grazer's greater reliance on 'income from timber sales' ($p \leq 0.007$) may relate to their larger holdings (Table 2). NIPF owners with more than 65 ha of forestland are more likely to actively manage their timber than those with less land (Weatherhead et al. 1982). Timber harvest practices might also be a factor. Westside forests are typically clearcut, an event occurring only rarely during a lifetime, depending on the acreage held. On the eastside, selection and partial cutting (Weatherland et al. 1982) allow landowners to harvest trees more often. Other regional differences; 'income from grain farming', 'leasing out grazing rights other than CRP land' and 'CRP payments', are consistent with region differences in climate and land use.

Motivations for Practicing Agroforestry, Including Forest Grazing

We asked respondents why other landowners such as themselves might not practice agroforestry. Both the forest grazers' and nonforest grazers' most frequent responses were 'lack of technical assistance/educational support', 'not practical in commercial situations', 'no access to livestock or livestock facilities' and 'lack of scientific research' (Table 4). More western Washington forest grazers indicated 'general bias against combining grazing and forestry' ($p \leq 0.034$) as a potential obstacle. This is likely due to differing land use histories and forest types.

Based upon their own experiences, forest grazers most often chose as potential advantages of agroforestry: 'integration of con-

servation and production goals', 'aesthetically pleasing', 'increases land unit income' and 'diversifies productive and economic base' (Table 4). Differences between forest grazers and nonforest grazers were not significant except that more forest grazers chose 'diversifies productive and economic base' ($p \leq 0.028$), suggesting a greater concern for economic benefits. The importance of aesthetics to both groups emphasizes that we need to learn what is considered aesthetically pleasing and how these criteria affect, and are affected by, land management practices. Eastern and western Washington forest grazers perceived similar potential advantages of agroforestry, but 'aesthetically pleasing' ($p \leq 0.022$) was significantly more important to westside forest grazers.

Economic factors were highlighted in the potential disadvantages of practicing agroforestry perceived by forest grazers: 'financial assistance lacking for unknown practice', 'livestock damage trees or crops', and 'establishment costs high'. Nonforest grazers had similar views, although fewer forest grazers saw 'establishment costs high' as a potential obstacle ($p \leq 0.033$) (Table 4). Establishment costs may be overestimated by nonforest grazers relative to actual costs incurred by forest grazers. 'Complicated or difficult to manage' ($p \leq 0.033$) was cited twice as often by nonforest grazers. Again, the requirements may be overestimated by the inexperienced. No actual data concerning establishment costs and management inputs are available to clarify these differing perceptions.

Westside forest grazers were more concerned that "livestock damage trees and crops" ($p \leq 0.026$) and compact soil ($p \leq 0.001$). The former perception may be a result of regional differences in tree and crop species as well as silvicultural and agricultural practices. Higher precipitation probably explains the concern about soil compaction since eastside soils are often wet for extended periods. Both forest grazers and nonforested grazers recognized that forest grazing may be hazardous to other resources, although there are few data quantifying potential damages. Distinctions between proper and improper grazing were not drawn, but should be included in field studies. Because behavior is influenced by beliefs, it is important that the actual hazards and external costs of forest grazing be clarified and shared with land owners.

Forest Grazing Characteristics

Forest grazing was the most common agroforestry practice used by survey respondents, and the only agroforestry practice employed by 71.9% of the agroforesters (Lawrence et al. 1992). In eastern Washington, 46.0% of all respondents grazed livestock on forestland. Median herd size (cattle only) was 67. In western Washington, 31.0% of the respondents grazed livestock on forestland, but the median herd size was only 20 head. Cow/calf pairs constituted 76.0% of all livestock on forested ranges. One quarter of the forest grazing occurred year round in the counties west of the Cascade mountains. All other forest grazing occurred between early April and early November. Most of Washington's forest ranges are inaccessible for 4 to 6 months annually due to snow or mud.

The most common forest types grazed in eastern Washington

were Douglas fir, ponderosa pine or Douglas fir mixed with ponderosa pine and western larch. Understory vegetation associated with these forest types includes snowberry (*Symphoricarpos albus* Gray), oceanspray (*Holodiscus discolor* (Pursh) Maxim.), bitterbrush (*Purshia tridentata* (Pursh)), bluebunch wheatgrass (*Agropyron spicatum* (Pursh) Scribn. & Smith), Idaho fescue (*Festuca idahoensis* Elmer), needle-and-thread (*Stipa comata* Trin. & Rupr.) and Sandberg bluegrass (*Poa sandbergii* Vasey) (Reid 1964, Harris and Chaney 1984). Upland forests are important in the summer and autumn when grasses at lower elevations are dry and have little nutritive value (Reid 1947, Reid 1964). Understory forage production varies from 112–4,491 kg/ha and is closely associated with the density of the forest overstory, precipitation, soils, and topography (Harris and Chaney 1984).

The most common forest types grazed in western Washington were Douglas fir alone or in association with western red cedar and red alder. The understory offers few preferred forage species except on recent clearcuts: velvet grass (*Holcus lanatus* L.), trailing blackberry (*Rubus ursinus* Cham. & Schlecht.), hairy cat's ear (*Hypochaeris radicata* L.), pearly everlasting (*Anaphalis margaritacea* (L.) B. & H.), and coltsfoot (*Petasites frigidus* (L.) Fries) provide a significant amount of palatable and nutritious forage (A. Dobkowski personal communication). Forage production on these forest sites ranges from 0–5,614 kg/ha (Harris and Chaney 1984), again depending on both environmental and management influences.

Leased lands play an important role in the forest grazing practices of Washington's NIPF owners. Median herd sizes were larger for forest grazers who lease additional land and the median parcel size of deeded forestland was double that of forest grazers who did not lease land (Table 5). Most leased lands are private rather than publicly owned. No doubt there is a minimum acreage (deeded and/or leased) below which forest grazing is not feasible. Grazing permits may contribute to this threshold: 11% of all respondents held public land grazing permits and 86% of the permit holders were eastside agroforesters. Having established the importance of grazing on NIPF lands, we should now learn more details of actual forest grazing practices, efficiencies, and problems.

Agroforestry Information Needs

Both landowners and technical advisors (Lawrence and Hardesty 1992) cited lack of technical assistance, educational support and scientific research as obstacles to implementing agroforestry practices. The type of information respondents would request is another indicator of their concerns and priorities. When asked what, if any, types of agroforestry training or extension information would be useful to them, 46% of all forest grazers requested information, primarily concerning 'livestock management in agroforestry systems' (100% of forest grazers requesting information), 'management of trees in agroforestry systems' (83%), 'using trees and shrubs as shelter/forage for livestock' (65%) and 'managing forestland for special forest products' (63%). Nonforest grazers were most interested in managing for special forest products (100% of those requesting information) and managing trees in agroforestry systems (91%). Interestingly, 43% of the nonforest grazers were also interested in managing livestock in agroforestry systems, suggesting that some might consider forest grazing if they knew more about it.

Information should be targeted to specific groups of NIPF owners. For example, both forest grazers and nonforest grazers were interested in information about 'management of trees in agroforestry systems', but the 2 groups appear to have different views of the role of trees in agroforestry systems. While differences exist in their perceptions of agroforestry, NIPF owners are sufficiently homogeneous in regards to personal characteristics and

motivations to be well served by properly focused educational programs.

Implications and Recommendations

Forest grazing was practiced by 39% of the resident NIPF owners responding to our survey. Parcel size was related to the likelihood of forest grazing although it is unclear if more land was obtained in order to produce livestock, or if livestock are used to optimize management of lands already controlled. A larger land area may better support forest grazing by contributing more diverse forage resources and greater opportunities for proper livestock and range resource management. The role of leased lands in forest grazing operations should be clarified, as well as the minimum acreage requirements for successful forest grazing. Cooperative programs or grazing associations may be useful both in accessing sufficient acreage and encouraging proper management.

Assessing range condition on forest lands is complicated by the fact that succession ultimately precludes much forage production, and that the effects of timber management practices and grazing practices are confounded. Harris and Chaney (1984) cite 1981 data from the Soil Conservation Service indicating that 65% of Washington's State and privately owned grazeable woodlands are producing "less than acceptable" amounts of forage relative to their potential in a given successional state. The state's grazeable woodlands are estimated to be producing only 31% of the forage they could be producing if all were in satisfactory condition, hence significant opportunities exist to improve both condition and forage production on privately owned forest lands. They recommend thinning and pruning trees on 56% of the state's grazeable woodlands, and note that few additional practices, other than proper grazing management, and site stabilization following timber harvest, are needed to improve the condition and productivity of these lands. The landowners' willingness to do these improvements is strongly dependent upon their reasons for owning land and their perceptions of various land use practices.

Although forest grazing is perceived by practitioners to contribute significantly to household income, the importance of non-income related motives for owning and managing land cannot be overemphasized. Focusing exclusively on owners' economic or livestock production objectives could be a serious error given the importance of passing land on to children, keeping it 'natural', conservation, aesthetics, and a current or future homesite. The number of forest grazers reporting these objectives suggests that they are not seen as incompatible with forest grazing. Targeting these objectives should increase the success of programs to improve conditions on privately owned forested ranges.

As in any opinion survey, our results must be qualified: actual behavior is not always consistent with respondent's statements. However, behavior is influenced by beliefs, regardless of the factual basis of those beliefs. We need to verify the accuracy of respondents' perceptions. In some cases, information already exists that may counter some of these perceptions. For example, Halloin (1991) recently reviewed the literature on livestock damage to trees, concluding that little hazard exists with proper management. Education might reassure NIPF owners in this regard and promote such proper management. In other cases, such as soil compaction, data are not available to determine the potential for forest soil compaction due to grazing. Economic analyses of forest grazing are urgently needed given the prevalence of economic motives, and the fact that economic justification is needed to access funds for improvement of these ranges. The significance of regional differences, both environmental and socio-economic, should be explored in recognition to the range of conditions under which forest grazing is practiced.

This work emphasizes the importance of understanding land

owner's actual beliefs and practices. For example, an earlier study of land management advisors (Lawrence and Hardesty 1992) did not identify soil compaction as a potential hazard of agroforestry, yet the landowners surveyed frequently expressed this concern. Understanding landowner's values, perceptions, and practices should result in more effective research and extension programs, and ultimately, in better land management.

Literature Cited

- Baumgartner, D.M. 1980.** Nonindustrial private forestland owners: Stevens County, Washington. p. 266-268. *In: Proc. 1979 Convention Soc. Amer. Forest., Soc. Amer. Forest., Bethesda, Md.*
- Blatner, K.A., D.M. Baumgartner, and L.R. Quackenbush. 1991.** NIPF use of landowner assistance and education programs in Washington State. *Western J. App. Forest.* 6:90-94.
- Bliss, J.C., and A.J. Martin. 1989.** Identifying NIPF management motivations with qualitative methods. *Forest. Sci.* 35:601-622.
- Byington, E.K. 1990.** Agroforestry in the temperate zone. Chapter 7. *In: K.G. MacDicken and N.T. Vergara (eds.). Agroforestry: Classification and Management.* John Wiley & Sons, N.Y.
- Colwell, D.E. 1954.** Harmonizing forest and range management in the ponderosa pine forests of central Washington. p. 208-210. *In: Proc. : Soc. Amer. Forest. Soc. Amer. Forest., Bethesda, Md.*
- Coville, F.V. 1898.** Forest growth and sheep grazing in the Cascade mountains of Oregon. *U.S. Div. Forest., Bull.* 15.
- Daniel, T.W., and M.E. Enslinger. 1945.** Grazing on the cutover land of western Washington. *State College of Washington. Agr. Exp. Sta. Population Bull.* 179.
- Dillman, D.A. 1978.** Mail and telephone surveys: The total design method. John Wiley & Sons, N.Y.
- Force, J.E., and H.W. Lee. 1991.** Nonindustrial private forest owners in Idaho. *Western J. App. Forest.* 6:32-36.
- Franklin, J.F., and C.T. Dyrness. 1973.** Natural vegetation of Oregon and Washington. *USDA Forest Serv. Gen. Tech. Rep. PNW-8.*
- Halluin, L. 1991.** Plantation grazing: A feasibility review. *Dept. of Natur. Resourc., Forest Land Manage. Div. Olympia, Wash.*
- Hanley, D.P., and D.M. Baumgartner. 1990.** Forest and forest products data: Washington State. *Washington State Coop. Ext., Pullman, Wash.* EB 1557.
- Harris, G.A., and M. Chaney. 1984.** Washington State grazing land assessment. *Washington State Univ. Coop. Ext. Pullman, Wash.*
- Hess, J.P. 1966.** Management of livestock grazing on private timberland. p. 231-232. *In: Proc. Soc. Amer. Forest. Soc. Amer. Forest., Bethesda, Md.*
- Ingram, D.C. 1931.** Vegetative changes and grazing use on Douglas fir cut-over land. *J. Agr. Res.* 43:387-417.
- Jardine, J.T., and M. Anderson. 1919.** Range management on the national forests. *USDA: Professional Paper. Bull.* 790.
- Judd, C.S. 1911.** Efficacy of goats in clearing brush land in the Northwest. p. 108-112. *In: Proc. Soc. Amer. Forest. Soc. Amer. Forest., Bethesda, Md.*
- Lawrence, J.H., and L.H. Hardesty. 1992.** Mapping the territory: Agroforestry awareness among Washington State land managers. *Agroforest. Syst.* 19:27-36.
- Lawrence, J.H., L.H. Hardesty, R.C. Chapman, and S.J. Gill. 1992.** Agroforestry practices of Washington State non-industrial private forest land owners. *Agroforest. Syst.* 19:37-55.
- Pearson, G.A. 1923.** Natural reproduction of western yellow pine in the Southwest. *USDA: Forest Serv. Bull.* 1105.
- Reid, E.H. 1947.** Forest grazing in the Pacific Northwest. p. 296-302. *In: Proc. Soc. Amer. Forest. Soc. Amer. Forest., Bethesda, Md.*
- Reid, E.H. 1964.** Forage production in ponderosa pine forests. p. 61-64 *In: Proc. Soc. Amer. Forest. Soc. Amer. Forest., Bethesda, Md.*
- Reid, E.H., L.A. Issac, and G.D. Pickford. 1938.** Plant succession on a cutover, burned, and grazed Douglas fir area. *Pacific Northwest Forest and Range Exp. Sta. Forest Res. Notes* 26.
- Sharrow, S.H., and W.C. Leininger. 1983.** Sheep as a silvicultural tool in coastal Douglas-fir forests. p. 219-232. *In: D.D. Hannaway (ed.). Proceedings foothills for food and forests: Int. Hill Lands Symp. Timber Press, Beaverton, Ore.*
- Weatherhead, D.J., R.C. Chapman, W.H. Kelso. 1982.** Characteristics of NIPF landowners of Spokane County, Washington. *Res. Bull. XB0928, Agr. Res. Center, Washington State Univ., Pullman, Wash.*
- Wetton, C.E. 1988.** A survey of private forest land owners in British Columbia. *FRDA Rep. 044. Forest Res. Develop. Agreement, Victoria, B.C.*

Dynamics of vegetation along and adjacent to an ephemeral channel

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Abstract

Ephemeral channels may be greater contributors to nonpoint sediment loads than perennial channels because of their abundance and lower vegetative cover. This study examines above- and belowground standing crop responses of selected vegetation classes and density of shrubs to grazing use and yearly weather variation along an ephemeral stream in northcentral Wyoming.

Aboveground biomass standing crop was determined yearly in channel, floodplain, and upland habitats in ungrazed and grazed pastures during the 4-year study. Belowground biomass and shrub densities were determined yearly in the channel habitat only.

Perennial grass standing crop in channels did not respond to grazing but decreased up to 73% with decreases in frequency and amount of precipitation. In floodplains, perennial grasses were not responsive to grazing; annual grasses were twice as abundant in grazed pastures. Vegetation standing crop in uplands was not influenced by grazing. Over the study period in all pastures, standing crop of blue grama (*Bouteloua gracilis* (H.B.K.) Lag. ex Grif-fiths) declined 4 fold while cool-season grasses increased 5 fold. Shrub density did not increase as much in grazed as in ungrazed pastures. Root biomass of the channel decreased 23% in years with less precipitation but was greater by 24% on concave than convex bank types. Location on channels influenced root biomass but grazing did not. Lack of general negative grazing influences on vegetation suggest short periods (10 days) of grazing as used in this study represent a sustainable management alternative for grazing in the cold desert.

Key Words: grazing, weather, aboveground biomass, shrub density, belowground biomass, channel morphology

Understanding impacts of large herbivore grazing on vegetation is critical in developing rangeland management plans. This is especially true for the plant communities associated with streams traversing arid and semiarid rangelands. Reviews by Platts (1982a, 1982b), Skovlin (1984), and Kauffman and Krueger (1984) and reports like U.S. GAO (1988) and Chaney et al. (1990) identify the importance of riparian zones for multiple uses. They focus on livestock grazing effects on perennial stream systems. Several investigators have reported on impacts of various livestock grazing management strategies on vegetation, channel integrity, recreation and fishery/wildlife values, and nonpoint pollution on riparian zones of perennial streams (Elmore and Beschta 1987, Platts 1989, Myers 1989, and Clary and Webster 1989). Less is known about ecology and multiple use of streamside zones along ephemeral streams.

Swanson et al.'s (1988) classification of riparian areas notes that the U.S. Bureau of Land Management's (1987) riparian zone policy excludes ephemeral streams or washes that do not exhibit the presence of vegetation dependent upon free water in the soil. Ephemeral streams, arroyos, washes, and gullies within semiarid lands can be sources of excessive sediment in perennial stream flow (Graf 1985). Cooke and Reeves (1976) attribute formation of arroyos in the U.S. Southwest and accelerated erosion to climatic effects triggered by road and trail development, and livestock grazing.

Although ephemeral streams carry only periodic flow, riparian plant species often occupy floodplain space due to hydraulic disturbance (Pickett 1980, Szaro 1990) and a subsurface water table. Riparian plants contribute to structural diversity of habitats and can be used to indicate where watershed improvement practices can supplement baseflow from streambanks and alluvium within ephemeral and intermittent channels. Baseflow augmentation is reviewed by Ponce and Lindquist (1990). Examples of changing streamflow from ephemeral to perennial are described by Heede (1977), Hooper et al. (1987), and Elmore and Beschta (1987). Watershed improvements to augment base flow are often associated with instream structures but structures are not always needed nor desirable.

Elmore and Beschta (1987) emphasize that instream improvements should not replace sound management of livestock grazing. Van Haveren and Jackson (1986) note that geomorphological processes will naturally heal riparian zones if watersheds are managed in a sound manner and time is not limiting. Van Haveren and Jackson (1986) pose 3 questions to be answered before using structures for modifying stream conditions: "i) will the structure permit the system to reach a condition of natural stability more rapidly than can be achieved passively? ii) are the benefits achieved by accelerated rehabilitation sufficient to justify the costs? and iii) will the achieved condition be self-sustaining, or will it be dependent upon the integrity of the structure? In most cases, the answer to all 3 questions should be yes".

The purpose of our overall study was to evaluate the potential of controlled livestock grazing to improve channel condition for baseflow augmentation on an ephemeral stream. We felt that controlled grazing should improve streambank vegetation cover, allowing more sediment to be deposited, thus raising the stream bottom at least as well or rapidly as no grazing. The ephemeral stream studied has isolated pockets of plains cottonwood (*Populus deltoides* Bartr. ex Marsh.) and coyote willow (*Salix exigua* Nutt.) along its floodplain and incised channel suggesting that the subsurface moisture already present could be enhanced by channel management. We evaluated cattle habitat selection and forage utilization rates (Smith et al. 1989, Smith et al. 1992), grazing and yearly weather effects on vegetation, sediment deposition rates, and channel dynamics.

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This paper reports on the part of our study concerning yearly dynamics of above- and belowground vegetation. Specific points addressed are yearly weather and grazing effects on: (1) Above-ground biomass of selected herbage classes in channel, flood plain and upland habitats, (2) Shrub density in channels on differing bank aspects (exposure to solar radiation), (3) Root biomass in channels on different reaches, reach types, and bank levels above channel floor.

Methods and Materials

Study Area and Treatments

The study area consisted of 5 adjacent pastures, each about 30 ha, located on Middle Fork just above its confluence with the South Fork of 15-Mile Creek. The site was located 15 km west of Worland, Wyo. The pasture sequence was established to maximize channel and riparian zone restoration through base flow augmentation (Heede 1977, 1982). The downstream pasture was ungrazed and reserved for future placement of structures, if needed, to raise the stream bed level. The pasture furthest upstream was ungrazed in an attempt to enhance growth of channel vegetation and thereby reduce stream flow velocity in the transition from a higher upstream to lower gradient in pastures downstream. The 3 pastures in between were grazed by cattle 1983–1986. Proceeding upstream, pastures were grazed in late May, mid July, and mid September. Thirty cow/calf pairs were placed in each pasture for 10 days to achieve a moderate stocking rate of .33 AUM/ha. Perennial forage species were utilized 25–50% (Smith et al. 1989 and 1992). Greasewood (*Sarcobatus vermiculatus* (Hook.) Torrey) was utilized 20–80% depending on season and plant part. Stockwater was provided in troughs in the flood plain of each pasture.

Precipitation amount and frequency were continuously recorded during the growing season at 3 stations located west to east along the stream course; 20 km upstream (Squaw Teats), 3 km upstream (Middle Fork), and at the enclosure. Stream flow gauging for this tributary of 15-Mile Creek were unavailable and personnel were not on site frequently enough to record flow frequency.

Each pasture contained about .7 km of channel and included the floodplain and upland. The stream bottom was from 1–2 m across with banks of about 3 m length, sloping at about a 44° angle from a nearly level floodplain. Bed materials averaged 58% sand, 25% silt, and 17% clay. We observed that fine sediments in low flows apparently sealed the channel and limited transmission of water into adjacent alluvium. The floodplain was about 100–150 m wide. The pastures were on a uniformly low gradient (0.03–0.05% slope) meandering area of the stream, located downstream from a higher gradient (0.06% slope), straighter section.

The area is in a 12.5 to 22.5 cm annual precipitation zone (SCS 1988). The vegetation is generally classified as cold desert shrub; however, the area differs from other cold desert areas of Wyoming and the intermountain west by having a higher frequency of summer precipitation (Fisser 1982), and consequently more warm-season grasses in the flora.

Three habitats were designated within the study area: incised channel, flood plain, and upland. Channel bank vegetation was largely herbaceous with Canada wildrye (*Elymus canadensis* L.) (all plant names used herein follow Dorn 1988); inland saltgrass (*Distichlis stricta* (Torrey) Rydb.); and slimflower scurfpea (*Psoralea tenuiflora* Pursh) as dominant perennials. Annuals, sixweek grass (*Festuca octoflora* Walt.) and cheatgrass brome (*Bromus tectorum* L.), were usually present. Big sagebrush (*Artemisia tridentata* Nutt. var. *tridentata*) and green and rubber rabbitbrushes (*Chrysothamnus vicidiflorus* (Hook.) Nutt., *C. nauseosus* (Pallas ex Pursh) Britt.) were common on channel banks.

The flood plain overstory was dominated by big sagebrush and greasewood. The understory vegetation was dominated by annu-

als, sixweeks fescue, cheatgrass brome, peppergrass (*Lepidium* spp.), and plantain (*Plantago* spp.). Perennials included rhizomatous wheatgrasses (*Elymus* spp.), Indian ricegrass (*Oryzopsis hymenoides* (R. & S.) Ricker ex Piper), and needleandthread (*Stipa comata* Trin. & Rupr.). Plains pricklypear (*Opuntia polyacantha* Haw.) was common. Plains cottonwood provided occasional relief to the otherwise monotonous flood plain landscape.

Uplands were dominated by blue grama (*Bouteloua gracilis* (H.B.K.) Lag. ex Griffiths) and plains pricklypear. Other species included needleandthread, Indian ricegrass, rhizomatous wheatgrass, sixweeks grass, and cheatgrass brome.

Sampling

Soil Water and Water Table Level

We measured soil water in each pasture using neutron attenuation throughout this study by placing access tubes (20 per pasture) near vegetation sampling sites at the edge and 1 and 2 m from channel edges with the bottom at 3 m depth near or below the level of the channel floor. Access tubes were also located at mid floodplain and near the outer edge. Observations were made monthly, May to September. We also placed 6 cm plastic observation wells up to 3 meters below the base level of the channel floor. Free water was not found and regular observations were not continued.

Aboveground Standing Crop

Current year's peak standing crop of herbaceous biomass was determined yearly in July in channel (1984–1986) and floodplain and upland (1983–1986) sites. This sampling date could have underestimated peak standing crop in the spring grazed pasture; however, noticeable regrowth occurred after grazing and before harvesting because of residual soil moisture. A double sampling procedure was used. Standing crop was estimated by species on permanently located quadrats along transects. Temporary quadrats located 1 m away from at least one-half of the permanent quadrats and with locations rotated yearly were estimated and then harvested to calculate a regression to adjust estimated standing crops.

Table 1. Total herbaceous vegetation and perennial grass standing crop (g/m²) of channels in pastures on Middle Fork, 15-Mile Creek.

	Years			Mean
	1984	1985	1986	
	(g/m ²)			
Total herbaceous pasture				
Control 1	42	36	36	38 ± 7.2
Spring 2	31	24	20	25 ± 5.9
Summer 3	52	36	35	41 ± 10.9
Fall 4	73	24	52	47 ± 15.7
Control 5	35	27	13	25 ± 8.8
MEAN ²	46 ± 11.0a	28. ± 6.0a	31 ± 5.2a	
Perennial grass pasture				
Control 1	28	12	26	22 ± 5.7
Spring 2	17	6	13	12 ± 3.2
Summer 3	49	9	22	27 ± 12.0
Fall 4	23	8	19	17 ± 4.5
Control 5	34	7	11	17 ± 8.1
Mean	30 ± 7.8b	8 ± 2.3a	19 ± 4.2ab	
Bank levels**				
Low	8	19	7	11 ± 1.9a
Mid	22	4	15	14 ± 2.6a
Top	61	2	34	32 ± 8.4b

¹Year means and main effect means ± standard error

²Means for bank levels (column) and years (row) followed by the same letter were not significantly different ($p \leq 0.05$).

**Significant interaction ($p \leq 0.05$), bank level × year.

In channels, sampling points were located on 5 cross-sections on a representative straight stream reach in each pasture. Quadrats (0.25 m²) were located on both banks at bottom, middle, and top of the bank for a total of 30 (10 for each bank level) quadrats estimated and an additional 18 quadrats harvested and estimated in each pasture. Vegetation standing crop was summarized by bank level in each pasture yearly for 2 categories, all herbaceous vegetation and all perennial grasses. Perennial forbs and annual plants were major constituents of the difference between the 2 vegetation categories.

In flood plains, permanent quadrats (0.5 m²) were systematically located on 2 types of transects (those designated short were perpendicular to and within 50 m of channel and long transects extending beyond short transects to upland edge) extending from channel bank edge to uplands on both sides of the channel from representative straight and meander reaches in each pasture. Each pasture had 60 quadrats estimated and an additional 30 quadrats estimated and harvested. Vegetation standing crop in each pasture was summarized yearly for short and long transects for 2 categories, all herbaceous vegetation and perennial grasses. The difference between the 2 vegetation categories was primarily annual species.

In uplands 1 transect of 10 permanent quadrats (0.5 m²) was randomly located for estimation of biomass. An additional 5 quadrats were estimated and standing crop harvested. Vegetation standing crop was summarized yearly for each pasture for 2 categories, all herbaceous vegetation and blue grama. The difference between the 2 vegetation categories was primarily cool-season grasses.

Shrub Density

Shrub density on channel banks was determined yearly (1984–1986) in June on 1 meander and 1 straight reach of channel in each study pasture. Sample units (10 per pasture) were 3.05 × 3.05 m in area and extended from channel bottoms to top of banks. All big sagebrush, green rabbitbrush, and rubber rabbitbrush plants were counted regardless of size class. Densities were summarized for each species and total shrubs for the 2 reaches in each pasture for south and north aspect banks yearly. Reach summary data were further identified with respect to bank types (concave, convex, and straight) and reach type (straight and meander).

Root Biomass

Root biomass was sampled at low, middle, and top bank levels on straight and meander reaches on the same 5 cross-sections at

each reach as other channel vegetation samples in each of the pastures. A steel tube, 6.35 cm diameter, was used. Sampling was restricted to the top 15 cm of soil because extensive preliminary sampling during 1982 indicated roots on the channel banks were evenly distributed by depth to 45 cm. Schreiner (1987) found an even root distribution with depth on building banks being covered with sediment during overflow events of a cold desert perennial stream. Schreiner (1987) speculated that the even distribution of roots by depth on building banks was a result of the plant's continued regrowth of roots in adjustment to continued surface accumulated sediments and wide fluctuations in channel water levels. Sampling only 15 cm deep satisfied our needs for assessing surface stability of the bank as provided by any potential change in root mass. Duplicate cores were taken at each sample location. Cores were washed in a device patterned after Lauenroth and Whitman (1971). Washed material was subsequently ashed at 500° C. The material referred to as root biomass is the ash-free fraction of all roots or other organic material and is an index to root mass. Data were summarized yearly for each of the 2 reaches in each pasture by bank level and bank aspect (south and north facing) and further identified as to pasture, bank type (concave, convex, or straight) and reach type (straight or meander). Concomitant measurements of surface soil texture were made from low, mid, and top bank positions. Channel cross section measurements (10 per pasture) evaluated channel filling or cutting before, during, and after the study.

Design and Data Analysis

The treatment layout consisted of a completely random design with 3 replications of grazing and 2 control replicates. The location of control pastures was deliberate rather than random to encompass the variations possibly existing over the length of the study site and results, we feel, in a more conservative probability of Type 1 error than average (Hurlbert 1984). In this design the best indication of grazing effects are treatment (grazed-non grazed) × year interactions. Significant interactions are indicative of different kinds or degrees of response over years from grazed compare to nongrazed pasture vegetation (Green 1978). Lines plotting the treatments over years would have a different slope if interactions were significant. Using the interaction to assess grazing effect removes the chance of spurious treatment effect resulting from possible differences between pastures at the outset of the study. The experimental unit for data analysis was yearly pasture mean or

Table 2. Total herbaceous vegetation and perennial grass standing crop (g/m²)¹ of flood plain habitats in pastures on Middle Fork, 15-Mile Creek.

	Years				Mean
	1983	1984	1985	1986	
	(g/m ²)				
Total herbaceous pasture					
Control 1	12	14	6	12	11 ± 2.1
Spring 2	6	4	8	13	8 ± 1.4
Summer 3	14	13	19	27	18 ± 2.5
Fall 4	23	10	10	38	20 ± 5.2
Control 5	22	7	7	21	14 ± 2.9
MEAN ²	16 ± 2.6ab	9 ± 2.0a	10 ± 1.6a	22 ± 3.8b	
Perennial grass					
Control 1	6	5	2	3	4 ± 0.8
Spring 2	3	2	2	1	2 ± 0.4
Summer 3	6	3	3	4	4 ± 0.7
Fall 4	3	4	3	6	4 ± 0.9
Control 5	8	3	3	6	5 ± 0.9
MEAN	5 ± 0.7a	3 ± 0.6ab	2 ± 0.3b	4 ± 0.9ab	

¹Year means and main effect means ± standard error.

²Means for years (rows) followed by the same letter were not significantly different ($p \leq 0.05$).

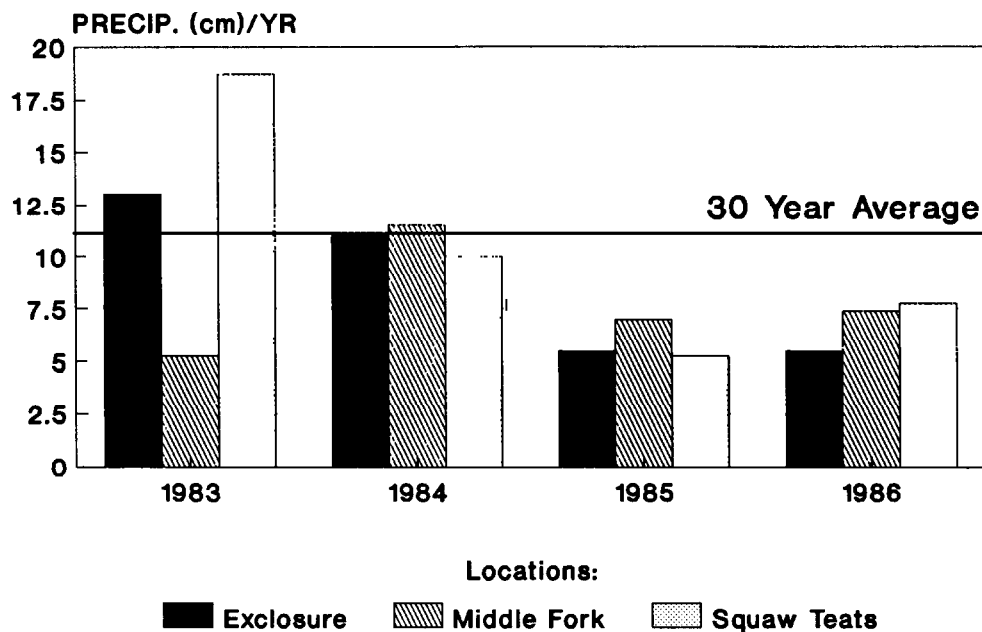


Fig. 1. Precipitation amount 1983-1986 for Middle Fork, 15-Mile Creek watershed, and 30-year average for Worland, May-September.

pasture mean of subcategories such as reach type or bank level.

The data analysis software was SAS (Freund et al. 1986). Repeated measures with H-F adjustment (years repeated) analysis of variance procedures were used. Duncan's multiple range test was used when significant differences occurred to separate means. A Type I error probability of $P \leq 0.05$ was used. Analysis for aboveground biomass was a factorial (4×2 for upland, $4 \times 2 \times 2$ for floodplain, $3 \times 2 \times 3$ for channel; factors: years, grazing-no grazing, transects (short-long for flood plain only), and bank level (low-middle-top for channel only). Shrubs were analysed with a $3 \times 2 \times 3 \times 2$ factorial; factors: years, grazing-no grazing, bank type (concave-convex-straight) and bank aspect (south-north). Root biomass analysis was a $3 \times 2 \times 3 \times 2 \times 3 \times 10$ factorial; factors were the same as for shrubs with the addition of bank levels (low-middle-top), and reaches (1-10). Root biomass of straight, convex, and concave bank types were analysed separately with a $3 \times 2 \times 2 \times 3 \times 5$ factorial; factors were years, grazing treatment, bank aspect (omitted for concave and convex banks), bank levels, and reaches.

Analyses included two-way interactions among factors. Three-way or higher interactions were not evaluated.

Results and Discussion

Aboveground Standing Crop Channel

Total herbaceous vegetation biomass of channels did not significantly vary with years, grazing treatment, or bank level. Interactions were not apparent (Table 1).

Perennial grass varied in response to years and bank level interaction but was not affected by grazing treatment. Perennial grass standing crop on the top bank was highest in 1984, generally the wettest of the 3 years; intermediate in 1986, with a similar number of precipitation events to 1984, but less precipitation; and lowest in 1985, with lowest number of events and precipitation amount in the growing season (Table 1, Figs. 1 and 2). Less standing crop than top bank on mid and low banks may have been due to bank disturbance and sediment deposition over vegetation by flow

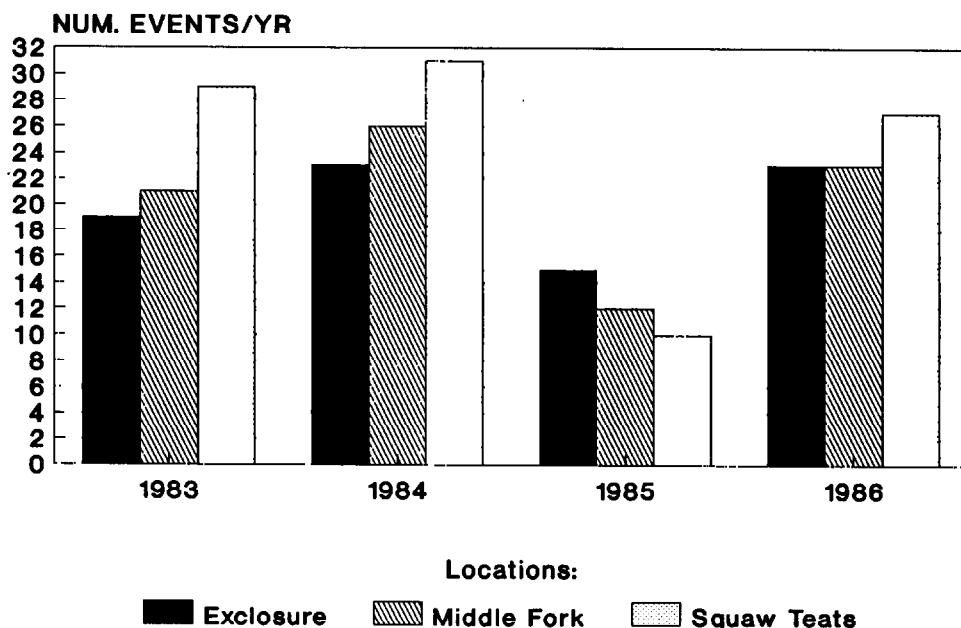


Fig. 2. Precipitation frequency for Middle Fork, 15-Mile Creek watershed, May-September, 1983-1986.

STRAIGHT CHANNEL REACHES

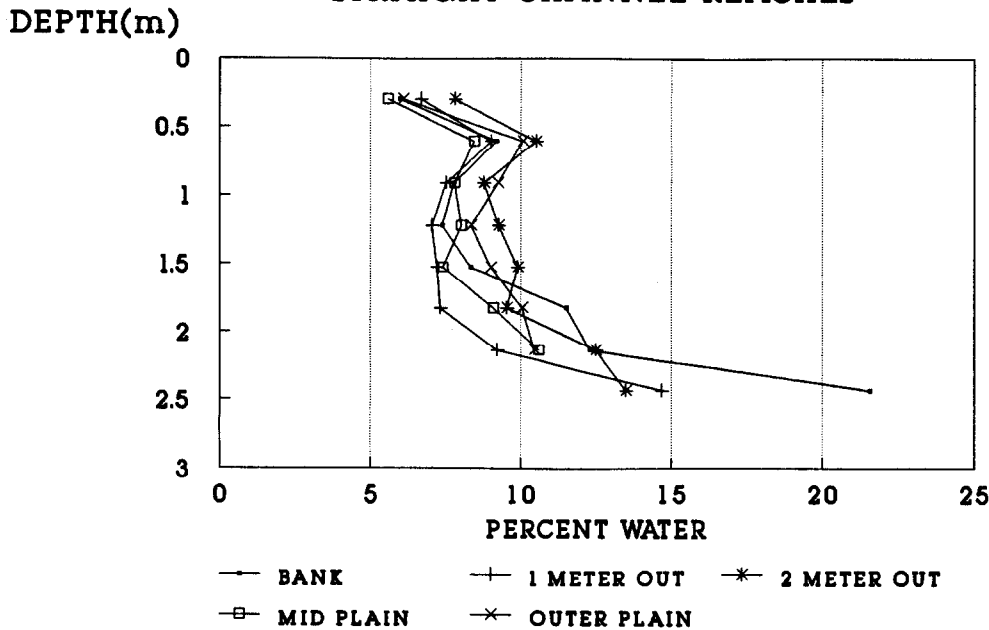


Fig. 3. Soil water content (% volume, mean of all seasons and years) at depths from floodplain surface and distances from the channel edge along 15-Mile Creek, 1983–1986.

events. Even though flow events commonly do not reach the top bank level, they may wet the root zone of plants of that position.

Significant bank level by year interaction with perennial grasses (Table 1) seemed to relate to precipitation frequency and amount. Highest standing crop on low bank and lowest standing crop on mid and top bank positions occurred in 1985 coincident with fewer precipitation events (Fig. 2) and somewhat lower precipitation than other years (Fig. 1) in contrast to the general trends in bank level and yearly standing crop. Reversal of trends in standing crop in 1985 may be due to less vegetation disturbance on the low bank by fewer stream flows coupled with less available moisture for plants on mid and top bank positions. Fewer events and lesser amounts of precipitation as in 1985 indicate a lower probability of streamflow from the large watershed contributing area to the narrow valley confines of the study area (Branson et al. 1981).

We observed no water recharge to the floodplains through channel banks after flow events but the neutron probe monitoring schedule never coincided with a flow event. Our neutron probe monitoring of soil water indicated no difference in soil water between positions located closer to the bank and those distant into the floodplain (Fig. 3). We conclude that the stream reach flowing through the study site is a perched stream, well above any watertable. Groundwater was observed in only one well above the upstream enclosure where outcrops of shale and sandstone occur along the narrows of the upstream valley.

We suggest that soil conditions on the channel floor and low bank, reached by most flow events, may more readily promote channel transmission loss vertically through the channel bottom than into bank slopes. The bottom of the channel is predominantly sand. Visual observations during all years showed that sand under the first few centimeters of the channel was moist during the growing season. In contrast, clay content is greater on the lower slope of the channel banks. The top of the slope is mostly sand.

Boelman (1989) studied surface/groundwater interactions on a cold desert perennial stream in Wyoming and found stream channels sealed by clay so water recharge to adjoining banks and through the channel bottom did not occur. Kraeger-Rovey (1991) also attributed channel sealing and retarded aquifer recharge to fine sized soil particles interspersed with larger gravels deposited in

horizontal lenses and suggested that these layers restricting flow may be formed by hydraulic discontinuity in streamflow or in place deposition of sediments. Minimal water movement into banks indicates that channel flow events represent a small supplement to local precipitation. Thus plant species on channel banks and adjacent floodplains would be limited to species only marginally different in water requirements from upland species except for those species like cottonwood able to exploit deep water sources.

Flood Plains

Total herbaceous standing crop and the perennial grasses component significantly varied with years of study (Table 2) but was not influenced by grazing treatment, transect type, or interactions. Annual variation in standing crop was only partially associated with summer precipitation amount (Fig. 1) and frequency (Fig. 2). The remainder of the variability was likely due to the influence of fall/winter precipitation and seasonal temperatures on germination and growth of annuals (represented by difference between herbaceous and perennial categories). The lack of graze-ungazed \times year interaction indicated no effect of grazing.

Uplands

Upland vegetation standing crop (Table 3) varied among years of study but not grazing treatment. Interaction of years with grazing treatment was not significant. The almost yearly decrease in standing crop of blue grama was somewhat associated with growing season precipitation amount (Table 3, Fig. 1). Herbaceous vegetation in contrast followed a similar pattern as flood plains herbaceous standing crop by increasing (although not significantly) in 1986 compared to 1984–1985. The difference in trend over years between all herbaceous and blue grama categories (annuals were a minor component of upland vegetation) in uplands suggests an increase in cool-season perennial grasses regardless of grazing or nonuse. Grazing in this study may represent a reduction of pressure from the season long grazing of the past and allowed cool-season grasses to increase. In addition the relatively low summer precipitation would have been unfavorable for blue grama.

Table 3. Total herbaceous vegetation and blue grama standing crop (g/m²)¹ of upland habitats in pastures on Middle Fork, 15-Mile Creek.

	Years				Mean
	1983	1984	1985	1986	
	(g/m ²)				
Total herbaceous pasture					
Control 1	21	11	11	11	14 ± 2.6
Spring 2	29	13	16	27	21 ± 4.0
Summer 3	24	22	15	42	26 ± 6.4
Fall 4	23	16	10	11	15 ± 3.0
Control 5	21	9	10	12	13 ± 2.8
MEAN ²	24 ± 1.4a	14 ± 2.3b	12 ± 1.2b	21 ± 6.7ab	
Blue grama pasture					
Control 1	19	11	11	1	10 ± 3.8
Spring 2	26	12	12	2	13 ± 5.0
Summer 3	19	12	13	12	13 ± 1.7
Fall 4	22	12	9	5	12 ± 3.6
Control 5	21	9	10	8	12 ± 3.0
MEAN	21 ± 1.3a	11 ± 0.6b	11 ± 0.8b	5 ± 2.1c	

¹Year means and main effect means ± standard error.

²Means for years (rows) followed by the same letter were not significantly different ($p \leq 0.05$).

Shrub Density

Shrub counts were dominated by seedlings and thus do not necessarily represent dominance relationships among species. These data may be more representative of reproductive potential. Unfortunately, we cannot partition seedlings from nonseedlings in our data set. All shrubs and each species varied among years. Grazing treatment main effect influenced only rubber rabbitbrush, but year × grazing treatment interaction influenced both green and rubber rabbitbrush. Bank type was not significant for any shrub. Bank aspect and year × bank aspect interaction significantly influenced all shrubs, big sagebrush, and green rabbitbrush.

Big sagebrush and green rabbitbrush densities were lowest in 1985, intermediate in 1984 and greatest in 1986 (Table 4) but

Table 4. Shrub density (plants/9.3 m²)¹ of channels in pastures on Middle Fork, 15-Mile Creek.

	Years		
	1984	1985	1986
	(plants/9.3 m ²)		
All shrubs			
MEAN	67 ± 4.3	33 ± 2.7a	98 ± 14.c
Big sagebrush			
MEAN	18 ± 1.8a	9 ± 1.1a	64 ± 13.5b
Green Rabbitbrush			
MEAN	27 ± 4.5ab	22 ± 2.3a	31 ± 2.8b
Rubber Rabbitbrush			
MEAN	21 ± 3.7a	2 ± 0.4b	3 ± 0.5b

¹Year means and main effect means ± standard error.

²Means for years (rows) followed by the same letter were not significantly different ($p \leq 0.05$).

numbers were apparently not closely related to amount or frequency of precipitation in those years. Unfortunately, distribution of precipitation events for winter and spring months for the site is not available. Rubber rabbitbrush was less abundant in 1985–1986 than in 1984 and therefore seemed positively associated with precipitation (Fig. 1 and 2).

Cattle grazing activity (soil disturbance, trampling, or consumption) appeared to cause green rabbitbrush amounts to decline (Table 5) in grazed pastures while increases occurred in ungrazed pastures from 1984 to 1986. Green rabbitbrush in the grazed pastures also did not respond similarly to yearly precipitation as that

Table 5. Green and rubber rabbitbrush density (plants/9.3 m²)¹ on channel banks in pastures on Middle fork, 15-Mile Creek.

	Years			MEAN ²
	1984	1985	1986	
	(plants/9.3 m ²)			
Green rabbitbrush pastures**				
Control (1)	17	23	41	27 ± 3.7a
Grazed (2,3,4)	34	19	25	26 ± 2.8a
Control (5)	20	31	38	29 ± 3.9a
MEAN	27 ± 4.5ab	22 ± 2.3a	31 ± 2.8b	
Rubber rabbitbrush pastures**				
Control (1)	39	2	5	15 ± 5.4a
Grazed (2,3,4)	15	2	2	6 ± 1.7b
Control (5)	23	2	3	9 ± 3.6ab
Mean	21 ± 3.7a	2 ± 0.4b	3 ± 0.5b	

¹Year means and main effect means ± standard error.

²Means for grazed-non grazed (column) and years (rows) followed by the same letter were not significantly different ($p \leq 0.05$).

**Significant interaction ($p \leq 0.05$), year and grazed-non grazed pastures.

in ungrazed pastures. Rubber rabbitbrush was more numerous in ungrazed pastures during the higher precipitation year, 1984, but declined in all pastures in drier years, 1985–1986.

Bank aspect influenced shrub density. Solar radiation subsequently affects soil surface temperatures and moisture. South facing banks, with greater exposure to solar radiation, should have had snow melted and soils warmed sooner in spring, providing germination sites while surface moisture levels were relatively high. This and low flow events that do not recharge upper bank slopes laterally would make winter and spring soil moisture recharge even more important as a yearly water supply for plants. The north facing bank would have had more snow accumulation and later melting. All shrub classes except rubber rabbitbrush had greater numbers (Table 6) on the south facing bank. Significant bank aspect × year interactions for all shrubs, big sagebrush and green rabbitbrush (Table 6) indicated that fluctuations in yearly numbers of plants were much greater on the south facing banks. This is expected because temperature and moisture regimes necessary for

Table 6. Shrub density (plants/9.3 m²) of channel bank aspects (north or south facing) in pastures on Middle Fork, 15-Mile Creek.

Shrub species	Years			MEAN ²
	1984	1985	1986	
Bank Aspect	(plants/9.3 m ²)			
All**				
South	73	35	139	82 ± 10.8a
North	60	32	57	50 ± 3.3b
MEAN	67 ± 4.3b	33 ± 2.7a	98 ± 14.2c	
Big sagebrush**				
South	17	9	101	42 ± 10.3a
North	19	10	26	19 ± 1.9b
MEAN	18 ± 1.8a	9 ± 1.1a	64 ± 13.5b	
Green rabbit-brush**				
South	38	24	34	32 ± 3.1a
North	17	20	27	21 ± 1.9b
MEAN	27 ± 4.5ab	22 ± 2.3a	31 ± 2.8b	
Rubber rabbit-brush				
South	18	2	3	8 ± 2.3a
North	24	2	3	10 ± 2.4a
MEAN	21 ± 3.7b	2 ± 0.4a	3 ± 0.5a	

¹Means and main effect means ± standard error.

²Means for aspects (column) and years (rows) followed by the same letter were not significantly different ($p \leq 0.05$).

**Significant interaction ($p \leq 0.05$), between years and bank aspect.

germination and growth of plants are more variable on this aspect.

Root Biomass

Belowground biomass over all bank types (Table 7) was significantly influenced by year effects, bank type, bank level, reach location, and reach × bank aspect interaction but not by grazing, bank aspect, or other interactions. Straight bank type root biomass

Table 7. Root biomass (g/m²)¹ in surface 15 cm of soils of channel banks in pastures on Middle Fork, 15-Mile Creek.

	(g/m ²)
a. Years	
1984	423 ± 32a ²
1985	361 ± 19ab
1986	324 ± 30b
b. Pastures ³	
Control (1,5)	386 ± 36a
Grazed (2,3,4)	358 ± 20a
c. Bank type	
concave	419 ± 38a
convex	318 ± 32b
straight	371 ± 21ab

¹Mean and main effect means ± standard error.

²Means for years, pastures, or bank type (columns) followed by the same letter were not significantly different ($p \leq 0.05$).

³Pastures were 1, downstream control, 2, spring grazed, 3, summer grazed, 4, fall grazed, and 5, upstream control.

(Table 8 a-d) was affected by years, bank levels, and reaches but not by other factors or interactions. Convex bank type root biomass (Table 8) was affected by years only. Concave bank type root biomass was not affected by any factor analysed.

Biomass of roots and other similar sized organic matter in channel bank soils increases bank stability and provides resistance to erosion during flow events (Whitlow and Harris 1979, Stevens and Waring 1985, Bohn 1989, Henszey et al. 1989). Belowground biomass of all bank types decreased by 23% from 1984 to 1986 (Table 7a and 8), regardless of grazing (Table 7b), in a period of decline in growing season precipitation of about 25% (Fig. 1). The

Table 8. Root biomass (g/m²)¹ in surface 15 cm of soils of straight and convex channel banks in pastures on Middle Fork, 15-Mile Creek.

	(g/m ²)
Straight banks	
a. Years	
1984	431 ± 41b ²
1985	376 ± 25ab
1986	305 ± 36a
b. Pastures ³	
Control (1,5)	413 ± 50a
Grazed (2,3,4)	342 ± 21a
c. Bank level	
Low	378 ± 30ab
Middle	426 ± 37b
Top	307 ± 37a
d. Reach number in (pasture ³)	
2(1)	545 ± 50c
4(2)	419 ± 40b
6(3)	353 ± 25ba
8(4)	256 ± 32a
10(5)	281 ± 47a
Convex banks	
Years	
1984	417 ± 78b
1985	316 ± 36ab
1986	220 ± 28a

¹Main effect means ± standard error.

²Means for years grazed-non grazed pastures, bank level, and reaches followed by the same letter were not significantly different ($p \leq 0.05$). Means are not shown for factors (except grazing) where differences between levels were not significant.

³Pastures were 1, downstream control, 2, spring grazed, 3, summer grazed, 4, fall grazed, and 5, upstream control.

declines suggest a relatively strong, positive relationship between belowground biomass and precipitation. A reduction in precipitation could have reduced moisture available for growth generally and additionally increased sediment deposition on the low bank position through reducing flow velocity in smaller flow events; however, cross section measurements do not indicate any major sediment deposition during the period (Fig. 4). Small additions of sediment were evident when nails and other transect markers on the low bank were covered. Increased sediment would result in the appearance of a decrease in roots as only the top 15 cm were sampled. Roots in concave banks however did not change over years as did the straight (Table 8a) and convex bank type (Table 8).

Concave banks had the highest biomass of the 3 types (Table 7c) while convex banks had the least. If bank erosion does not occur and a meander remains stable, the concave bank has the deepest point in the thalweg and the point of last deposition of fine sediment as flow diminishes (Andrews 1982). Because this stream is ephemeral, the last flowing or ponding of water would occur on the outside bend of meanders. The meander pool has a likelihood of being sealed and loss of the water to the aquifer would be slow (Boelman 1989). Last flowing or ponded water along the deepest reach of the thalweg could benefit growing vegetation along the concave curve of meander low banks. During higher flows, sediments are removed from the concave pool and deposited on the convex banks (Leopold and Langbein 1966). The convex bank, inside the meander bend, would receive the least amount and uniformity of water for plant growth. In addition convex banks receive the greatest sediment deposition during flow events thus continually covering roots to a deeper depth.

Root distribution on straight reaches varied with bank level. The middle bank level had root abundance significantly greater than top bank but similar to low bank level (Table 8c). This effect may be due to differential frequencies of wetting. The lower bank levels undoubtedly are wetted more frequently than the top bank posi-

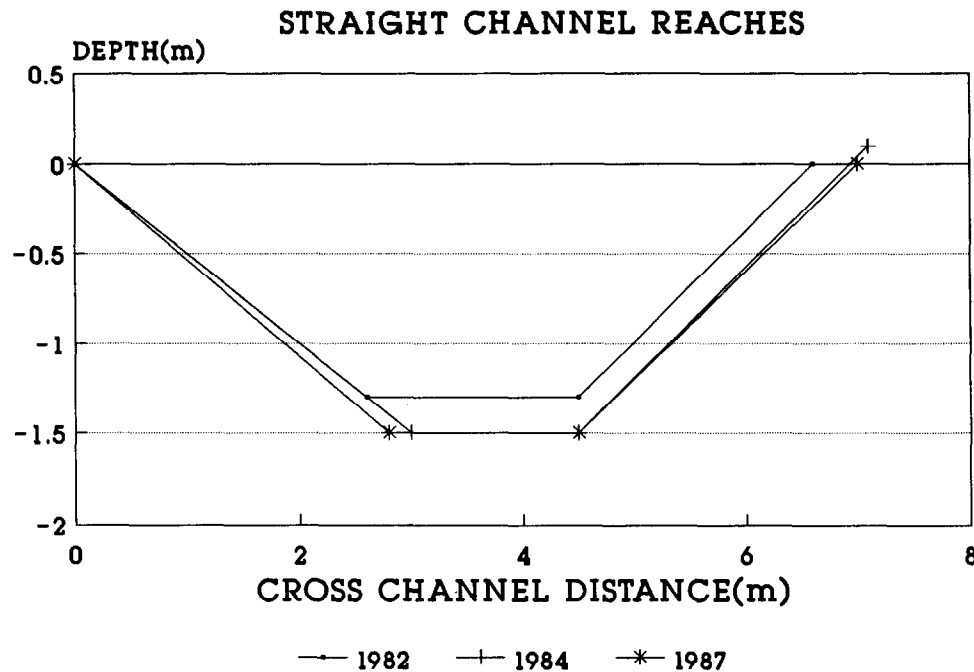


Fig. 4. Channel profile of straight reaches before, during, and after grazing of 15-Mile Creek.

tion. Middle bank vegetation roots have less distance to grow to tap low bank water supplies than top bank plants during low flow events. Middle bank plants are also closer to the wetted channel perimeter during low flows and can take advantage of the minimal lateral flow of soil water.

Low banks trapped sediments in the band of upright Canada wildrye growing near the channel bottom and were subject to more frequent small flows. The mid bank position vegetation was dominated by inland saltgrass and because of its low growth form may offer less resistance to flow and low ability to trap sediment. The top bank level had more shrubs and upright perennial grasses and was near the height where banks cease to confine high flow events. Because of water spreading into flood plains and resultant decreases in flow velocity during high flow events, large amounts of sediment can be deposited at the junction of top bank and flood plain. Overbank flow events that could have deposited sediment on the top bank and floodplain occurred only twice during study years, both before 1984.

We found no evidence that grazing influenced belowground biomass in the course of our study (Tables 7b and 8b). Comparison of straight reaches (Table 8d) indicates the general decline in belowground biomass progressively upstream from the downstream control pasture (1). The upper pasture is located in the stream gradient transition zone just below a valley constriction. An adjustment in flow dynamics is being made from 0.06% to 0.03% stream gradient through the study area. Deposition of sediments generally occurs as flow velocity decreases with gradient. Deposited sediments should decrease and be finer textured further downstream from the transition area. However, no differences in cross section area of the channels or soil texture occurred to support this hypothesis (Fig. 4).

Conclusions

Grazing at the rates and seasons used in this study appeared to be unimportant to vegetation dynamics of the ephemeral channels. Relatively moderate utilization levels (Smith et al. 1989 and 1992) during the years of grazing treatments did not suggest negative changes should occur. Vegetation dynamics above- and belowground appeared to be mainly a function of yearly fluctuation in

moisture regimes and locations in and along the channel. Aboveground biomass and belowground biomass decreased with decreasing growing season precipitation amount. Shrub density seemed most closely related to precipitation frequency probably through its effect on seedlings of shrubs.

Channel bank level affected both above- and belowground biomass but differently. Aboveground perennial grasses seemed most responsive to possible vegetation removal or covering by low events, being less abundant on middle and low bank. Belowground biomass in contrast had highest abundance on middle and low bank levels perhaps in response to increased water provided by flow events. Possibly faster accretion of sediment on the low and top bank may also tend to decrease belowground biomass in the surface 15 cm of soil compared to the mid level position. Belowground biomass was reduced at upstream locations in a steep to flatter stream channel gradient change.

Sediment accretion rates in channels did not indicate a great potential for grazing to substantially alter this channel in the short term. For land managers addressing appropriateness of public land uses in the Bighorn Basin and other cold desert areas, a noteworthy aspect of this study is that moderate grazing has not shown detrimental effects on vegetation.

Literature Cited

- Andrews, E.D. 1982. Bank stability and channel width adjustments, East Fork River, Wyoming. *Water Resource. Res.* 18:1184-92.
- Bohn, C. 1989. Management of winter soil temperatures to control stream-bank erosion. p. 69-71. *In*: Gresswell, R.E.; B.A. Barton; J.L. Kershner (eds.) *Practical approaches to riparian resource management: an educational workshop*; May 8-11, 1989. Billings, Mont. USDI BLM.
- Chaney, E., W. Elmore, and W.S. Platts. 1990. *Livestock grazing on western riparian areas*. Produced for US-EPA by Northwest Resource Information Center, Inc., Eagle, Ida.
- Branson, F.A., G.F. Gifford, K.G. Renard, and R.F. Hadley. 1981. *Rangeland Hydrology*. Soc. Range Manage. Denver, Colo. p. 73-110.
- Boelman, S.D. 1989. *Groundwater modeling of cold desert riparian zones along a stream tributary to the upper Colorado River*, Masters Thesis, Civil Engineering, Univ. Wyo., Laramie.
- Clary, W.P., and B.F. Webster. 1989. *Managing grazing of riparian zones in the intermountain region*. USDA Forest Serv. Intermount. Forest and Range Exp. Sta. GTR. INT-263.

- Cooke, R.U., and R.W. Reeves. 1976. Arroyos and environmental change in the American south-west. Clarendon Press. Oxford. 213 p.
- Debano, L.F., and L.J.J. Schmidt. 1990. Potential for enhancing riparian habitats in the southwestern United States with watershed practices. *Forest Ecol. Manage.* 33:385-403.
- Dorn, R.D. 1988. Vascular plants of Wyoming. Mountain West Publish., Box 1471, Cheyenne, Wyo. 82003.
- Elmore, W., and R.L. Beschta. 1987. Riparian areas: perceptions in management. *Rangelands* 9:260-265.
- Freund, R.J., R.C. Littell, and P.C. Spector. 1986. SAS system for linear models. SAS Institute Inc. Cary, N.C.
- Fisser, H.G. 1982. Vegetation response to grazing management strategies on semiarid shrublands of Wyoming. p. 162-165. *In:* D.D. Briske and M.M. Kothman (eds.) *Proc. Nat. Conf. on Grazing Techniques*. Texas A&M Univ. College Station. Nov. 10-13.
- Graf, W.L. 1985. The Colorado River-instability and basin management. Assoc. Amer. Geograph. Washington, D.C.
- Green, R.H. 1978. Sampling design and statistical methods for environmental biologists. John Wiley & Sons. N.Y.
- Heede, B.H. 1977. Case study of a watershed rehabilitation project: Alkali Creek, Colorado. USDA Forest Serv. Rocky Mountain Forest and Range Exp. Sta. Ft. Collins, Colo. Res. Paper RM189.
- Heede, B.H. 1980. Stream dynamics: an overview for land managers. USDA Forest Serv., Rocky Mountain Forest and Range Exp. Sta. Fort Collins, Colo. Gen Tech. Rep. RM-72.
- Heede, B.H. 1982. Gully control: Determining treatment priorities for gullies in a network. *Environ. Manage.* 6:5441-51.
- Henszey, R.J., T.A. Wesche, and Q.D. Skinner. 1989. Evaluation of the state-of-the-art streambank stabilization. Rep. to Wyoming Dep. Environ. Quality, Cheyenne, Wyo.
- Hooper, R., B.P. Van Haveren, and W.L. Jackson. 1987. The Sheep Creek Resource Conservation Area Project. *Proc., XVIII Conf. Int. Erosion Control Assoc.* Reno, Nev. February 26-27, p. 117-126.
- Hurlbert, S.H. 1984. Pseudoreplication and the design of field experiments. *Ecol. Monog.* 54:187-211.
- Kauffman, J.B., and W.C. Krueger. 1984. Livestock impacts on riparian ecosystems and streamside management implications...A review. *J. Range Manage.* 37:430-437.
- Kraeger-Rovey, C. 1991. Studies of surface and ground water interactions in two Arizona stream systems. p. 415-424. *In:* *Proc. Colorado Water Engineering and Manage. Conf.* February 27-28, Colo. Water Resour. Res. Inst. and Office of State Engineer, Denver, Colo.
- Lauenroth, W.K., and W.C. Whitman. 1971. A rapid method for washing roots. *J. Range Manage.* 34:308-309.
- Leopold, L.B., and W.B. Langbein. 1966. River meanders. The physics of everyday phenomena. *Sci. Amer.* 214:60-70.
- Myers, L.H. 1989. Grazing and riparian management in southwestern Montana. p. 117-120. *In:* Gresswell, R.E.; B.A. Barton; J.L. Kershner (eds.) *Practical approaches to riparian resource management: an educational workshop*. May 8-11. Billings, Mont. USDI BLM.
- Pickett, S.T.A. 1980. Non-equilibrium coexistence of plants. *Bull. Torrey Bot. Club.* 107:238-248.
- Platts, W.S. 1982a. Sheep and cattle grazing strategies on riparian stream environments. p. 251-270. *In:* J. Peck, (ed.) *Wildlife-Livestock Relationships Symp. Proc. Univ. Idaho Forest, Wildl., and Range Exp. Sta. Moscow.*
- Platts, W.S. 1982b. Livestock and riparian fishery interactions: What are the facts? *Trans. North Amer. Wildl. Natur. Resources Conf.* 47:507-515.
- Platts, W.S. 1989. Compatibility of livestock grazing strategies with fisheries. p. 103-110. *In:* Gresswell, R.E.; Barton; J.L. Kershner (eds.) *Practical approaches to riparian resource management: an educational workshop*; May 8-11. Billings, Mont. USDI BLM.
- Ponce, V.M., and D.S. Lindquist. 1990. Management of baseflow augmentation: a review. *Water Resources Bull.* 26:259-268.
- Schreiner, T.A. 1987. Characterization of root biomass and soil moisture in riparian zones of southeastern Wyoming. Masters Thesis. Range Manage. Dep. Univ. of Wyoming, Laramie.
- SCS. 1988. Technical guide, section IIB, major land resource area (32), 5-9th Big Horn Basin (5-9 BH). USDA SCS. Casper, Wyo.
- Skinner, Q.D., J.L. Dodd, J.D. Rodgers, and M.A. Smith. 1985. Antidesertification of riparian zones and control of nonpoint source pollution. p. 382-386. *In:* *Perspectives on nonpoint source pollution*. US-EPA. 440/5-85-001, Office of Water Regulations and Standards, Washington, D.C.
- Skovlin, J.M. 1984. Impacts of grazing on wetlands and riparian habitat: a review of our knowledge. p. 1001-1003. *In:* *Nat. Acad. Sci./Nat. Res. Counc. Developing strategies for rangeland management*. Westview Press, Boulder, Colo.
- Smith, M.A., Q.D. Skinner, J.L. Dodd, J.D. Rodgers, and R.E. Siekert. 1989. Channel and riparian zone responses to cattle grazing along 15-Mile Creek, Bighorn Basin, Wyoming. Final Rep. to Wyoming Dep. Environ. Quality. Range Manage. Dept. Univ. Wyoming. Laramie. 82071.
- Smith, M.A., J.D. Rodgers, J.L. Dodd, and Q.D. Skinner. 1992. Habitat selection by cattle along an ephemeral channel. *J. Range Manage.* 45:385-390.
- Stevens, L.E., and G.L. Waring. 1985. The effects of prolonged flooding on the riparian plant community in Grand Canyon. p. 81-86. *In:* J.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Ffolliott, and R.H. Hamre (tech. coord.) *Riparian ecosystems and their management: reconciling conflicting uses*. First North Amer. Riparian Conf. 16-18 April, 1985, Tucson, Ariz. USDA Forest Serv. Rocky Mountain Forest Range Exp. Sta. Gen. Tech. Rep. RM 120.
- Swanson, S.R., S.L. Miles, and K. Genz. 1988. Classifying rangeland riparian areas: the Nevada task force approach. *J. Soil and Water Conserv.* 43:259-263.
- Szaro, R.C. 1990. Southwestern riparian plant communities: site characteristics, tree species distributions, and size class structures. *Forest Ecol. Manage.* 33:315-334.
- U.S. Bureau of Land Management. 1987. Riparian area management policy. USDI BLM. Washington, D.C.
- U.S. General Accounting Office. 1988. Public rangelands: some riparian areas restored but widespread improvement will be slow. Report to Congr. Requestors, GAO/RCED-88-105, Washington, D.C.
- Van Haveren, B.P., and W.L. Jackson. 1986. Concepts in stream riparian rehabilitation. *Trans. North Amer. Wildl. and Nat. Resources Conf.* 51:280-289.
- Whitlow, T.H., and R.W. Harris. 1979. Flood tolerance in plants: a state-of-the-art review. USDOD Army Eng. Waterways Exp. Sta. CE. Vicksburg, Miss. Tech. Rep. E-79-2.

Effect of grazing strategies and pasture species on irrigated pasture beef production

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Abstract

Irrigated cool-season grasses can be used as complementary forages with other forage resources. Improved efficiency of animal production from irrigated pasture could increase their utility as a complementary forage. The factors of species composition, grazing management, irrigation, and fertilization all have the potential to affect efficiency of irrigated pasture production. Specific objectives of this study were: (1) to determine the effect of deferring irrigated pasture and restricting irrigation water and fertilization during mid-summer on pasture and livestock production; and (2) to evaluate different pasture stands for adaptability to different grazing strategies. Eight, adjacent 1.25-ha pastures were established as 2 replications of 2 different pasture stands grazed under 2 grazing management strategies. Pasture stands consisted of intermediate wheatgrass (*Agropyron intermedium* Host. Beauv.) as a monoculture (IWG) and a 4-species mixture (MIX) of orchardgrass (*Dactylis glomerata* L.), smooth bromegrass (*Bromus inermis* Leyss.), meadow bromegrass (*Bromus biebersteinii* R. & S.), and Garrison creeping foxtail (*Alopercurus arundinaceus* Poir.). Grazing treatments with yearling steers consisted of season-long grazing (SLG) and a graze-defer-graze (GDG) strategy. For the GDG pastures, 38% less fertilizer and 34% less irrigation water were applied, but animal days of grazing were reduced only 16% over the 3-year study. Animal weight gains were comparable between pasture types when considered over the entire grazing season but were higher for IWG early in the growing season and for MIX late in the season. Persistence of pasture stand was better for the MIX pastures than IWG pastures which were invaded by annual weeds after the first grazing season. Highest gains ha^{-1} were from the SLG pastures because of more days of grazing, but animal productivity was not proportionally reduced for the GDG strategy. The MIX pastures were suited for either grazing strategy.

Key Words: animal gain, forage quality, carrying capacity, complementary forage

In many areas of the Great Plains, irrigated pasture of cool-season grasses can increase the forage options available to livestock producers. Cool-season pastures permit development of complementary forage systems with associated rangeland and other forage resources. Complementary forage systems have the potential to increase production per unit of land, improve animal performance, provide an alternative to harvested feeds, and increase forage availability during drought (Nichols 1989). The impact of using complementary forages with range has been documented (Allen 1972, Anderson and Jernstedt 1971, Ford et al. 1986, Hart et al. 1988, Lodge 1963, McIlvain and Shoop 1973,

Moore 1970, Smoliak 1968). However, management practices used and type of forage resources considered are highly variable depending on locality and goals of the producer. Irrigated cool-season grasses are a viable complementary forage in many areas of the Great Plains because of the associated rangeland dominated by warm-season grasses.

Animal production can be high from irrigated cool-season grasses under excellent management. Animal gain exceeded 800 kg ha^{-1} in 13 out of 35 trials at 10 different locations (Nichols and Clanton 1985). A major constraint to greater use of irrigated pasture has been high production cost in relation to other forage resources, not their productivity or utility as a complementary forage. Based on a projected level of production of 30 animal units months (AUM) ha^{-1} , total costs per AUM in southwest Nebraska have been estimated at \$22.20 (Agricultural Economics Staff 1991). This production level represents the upper limits that can be achieved with season-long grazing under excellent grazing management with high water and fertilizer applications (Nichols and Clanton 1985).

A previous study at North Platte, Neb., has shown that the least efficient period for animal production from irrigated cool-season grasses was during mid-summer when pasture and animal production declined, but irrigation water and fertilizer requirements remained high in order to stimulate pasture production (Nichols and Moore 1977). Cool-season grasses decline in productivity during periods of high ambient temperature. This factor has a major impact on the overall efficiency of irrigated pasture. If this period of decreased pasture productivity and efficiency could be circumvented by using other forages, the positive aspects of irrigated, cool-season grasses as a component of a complementary forage system could be improved.

The objectives of this study were to compare pasture and animal production from irrigated pasture seeded to different pasture species which were grazed season-long, compared to deferred during mid-summer without irrigation or fertilization.

Methods and Materials

The study area was located on the University of Nebraska West Central Research and Extension Center at North Platte. Pastures were on nearly level class I land with deep, fertile soils classified as Cozad silt loam (*Typic Haplustolla*). Field plot design was a factorial arrangement of 2 pasture types under 2 summer grazing management strategies, replicated twice and conducted over 3 grazing seasons during 1985-87.

Eight adjacent, 1.25 ha irrigated pasture units were randomly assigned to 2 replications of 2 types of pasture and seeded to: (1) a monoculture of 'Slate' intermediate wheatgrass (*Agropyron intermedium* Host. Beauv.) designated as IWG; and (2) a mixture (MIX) of 'Sterling' orchardgrass (*Dactylis glomerata* L.), 'Lincoln'

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Table 1. Precipitation (Mar.-Sept.), irrigation water, and fertilizer for season-long (SLG) and graze-defer-graze (GDG) grazing strategies.

Year	Precipitation ---mm---	Irrigation		Nitrogen	
		SLG	GDG	SLG	GDG
1	287	297	198	290	180
2	401	315	178	290	180
3	386	290	221	290	180
Average	358	300	198	290	180

smooth brome grass (*Bromus inermis* Leyss.), 'Regar' meadow brome grass (*Bromus biebersteinii* R. & S.), and 'Garrison' creeping foxtail (*Alopecurus arundinaceus* Poir.). Seeding rates (pure live seed basis) were 25 and 18 kg ha⁻¹ for IWG and MIX, respectively. Results from an earlier study which evaluated 8 grasses for irrigated pasture were used as a basis for the selection of plant materials for this study (Nichols et al. 1976). Pastures were planted in late August of 1984 and grazing commenced in May of 1985. Excellent stands were established on both pasture types when grazing trials commenced.

Within each pasture type, 2 summer grazing management strategies were randomly assigned: (1) season-long grazing (SLG); and (2) a graze-defer-graze (GDG) strategy. Season-long grazing consisted of maintaining steers on irrigated pasture for the 148-day grazing season starting about 1 May. The graze-defer-graze treatment was identical, with the exception that steers were removed from irrigated pasture during the summer deferment period. The grazing season was divided into 3 periods for data collection, which corresponded to the pasture management strategy for the GDG treatment. During period I, all pastures were grazed for 60 days. During period II, all cattle assigned to the GDG treatment were removed from pasture and put on a subirrigated meadow site for 56 days which was lush and immature and considered to be similar to forage on the study site. All pastures were grazed during period III for 32 days.

Crossbred yearling steers with an average initial live weight of 275 kg were used to evaluate the different pasture treatments. Steers were adapted to similar forage for 14 days before being placed on the test pastures. Stocking rates were 10 steers/pasture unit (8 head ha⁻¹) during year 1 and 8 steers (6.42 head ha⁻¹) for

years 2 and 3. This initial base stocking rate was maintained on all pastures throughout the grazing periods. These steers were designated as "tester" steers from which average daily gain (ADG) was calculated from weights taken at the start of the grazing season and at the end of each period following 16 hours without feed and water. A group of similar steers was maintained on extra irrigated pasture of the same forage species and used as "put and take" animals for regulating forage availability. Steer days of grazing from the "put and take" group were added to steer days from the "tester" group for total animal days ha⁻¹ (ADH), but were not used for calculation of ADG. Weight gain ha⁻¹ (WGH) was calculated as ADG × ADH. Procedures and computations essentially follow those suggested by Mott (1959) and summarized by Matches (1970).

Each pasture unit was rotationally grazed in 5 equal-size paddocks, (0.25 ha) divided by single-strand electric fence. Rotation of steers among paddocks was flexible, but was generally on a 5- to 7-day basis, which allowed 25 to 28 days for pasture recovery. Relative forage availability was estimated daily for all pastures. Since the entire experimental area was small (10-25 ha), ocular estimates of forage availability could be readily made by comparing forage height and mass. Steers from the "put and take" group were placed on pasture to maintain comparable forage availability among treatment pastures. Steers were moved to a fresh paddock when only about 60% of the forage was utilized in order to allow the steers a high degree of selectivity and to maintain pasture productivity.

Irrigation was by solid-set, sprinkler irrigation which watered each treatment pasture independently. Irrigation was used to supplement natural precipitation to maintain near optimum grass growth. The soil water status was monitored by gravimetric sampling. All pastures received equal irrigation amounts throughout the growing season with the exception of GDG pastures, which were not irrigated during period II. Difference in total irrigation water applied to the grazing strategies is presented in Table 1. One week before the start of grazing for period III, irrigation water was applied to the GDG pastures to equalize the soil water status among the treatment pastures.

Nitrogen fertilizer was applied equally across all pastures with the exception of period II. Total nitrogen (N) applied each year was 290 kg ha⁻¹ for the SLG pasture and 180 kg ha⁻¹ for the GDG pasture (Table 1). Granular, ammonium nitrate (NH₄NO₃) was

Table 2. Average daily gain (kg) of steers grazing pastures seeded to either a mixture of 4 species (MIX) or intermediate wheatgrass (IWG) and grazed either season long (SLG) or graze-defer-graze (GDG).

Period	Pasture type			Management strategy			SEM ¹
	MIX	IWG	P>F	SLG	GDG	P>F	
	----- (kg) -----						
Year 1							
I	1.06	1.98	0.022	1.20	1.06	0.025	0.06
II ²	0.70	0.68	0.798	(0.69)	(0.85)	—	0.13
III	0.72	0.73	0.920	0.80	0.65	0.101	0.11
Cumulative	0.89	0.93	0.186	0.91	0.90	0.660	0.04
Year 2							
I	0.44	0.56	0.184	0.55	0.44	0.226	0.12
II ²	0.96	0.84	0.104	(0.92)	(0.106)	—	0.08 ³
III	0.60	0.44	0.089	0.64	0.41	0.030	0.11
Cumulative	0.70	0.68	0.687	0.71	0.67	0.405	0.06
Year 3							
I	0.50	0.59	0.083	0.58	0.53	0.246	0.06
II ²	0.94	0.68	0.032	(0.82)	(1.11)	—	0.07
III	0.84	0.64	0.029	0.86	0.62	0.017	0.10
Cumulative	0.79	0.72	0.088	0.73	0.77	0.317	0.05

¹Standard error of the mean.

²Statistical analysis included only average daily gain of steers on SLG. Values under management strategy columns are for information only.

applied to all pastures before "green up" in the spring at the rate of 56 kg ha⁻¹. This was followed by 20 kg ha⁻¹ applications of 28% N solution as urea-ammonium nitrate [(NH₂)₂CO • NY₄NO₃] through the irrigation system when irrigating on about 10-day intervals. Laboratory soil analyses did not indicate a need for other nutrients during the study period.

Statistical procedures used were SAS (1985). Period I, period III, and trial ADG, ADH, and WGH data were analyzed using a model appropriate for a replicated 2 × 2 factorial design. The initial model included main effects for pasture type, grazing management and year, and the associated interactions, as well as pasture type by grazing management within replicate. This term was used as the error term for testing pasture type, grazing management, and their interaction. A significant year effect ($P < 0.05$) was observed for each parameter; therefore, data were subsequently analyzed within each year using a model including pasture type, grazing management, and the associated interaction. A significant pasture type by grazing management interaction was present for ADH for period I and trial data; therefore, simple effects were analyzed using a completely randomized design with separation of means by protected least significant difference. Animals on the GDG treatment were removed from the study during period II; therefore, data from this period was analyzed as a completely randomized design using only the data from the SLG treatment.

Results

The experimental design of the study stipulated that no fertilizer or irrigation water be applied to the GDG treatment during period II when grazing was deferred. This resulted in 38% less total nitrogen and 34% less irrigation water applied over the grazing season each year compared to the SLG pasture (Table 1). The reduction of these 2 primary production inputs in relation to the overall pasture productivity is important in evaluating the feasibility of the practices. Costs associated with these inputs would be highly variable depending on irrigation and fertilization practices and prices and should be evaluated on an individual ranch or farm basis. In addition, if the economics are viewed within the context of a production system, the forage resources grazed during the deferment period could include many different forage options, which would have a major impact on overall pasture and animal production.

Average Daily Gains

Significant year by treatment interactions ($P < 0.01$) were present over all periods for ADG; however pasture type by grazing management interaction was not present ($P > 0.40$). The distribution pattern of steer gains over the entire growing season was influenced by pasture type (Table 2). In year 1, steers grazing IWG gained 86% more during the early growing season (period I) than steers grazing MIX, but as the growing season progressed, there were no differences between pasture types during the last 2 grazing periods. The tendency for better steer gains for IWG during period I continued for years 2 and 3, but differences became less pronounced between pasture types. By mid-season there was a shift to better animal gains from MIX. Average daily gain was higher during the periods II and III for MIX than IWG pastures for years 2 and 3. Average daily gains for MIX were 14 and 36% higher for periods II and III respectively for year 2, and 38 and 31% higher for the same periods for year 3. Cumulative ADG was not different for years 1 or 2, but the MIX pastures produced higher gains during year 3. When considering steer performance over the entire growing season, the change from better gains early by IWG to better gains mid to late season by MIX tended to mask any differences in animal performance by periods between the 2 pasture types.

Based on animal performance, these data indicated that IWG provided a higher quality forage during the early part of the

growing season, whereas MIX pastures produced a better quality forage during the mid to late part of the growing season. The fact that management practices and forage availability were similar for both pasture types supports this interpretation.

The effects of management strategy on steer performance were not different ($P > 0.86$) over the 3-year study. Overall ADG for each year was not different between SLG and GDG for any of the 3 years, but there were significant effects associated with time of grazing (periods) within a specific year (Table 2). Season-long grazing produced higher ADG during period I of the first year, but not during succeeding years.

The most obvious differences in steer performance were for period III. Steers on the GDG pastures gained 0.15, 0.23, and 0.24 kg day⁻¹ less than steers on the SLG pastures for years 1, 2, and 3, respectively (Table 2). These lower gains were considered a response to higher gains realized during period II when the GDG steers were rotated off the test pasture to a lightly stocked subirrigated meadow during the deferment period. Evidently, a higher quality diet and/or increased forage intake was realized by the steers on the subirrigated meadow during period II which had a negative effect on gains during period III. Mean weight gains for management strategies are shown in parentheses in Table 2, but were not considered part of the treatments and were not included in the analyses.

Based on these data, it would not be anticipated that grazing strategy alone would have an effect on steer gains while on irrigated pastures. If steers were rotated from irrigated pasture during the deferment period to other forages (either higher or lower in quality), a differential response in steer gains could be anticipated when rotated back to irrigated pasture.

Animal Days/Hectare

Total animal days of grazing (animal number × days grazed) provided a measure of pasture productivity as affected by pasture type and grazing strategy. A pasture type by grazing strategy interaction ($P < 0.05$) was present overall and for period III; therefore, simple effect means are presented for each year and period (Table 3). Total ADH was higher during years 1 and 3 for the SLG management strategy than for GDG over both pasture types (Table 3). During year 2, total ADH was higher for the SLG-MIX

Table 3. Animal days ha⁻¹ (ADH) simple effect means for pastures seeded to either a mixture of 4 species (MIX) or intermediate wheatgrass (IWG) and grazed either season long (SLG) or grazed-defer-graze (GDG).

Period	MIX		IWG		SEM ¹	P>F
	SLG	GDG	SLG	GDG		
----- Animal days ha ⁻¹ -----						
Year 1						
I	462	465	474	482	27	0.954
II ²	435a ³	—	386b	—	35	0.006
III	151a	173a	153a	237b	20	0.098
Total	1048a	638b	1013a	719b	47	0.007
Year 2						
I	428a	509b	410a	509b	10	0.004
II ²	294a	—	205b	—	12	0.033
III	190a	190a	114b	227c	7	0.001
Total	912a	699b	729b	736b	17	0.003
Year 3						
I	437a	502b	400c	509b	7	0.001
II ²	329	—	319	—	25	0.803
III	208ac	254ac	190b	269c	15	0.071
Total	974a	756b	909c	778b	17	0.003

¹Standard error of the mean.

²Statistical analysis includes only ADH for SLG.

³Row means with different letters differ ($P < 0.05$).

Table 4. Weight gain (kg ha⁻¹) from pastures seeded to a mixture of 4 species (MIX) or intermediate wheatgrass (IWG) and grazed either season long (SLG) or graze-defer-graze (GDG) for 3 years.

Period	Pasture type			Management strategy			SEM ¹
	MIX	IWG	P>F	SLG	GDG	P>F	
	kg/ha ⁻¹						
Year 1							
I	475	554	0.042	542	486	0.100	17
II ²	352	253	0.101	(274)	—	—	16
III	112	137	0.175	119	130	0.471	10
Total	939	944	0.168	935	616	0.003	31
Year 2							
I	196	245	0.269	223	218	0.908	24
II ²	282	168	0.045	(227)	—	—	16
III	112	63	0.022	95	80	0.303	8
Total	590	476	0.280	545	298	0.006	28
Year 3							
I	230	261	0.129	233	258	0.233	10
II ²	301	211	0.045	(257)	—	—	18
III	185	137	0.010	166	156	0.432	7
Total	716	609	0.001	656	414	0.001	5

¹Standard error of the mean.

²Statistical analysis included only weight gain of steers on SLG. Values under management strategy are for information only.

treatment than the other combinations. No differences were observed among SLG-IWG and the 2 pasture types that were deferred. Pasture productivity would be expected to be higher for the SLG steers since they grazed an extra 56 days during period II when the steers on the GDG management strategy were removed from the study and grazed on subirrigated meadows.

For year 1, there were no differences in total ADH between pasture types when grazed under either grazing treatment. Since the first year of grazing was on stands that were established the previous fall, this probably does not reflect potential differences between pasture types. However, by years 2 and 3, MIX pastures produced 183 and 65 more ADH, respectively, than IWG when grazed season-long. Under GDG, there were no significant differences between pasture types for either year 2 or 3.

When ADH for SLG for period II was subtracted from the total to equalize time on pasture, the total grazing capacity for GDG-MIX was increased over SLG-MIX by 4, 13, and 17% for years 1, 2, and 3, respectively. The GDG strategy with IWG increased the total grazing capacity over SLG by 15, 41, and 32% over the same year sequence.

In general, these data suggested that MIX was more productive than IWG under SLG, but there was no difference under GDG. The GDG management strategy increased the grazing capacity for both MIX and IWG when ADH were equalized.

Animal Gains/Hectare

The animal gains ha⁻¹ (AGH) shown in Table 4 are a product of ADG and ADH and thus reflect in one value the total pasture productivity. Gains of steers while on subirrigated meadow pasture (period II) were not included in total pasture production values. Year by treatment interaction ($P<0.03$) indicated that animal gains could be expected to be variable among years depending on grazing strategy and/or pasture stand. There was no pasture type by management strategy interaction ($P<0.05$).

Although significant only for the first period in year 1, IWG tended to produce more AGH than MIX during the early portion of the growing season for all years (Table 4). However, for periods II and III, AGH from MIX exceeded gain from IWG by 114 and 49 kg ha⁻¹ for year 2, and by 90 and 48 kg ha⁻¹ for year 3, respectively. The seasonal distribution pattern of AGH as affected by pasture type was similar to trends indicated for both animal performance and pasture production which was previously discussed. This would be expected since AGH is a produce of these values.

The significant increase in total AGH for the SLG management strategy resulted from the gain realized during Period II when the steers were removed from the GDG pastures and placed on subirrigated meadow (Table 4). With the exception of Period I, year 1, there were no differences between grazing strategies for Periods I and III. Discounting the gain from the SLG pastures during Period II resulted in no effect of grazing strategy on AGH when considered over the grazing season.

Pasture Persistence

Visual observations indicated that weedy species were becoming a severe problem in the IWG pastures after the first grazing season. Yellow foxtail (*Setaria lutescens* Weig.), green foxtail (*Setaria viridis* L.), and redroot pigweed (*Amaranthus retroflexus* L.) increased each year of the study, becoming most abundant during year 3. In contrast, MIX pastures maintained excellent grass stands throughout the study resisting invasion of weedy species and persisting equally well under both grazing management strategies.

Discussion and Conclusions

Carrying capacity and animal gain were highest for the SLG grazing strategy, primarily due to the extra 56 days of grazing, while the GDG steers were off the test pastures on subirrigated meadow forage. Fertilizer and irrigation water were reduced by 38 and 34%, respectively, for the GDG pastures during this period, but the total carrying capacity was reduced only 16% over the 3-year study. These data indicate that the season-long production efficiency of irrigated pasture could be improved by removing steers from irrigated pasture and eliminating irrigation and fertilization during mid-summer when the productivity of cool-season grasses declines. The GDG practice fits within the concept of using alternative forage resources during different segments of the growing season when each are most productive, which was described by McIlvain and Shoop (1973) as a "complementary forage system."

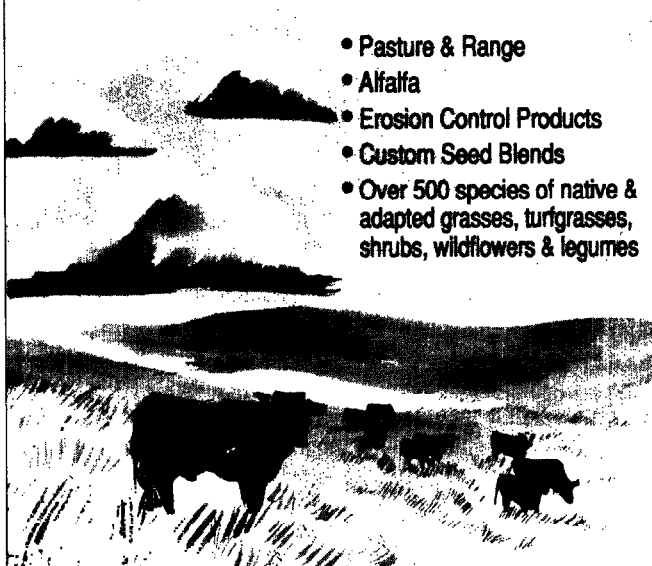
A comparison of MIX and IWG pasture stands in relation to grazing strategy indicated that animal performance was comparable overall, but that steer gains on IWG were generally higher early in the growing season, whereas MIX pastures produced better gains late in the grazing season. Persistence of stands was better for the MIX pastures compared to IWG which became invaded by weedy species. Both ADG and AGH were higher for MIX than

IWG. Under the conditions of this study, the MIX pastures were better adapted to either grazing strategy than the IWG pastures.

Literature Cited

- Agricultural Economics Staff.** 1991. Estimated crop and livestock production costs. Univ. Nebraska Coop. Ext. Ser. Pub. EC91-872. Lincoln, Neb. p. B13-14.
- Allen, H.R.** 1972. Grassland production systems compared with grain production. South Dakota State Univ. Agr. Exp. Sta. Bull. 600. Brookings.
- Anderson, W.E., and J.J. Jernstedt.** 1971. Evaluating multiple economic effects of forage development and management. *J. Range Manage.* 24:174-180.
- Ford, M.J., D.C. Clanton, and M.E. England.** 1986. Computer simulation—retained ownership profitable. Univ. Nebraska Agr. Exp. Sta. Beef Cattle Rep. MP-50, Lincoln, Neb. p. 6-9.
- Hart, R.E., J.W. Waggoner, Jr., T.G. Dunn, C.C. Kaltenbach, and L.D. Adams.** 1988. Optional stocking rate for cow-calf enterprises on native range and complementary improved pastures. *J. Range Manage.* 41:435-440.
- Lodge, R.W.** 1963. Complementary grazing systems for Sandhills of the Northern Great Plains. *J. Range Manage.* 16:240-244.
- Matches, A.G.** 1970. Pasture research methods. Proc. Nat. Conf. Forage Qual. Eval. Util., Univ. Nebraska. Lincoln, Neb. Sect. I, p. 1-32.
- McIlvain, E.H., and M.C. Shoop.** 1973. Use of farmed and tame pasture to complement native range. Proc. Great Plains Beef Symposium, Lincoln, Neb. p. L, 1-19.
- Moore, R.A.** 1970. Symposium on pasture methods for maximum production: pasture systems for a cow-calf operation. *J. Anim. Sci.* 30:133-137.
- Mott, G.O.** 1959. Intersociety forage evaluation symposium: IV. Animal variation and measurement of forage quality. *Agron. J.* 51:223-226.
- Nichols, J.T., M.J. Moore, L.F. Hoatson, and D.C. Clanton.** 1976. Grasses for irrigated pasture. Univ. Nebraska Beef Cattle Rep. EC76-218. Lincoln. p. 12-13.
- Nichols, J.T.** 1989. Range, plus complementary forages for beef cattle production p. 196-203. In: Proc. Amer. Forage and Grassland Congr., Univ. Guelph, Ontario, Canada.
- Nichols, J.T., and D.C. Clanton.** 1985. Irrigated pastures, p. 507-516. In: M.E. Heath, R.F. Barnes, D.S. Metcalf (eds.) *Forages, the Science of Grassland Agriculture*, 4th ed. Iowa State Univ., Press, Ames.
- Nichols, J.T., and M.J. Moore.** 1977. Beef production and water use efficiency of eight irrigated pasture grasses. Proc. Irrigation Short Course, Jan. 24-25, 1977. Univ. Nebraska, Lincoln, Neb. p. 112-116.
- SAS.** 1985. SAS users guide: Statistics. SAS Inst. Inc., Cary, N.C.
- Smoliak, S.** 1968. Grazing studies on native range, crested wheatgrass and Russian wildrye pastures. *J. Range Manage.* 21:47-50.

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Potential forage species for deer in the southern mixed prairie

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Abstract

Improving wildlife habitat through the introduction of nutritious forage species is a management tool that may be used to increase target populations such as deer. By increasing deer numbers the potential of leasing hunting rights on private land is improved. Our objectives were to evaluate and compare establishment and production of 2 browse species and to determine the production and nutritional quality among 6 forb species in rangeland conditions. Browse species were littleleaf lead-tree (*Leucaena retusa* Gray) and four-wing saltbush (*Atriplex canescens* (Pursh) Nutt.). Forb species were 'Cody' alfalfa (*Medicago sativa* L.), 'Rennumex' sainfoin (*Onobrychis viciifolia* Scop.), 'Howard' subterranean clover (*Trifolium subterraneum* L.), 'Nungarin' subterranean clover (*T. subterraneum* L.), 'Eldorado' Englemann daisy (*Engelmannia pinnatifida* Nutt.), 'Sabine' Illinois bundleflower (*Desmanthus illinoensis* (Michx.) MacM.), and 'Plateau' awnless bush sunflower (*Simsia calva* (Engelm. & Gray) Gray). Establishment for littleleaf lead-tree was 21%. Successful four-wing saltbush plants produced a greater volume (m³) of plant material than littleleaf lead-tree. Littleleaf lead-tree leaf material contained crude protein (CP) values from 11.6 to 16.9%. Of the cool-season forbs, alfalfa, and sainfoin produced the most above-ground phytomass at 23.6 and 22.6 g/m of planted row, respectively. The subclovers produced intermediate amounts of phytomass while Englemann daisy produced negligible amounts at phytomass of 4.2 g/m of planted row. Warm-season forages, awnless bush sunflower, and Illinois bundleflower produced an average of 115.1 and 120.2 g/m of planted row, respectively. Seasonal CP means were greatest for alfalfa (16.5%) and awnless bush sunflower (16.1%). Our results suggest that littleleaf lead-tree, awnless bush sunflower, and Illinois bundleflower would supply adequate supplemental forage from summer to fall; subterranean clovers, alfalfa, and sainfoin would provide forage from winter through spring. Four-wing saltbush could provide forage year-round.

Key Words: browse, crude protein, forbs, habitat management, *Odocoileus* spp., phytomass

Increasing forage quality and quantity is a worthy management goal for increasing deer populations. Landowners are generally interested in such management due to the economic return for recreational opportunities on their rangeland (Steinbach and Ramsey 1988). At present, white-tailed deer (*Odocoileus virginianus*) are the single-most hunted big game species in Texas and generate the greatest economic return (Smith 1986).

Both domestic agricultural and native plant species have been planted as supplemental food plots to increase deer numbers (Herbel 1986, Vanderhoof and Jacobson 1989). White-tailed deer select primarily the succulent, new green growth of browse, forbs, and grasses, with browse and forbs being most preferred (Bryant et al. 1979, Sowell et al. 1985). Thus, these forage types should be emphasized when establishing food plots for deer. Periods of nutritional stress experienced by deer in the southern mixed prairie coincide with late summer to winter when forage quality is low because forbs are scarce and grasses are mature and of low nutritional value. In addition, periodic summer droughts may occur during the nutritionally demanding periods of lactation, antler growth, and fawn growth, increasing the nutritional problems facing deer populations. Introduction of evergreen and drought-tolerant shrubs could provide protein, phosphorus, and energy particularly during the winter stress periods (Cook 1972, Dietz 1972, Stidham et al. 1982). Forbs and legumes could supplement deer diets from spring through autumn.

Our study attempted to assess the feasibility of introducing potentially beneficial forages to the southern mixed prairie region of Texas, a region of low deer densities and insufficient forage (Cook and Harwell 1978, Wiggers et al. 1984, Bryant and Morrison 1985). The objective was to determine phytomass production and nutritional quality among 2 browse and 6 forb species in rangeland conditions.

Study Area

The study was conducted on a 7,300-ha private ranch in Foard County, about 8 km west of Crowell, Tex. The ranch is located in the southern mixed prairie, which is characterized by rolling to moderately rough terrain (Cook and Harwell 1978) with elevations of 427 to 520 m (Koos and Dixon 1964). Although the average annual precipitation is 61 cm, there is considerable annual variation. Rainfall primarily occurs as localized, convective thunderstorms from May to October. Winters are mild, although abrupt temperature changes may occur as a result of cold fronts. Summers are hot with daily maximum temperatures often above 38° C. High temperatures are generally accompanied by low humidity and light winds. The average growing season lasts about 220 days (1 April to 10 November).

Study plots were located on level rangeland of the Tillman-Vernon soil association. Although soil nutrient analyses detected differences among soil nutrients on study plots, no soil nutrients were limiting to plant performance. Soil texture ranged from clay to clay loam. Native vegetation included tall and mid-grasses with an overstory of honey mesquite (*Prosopis glandulosa* Torr. var. *glandulosa*), red-berry juniper (*Juniperus pinchotii* Sudw.), and lotebush (*Ziziphus obtusifolia* (T. & G.) Gray) [scientific and common names according to Correll and Johnston (1979)]. All livestock had been removed from the ranch 3 years before initia-

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Table 1. Establishment means¹ (%) and standard errors (in parentheses) for four-wing saltbush (*Atriplex canescens* (Pursh) Nutt.) planted 19–20 Sept. and 13–15 Oct. 1986 in Foard County, Texas.

	Evaluation-month (1987)											
	June		July		Aug.		Sept.		Oct.		Nov.	
Planting date	Sept.	Oct.	Sept.	Oct.	Sept.	Oct.	Sept.	Oct.	Sept.	Oct.	Sept.	Oct.
Means (%)	0.3	0.3	0.4	0.3	0.3	0.2	0.3	0.2	0.3	0.2	0.3	0.2
Standard (%) Errors	(0.1)	(0.0)	(0.2)	(0.0)	(0.2)	(0.0)	(0.2)	(0.0)	(0.2)	(0.0)	(0.2)	(0.0)

¹N = 4 samples per mean.

tion of this study.

Methods

Browse Species

Two browse species were selected for planting based on their potential nutritive value during winter or mid- to late-summer, palatability, and potential adaptability to the study area (Vines 1960, USDA For. Serv. 1974, Correll and Johnston 1979).

Four, 2-ha plots were selected with similar topography and soil type. Plots were cleared with a bulldozer, deep-plowed, and disced. A potential winter forage species, four-wing saltbush (*Atriplex canescens* (Pursh) Nutt.), was seeded on 19–20 September 1986 at 2 plots, and 13–15 October 1986 at the remaining 2 plots. On each plot, this species was randomly assigned to 3 rows. Our manual seeding method simulated broadcast seeding, followed by light harrowing to insure seed-soil contact, on a well-disced seedbed. Plots were not fenced to exclude ungulates because no livestock were on the ranch, and deer densities, determined by helicopter surveys (fall 1986, fall 1987), were extremely low (1 deer per 265 ha). Fencing for small herbivores (e.g., lagomorphs) was cost prohibitive.

On each plot, the number of live seedlings was counted within 15-m randomly selected transects on 2 of 3 rows. Using the known seeding rate, percent establishment was calculated monthly from June to November 1987. The volume (m³) of four-wing saltbush was determined 21 months after seeding according to Bryant and Kothmann (1979).

Littleleaf lead-tree (*Leucaena retusa* Gray), the summer forage shrub, was seeded in early May 1987 on 2 plots in the same manner as four-wing saltbush. However, seeds were scarified by immersing in boiling water for 1 min, rinsing with cool water, drying overnight at room temperature, then inoculating with bacteria (*Rhizobium* spp.) before planting. Percent establishment of lead-tree seedlings was determined for 5 evaluation-months (July to November 1987). Transect length was reduced to 0.5-m because of high seedling density.

Establishment data of browse species over time were analyzed using analysis of variance (ANOVA) in a completely randomized design, split-split plot arrangement. Species' means were separated using Fisher's Least Significant Difference (FLSD) procedure (Steele and Torrie 1980:176). The level of significance was 0.05 in this and all subsequent statistical tests.

The abundant number of seedlings produced by littleleaf lead-tree allowed six, 0.5-m segments from each plot to be clipped in October 1987. Clipped forage was dried in a forced-air oven at 60° C to a constant weight. Leaves were separated from stems and leaves were weighed to the nearest 0.01 g. Lead-tree leaves were ground in a Thomas-Wiley laboratory mill to pass a 1-mm mesh screen and were analyzed for nitrogen (%) using the micro-Kjeldahl technique (AOAC 1984). Leaf dry matter (%) and organic matter (%) were determined for reporting leaf crude protein (CP) (% N ÷ 6.25) on a dry organic matter basis. Leaf weight and CP data from each plot were analyzed using ANOVA in a completely randomized design to determine if there were plot differences in

production and nutritional quality. Means were separated using FLSD procedure.

Forb Species

Four cool-season and 2 warm-season forbs were selected for evaluation based on their potential adaptability, palatability to herbivores, perennial or self-seeding characteristics, and nutrient quality (Vines 1960, Leffel 1973, Correll and Johnston 1979). Cool-season species, including 4 legumes, 'Cody' alfalfa (*Medicago sativa* L.), 'Renumex' sainfoin (*Onobrychis viciifolia* Scop.), 'Howard' subterranean clover (*Trifolium subterraneum* L.), 'Nungarin' subterranean clover (*T. subterraneum* L.), and 1 composite, 'Eldorado' Engelmann daisy (*Engelmannia pinnatifida* Nutt.), were seeded on 19–20 September 1986 at 2 plots, and on 13–15 October 1986 at the 2 remaining plots. Warm-season species, including 1 legume, 'Sabine' Illinois bundleflower (*Desmanthus illinoensis* (Michx.) MacM.), and 1 composite, 'Plateau' awnless bush sunflower (*Simsia calva* (Engelm. & Gray) Gray), were seeded on 16–17 May 1987 at 2 plots.

Plot design for forb species was the same as that for browse species. Cool-season forbs were seeded on 15 rows (5 species × 3 replications per species) adjacent to browse species. Warm-season forbs were seeded (2 species × 3 replications per species) on 16–17 May 1987 at 2 plots adjacent to fall browse and forb plantings. All legumes were inoculated with *Rhizobium* spp. before planting. Phytomass of each cool-season species was determined on each plot by clipping 2 randomly selected 0.5-m segments from 2 of the species' 3 rows for 9 evaluation-months (December, April to November). Phytomass of warm-season species was determined using the same clipping method for 5 evaluation-months (July to November). No 0.5-m segment was clipped more than once. Clipped forage was dried in a forced-air oven at 60° C to a constant weight then weighed to the nearest 0.01 g. To determine production differences among species over time, cool-season forb phytomass data were analyzed as described for winter browse establishment data, and phytomass data of warm-season forbs were analyzed using an ANOVA in a randomized block design, split plot arrangement.

Dried forage of each forb species was analyzed for percent CP as described for lead-tree leaves. Percent CP of cool-season and warm-season forbs was analyzed statistically in the same manner as for browse.

Results

Browse Species

Four-wing saltbush maintained a fairly constant, but low, percent establishment through all evaluation-months (Table 1). The average volume per four-wing saltbush seedling was 2.08 m³, 21 months after planting.

Littleleaf lead-tree, the warm-season woody legume, had a mean establishment of 21%. Five months after planting (October 1987), all rows of littleleaf lead-tree seedlings produced abundant leafy material (\bar{x} = 55 g/m) and percent CP of leaf material varied from

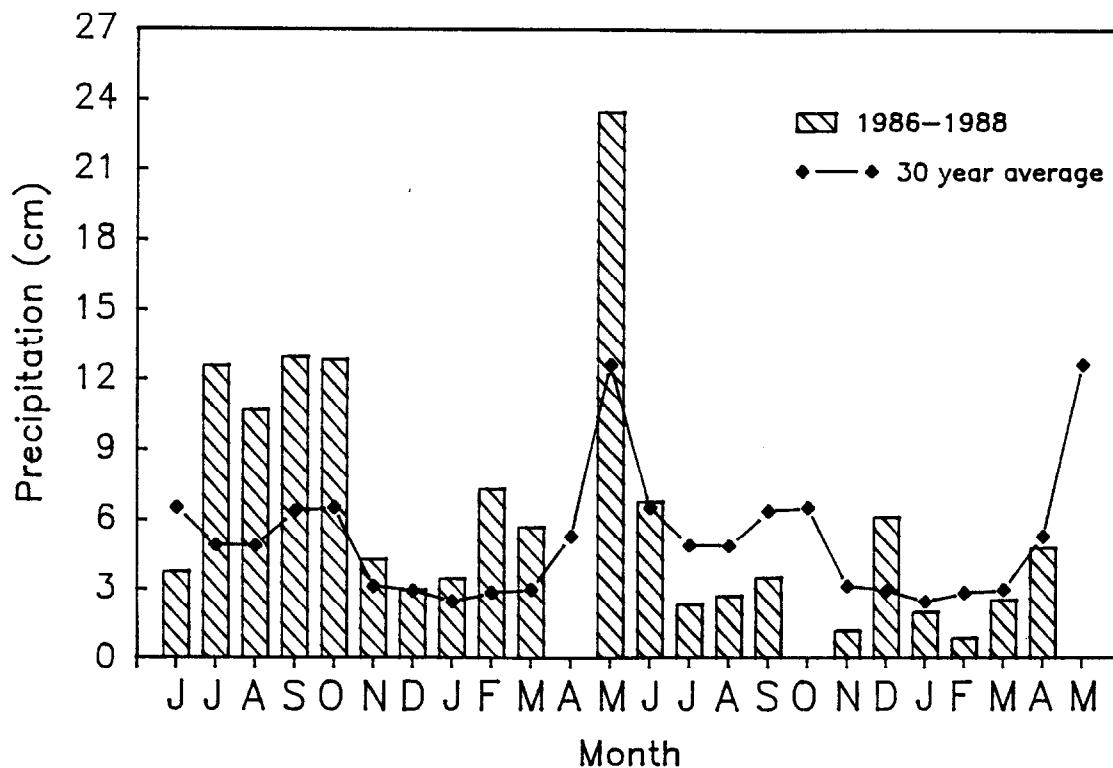


Fig. 1. Monthly and 30-year average precipitation (cm) recorded in Crowell, Texas, approximately 8 km west of the study site.

16.9% on plot 2 to 11.6% on plot 1.

Forb Species

Howard and Nungarin subterranean clovers were the only cool-season species that had germinated and produced enough above-ground vegetation to clip in December 1986 (Table 2). Although by April 1987 alfalfa and sainfoin had produced above-ground vegetation, phytomass values of the subterranean clovers were still greater ($P < 0.05$). Howard and Nungarin subterranean clovers produced seed then became senescent in May 1987. From June to November 1987, alfalfa and sainfoin had the greatest ($P < 0.05$) phytomass values of the cool-season species. Engelmann daisy did not produce significant phytomass in any evaluation-month. Seeds of the subterranean clovers were expected to germinate by November 1987. However, the preceding summer and fall were

very dry (Fig. 1) and no regeneration occurred.

Phytomass of warm-season species, Illinois bundleflower and awnless bush sunflower (Table 3), exceeded those of the cool-season species from August to November 1987 (Table 2). By November 1987, leaves had become senescent and production declined. These warm-season species produced forage by July and continued to produce forage through October 1987, regardless of dry conditions.

Planting date did not affect cool-season forb CP concentrations ($P > 0.05$). In some sampling periods, 1 or more of the cool-season forbs did not produce enough phytomass to permit CP analysis (Table 4). Nungarin and Howard subterranean clovers had the greatest average CP concentrations, followed by alfalfa and sainfoin. Crude protein values of alfalfa, sainfoin, and Engelmann daisy declined from July to November 1987 as they matured. An

Table 2. Monthly mean phytomass^{1,2,3} (g/m of row) and standard errors (in parentheses) of cool season forb species planted 19–20 Sept. and 13–15 Oct. 1986 in Foard County, Texas.

Species	Monthly phytomass									Species mean
	Dec. 86	April 87	May 87	June 87	July 87	Aug. 87	Sept. 87	Oct. 87	Nov. 87	
	(g/m of row)									
Alfalfa	—	19.2abc (2.0)	11.7a (1.6)	41.1a (6.0)	51.8a (7.1)	25.3a (4.1)	24.1a (4.6)	6.2a (2.4)	9.4a (1.8)	23.6 (5.6)
Engelmann daisy	—	0.0a (0.0)	4.0a (1.2)	8.5b (3.0)	9.4b (2.4)	4.0a (1.6)	5.2a (2.3)	0.5a (0.4)	2.0a (0.8)	4.2 (1.2)
Sainfoin	—	17.3ab (1.8)	21.8a (3.9)	34.8a (3.8)	45.5a (11.6)	27.0a (5.2)	14.0a (2.8)	10.0a (3.2)	10.3a (2.3)	22.6 (4.4)
Howard subclover	22.4a (5.9)	59.4d (11.0)	0.0a (0.0)	0.2b (0.1)	0.7b (0.3)	—	—	—	—	16.5 (11.5)
Nungarin subclover	21.8a (7.3)	39.4bc (5.9)	0.0a (0.0)	0.0b (0.0)	0.0b (0.0)	—	—	—	—	12.2 (8.0)

¹Means followed by different letters within months are different ($P < 0.05$).

²N = 16 samples per mean.

³Dash (—) indicates no phytomass available to measure.

Table 3. Monthly mean phytomass^{1,2} (g/m of planted row) and crude protein (%CP) (standard errors in parentheses) of warm season forb species planted 16–17 May 1987 in Foard County, Texas.

Species	Evaluation month										Species mean	
	July		Aug.		Sept.		Oct.		Nov.			
	(g/m)	(%CP)	(g/m)	(%CP)	(g/m)	(%CP)	(g/m)	(%CP)	(g/m)	(%CP)	(g/m)	(%CP)
Awnless bush sunflower	21.4 (7.6)	27.7 (0.5)	112.1 (33.3)	14.7 (0.2)	146.0 (35.9)	16.7 (1.4)	191.2 (38.9)	13.4 (0.7)	105.0 (30.9)	8.0 (0.6)	115.1 (28.0)	16.1 (3.2)
Illinois bundleflower	41.9 (6.9)	21.1 (1.1)	225.8 (23.7)	13.2 (0.2)	150.7 (24.6)	10.9 (0.8)	127.4 (26.0)	7.6 (0.4)	55.3 (9.5)	4.3 (0.1)	120.2 (33.5)	11.4 (2.8)

¹No differences ($P>0.05$) were found among species means or evaluation-month means, and species \times evaluation-month interaction was not significant ($P>0.05$).

²N = 8 samples per biomass mean, N = 4 samples per %CP mean.

exception was Engelmann daisy, which produced some new growth of higher CP values in October and November 1987. Although no differences were detected among warm-season forb species' CP means ($P>0.05$) (Table 3), their CP values were numerically greater than those of the cool-season forb species from July to September 1987. The CP values of awnless brush sunflower and Illinois bundleflower declined as they matured in October and November 1987, as did the CP values of the cool-season forb species.

Discussion

Browse Species

Four-wing saltbush demonstrated potential for winter browse species establishment. Drier than average environmental conditions during this study (Fig. 1) may have impacted establishment. Environmental conditions can not be controlled when conducting a rangeland study, hence they may have a strong impact on any final outcome.

Although percent establishment of four-wing saltbush was low, seedlings were able to produce a significant amount of above-ground phytomass as measured by volume. This shrub demonstrated potential as a winter forage browse species. Thus, four-wing may be considered as a supplemental winter forage species in the southern mixed prairie. Although we did not have enough four-wing plant material available to analyze for CP content, other studies in Texas have shown that percent CP and digestible organic matter in four-wing saltbush leaves exceed the minimum requirements of cattle, sheep, goats, and deer throughout the year (Peter-

son et al. 1987). In addition, four-wing saltbush was found to be a good potential source of fall and winter nutrients in the Oklahoma tallgrass prairie (Stidham et al. 1982).

The warm-season browse species tested, littleleaf lead-tree, established successfully despite dry conditions. In addition to the high percent establishment of littleleaf lead-tree, its leaves were determined to be high in CP, a good indication of their nutritive quality. Whisenant et al. (1985) found that littleleaf lead-tree CP ranged from 15% to 34% (\bar{x} = 22%) from May to November with highest values in the spring and autumn. The variation in leaf CP found between plots in this study may have been due to seedling density. On plot 2 seedlings were half as dense, thus less crowded.

Littleleaf lead-tree seedlings were the only shrubs consumed by lagomorphs during this study. Several seedlings were eaten to ground level, yet resprouted rapidly. These seedlings appeared to be very hardy. No shrub seedlings were affected by grasshoppers.

The hot, dry summer season in central and south Texas is often critically deficient in available nutritious forage (Teer et al. 1965, Bryant et al. 1980, Whisenant et al. 1985). In addition, late-summer and fall herbaceous forage quality declines with maturity (Moen 1973:309). In this study, littleleaf lead-tree produced available leaf material high in CP from July to November. The amount of CP in littleleaf lead-tree leaves sampled in October 1987 was sufficient to support most physiological stages of deer (Verme and Ullrey 1984:117). Based on percent establishment, forage production, and nutritional quality, littleleaf lead-tree appeared to have excellent potential as a spring-through-fall forage for deer in the southern mixed prairie.

Table 4. Monthly mean crude protein^{1,2} (%) and standard errors (in parentheses) of cool season forb species planted 19–20 Sept. and 13–15 Oct. 1986 in Foard County, Texas.

Species	Monthly crude protein									Species mean
	Dec. 86	April 87	May 87	June 87	July 87	Aug. 87	Sept. 87	Oct. 87	Nov. 87	
						(%)				
Alfalfa	— ³	26.2a (0.7)	18.2a (0.2)	22.0a (0.1)	17.4b (0.6)	12.5a (0.5)	12.6a (0.5)	13.1b ⁶ (0.8)	9.8a (0.3)	16.5 (2.0)
Engelmann daisy	—	—	16.1ab (0.5)	18.5bc (1.1)	14.6c (0.6)	9.3a (0.8)	9.8a (0.3)	18.1a ⁵ (0.4)	13.5a ⁴ (2.6)	14.3 (1.4)
Sainfoin	—	13.9b (0.6)	12.2b (0.6)	16.4c (1.0)	12.9c (0.6)	9.6a (0.6)	9.7a (0.8)	11.0b (0.4)	9.2a (0.5)	11.9 (0.9)
Howard subclover	20.1a ⁴ (2.9)	20.5ac (0.8)	—	20.7ab ⁵ (0.6)	20.0a ⁴ (0.9)	—	—	—	—	20.3 (0.2)
Nungarin subclover	23.1a ⁴ (1.6)	18.4bc (0.6)	—	—	—	—	—	—	—	20.8 (2.4)

¹Means followed by different letters within columns are different ($P<0.05$).

²N = 8 samples per mean unless otherwise indicated.

³Sample too small to analyze

⁴N = 4 samples per mean.

⁵N = 2 samples per mean.

⁶N = 6 samples per mean.

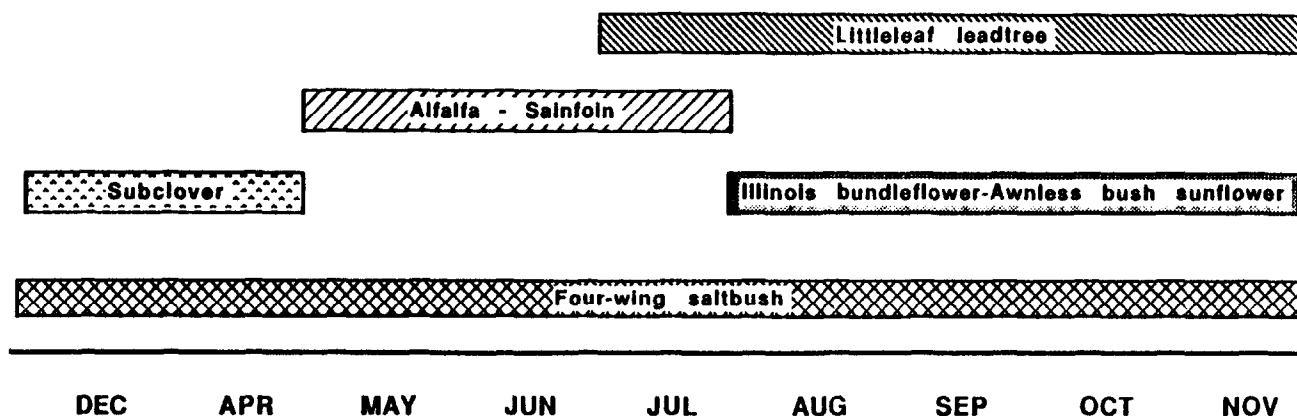


Fig. 2. Peak crude protein and biomass production periods of successfully introduced browse and forb species in the southern mixed prairie of Texas.

Forb Species

Subterranean clover varieties had good phytomass production in the first cool season (December 1986–April 1987); however, there was much less precipitation in the second cool season (December 1987–April 1988) (Fig. 1) and the subterranean clovers failed to reseed, and thus produced no measurable phytomass. Subterranean clovers are adapted to climates with relatively warm, moist winter and dry summers (Smith et al. 1986, Holt and Haferkamp 1987). The southern mixed prairie is characterized by annual fluctuations in the amount of precipitation received. Thus, Howard and Nungarin subterranean clovers may not be dependable winter forage species in this region during dry years.

Alfalfa and sainfoin were productive through July 1987. Engelmann daisy consistently produced less forage than the other cool-season species. The decline in phytomass of alfalfa and sainfoin from July 1987 to November 1987 coincided with the lack of precipitation and low observed soil moisture. In addition, consumption by grasshoppers and lagomorphs contributed to the decline in phytomass of alfalfa and sainfoin.

Illinois bundleflower and awnless bush sunflower were highly productive under the conditions of this study. These species appear to have good potential as summer through autumn forage.

The variety of forbs planted in this study provided forage at different phenological stages, or maturity, through all seasons, thus providing digestible material for wildlife throughout the year. Forage species in an immature phenological stage are more digestible and contain more protein than those in mature stages (Moen 1973:309). Howard and Nungarin subterranean clovers provided forage from December to April when adequate precipitation was available. Alfalfa and sainfoin provided forage from May to July, and Illinois bundleflower and awnless bush sunflower provided forage from August to November. All species planted were perennials or self-seeding annuals and possibly could persist for several years if precipitation was adequate.

Forbs introduced on the study sites provided CP concentrations within a 13 to 20% range through all months of the study, well above the minimum level of 6 to 7% required for deer to survive (Dietz 1965). The introduced forbs provided CP concentrations adequate for deer reproduction, growth, and antler development (French et al. 1956, Dietz 1965, Verme and Ullrey 1984, Sowell et al. 1985, Johnson et al. 1987).

Conclusions

The introduction of a warm-season shrub and warm-season forbs, and to a lesser degree, cool-season forbs, was relatively successful even under drought conditions. Littleleaf lead-tree, Illi-

nois bundleflower, and awnless bush sunflower demonstrated high percent establishment, phytomass, and nutritive qualities. Alfalfa, sainfoin, and the subterranean clovers also performed well, although the clovers did not successfully re-seed during this study, perhaps due to inadequate fall and winter precipitation. Maintaining forage species that peak in phytomass production and CP levels at different months throughout the year, as illustrated in Figure 2, would provide reasonable good quantity and high quality forage at all times and offer potential to enhance deer nutrition and thus, deer numbers, in the southern mixed prairie.

The findings of this study were collected over 2 growing seasons in the southern mixed prairie region of Texas. Although it would have been optimal to replicate the entire planting scheme the second year and to collect data for a longer period of time, this scenario was not logistically feasible. Future studies should pursue additional plantings to allow determination of response of these and other forage species to additional environmental factors. Our results do, however, allow some preliminary recommendations. To provide a desirable forage combination from summer to fall, we recommend a forage mixture of 30% awnless bush sunflower, 40% Illinois bundleflower, and 30% littleleaf lead-tree. A mixture of 20% subterranean clovers, 40% alfalfa, and 40% sainfoin should provide desirable forage and four-wing saltbush desirable browse during winter and spring. Cool-season agronomic grasses are eaten by deer in this region (Sowell et al. 1985) and may be considered as part of a planting strategy to supply winter nutrition (Wiggers et al. 1984). Because the browse and forb species we tested are perennials or self-seeding annuals, annual seasonal revegetation may not be required as it is for cereal grains. Introducing forage species to an area lacking certain seasonal nutritive components has the potential to improve the habitat for wildlife species. When there is an economic benefit derived from increasing the numbers of a particular species, the costs of introducing forage species may be defrayed. Several of the forage species evaluated in our study demonstrated good potential in the Texas southern mixed prairie.

Literature Cited

- AOAC 1984. Official methods of analysis. Thirteenth ed. Assoc. Off. Anal. Chemists, Washington, D.C.
- Bryant, F.C., and M.M. Kothmann. 1979. Variability in predicting edible browse from crown volume. *J. Range Manage.* 32:144–146.
- Bryant, F.C., M.M. Kothmann, and L.B. Merrill. 1979. Diets of sheep, Angora goats, Spanish goats, and white-tailed deer under excellent range conditions. *J. Range Manage.* 32:412–417.

- Bryant, F.C., and M.M. Kothmann, and L.B. Merrill. 1980.** Nutritive content of sheep, goat, and white-tailed deer diets on excellent condition rangeland in Texas. *J. Range Manage.* 33:410-414.
- Bryant, F.C., and B. Morrison. 1985.** Managing plains mule deer in Texas and Eastern New Mexico. Manage. Note 7. Range and Wildl. Manage., Coll. Agr. Sci. Texas Tech Univ., Lubbock.
- Cook, C.W. 1972.** Comparative nutritive values of forbs, grasses, and shrubs, p. 303-310. *In:* C.M. McKell (ed.), *Wildland shrubs—their biology and utilization*. U.S. Forest Serv. Gen. Tech. Rep. INT-1.
- Cook, R.L., and W.F. Harwell. 1978.** Big game investigations. Job 1. White-tailed deer population trends. Job Perf. Rep. Tex. Fed. Aid Proj. W-109-R-1.
- Correll, D.S., and M.C. Johnston. 1979.** Manual of the vascular plants of Texas. Univ. of Texas Print. Div., Austin.
- Dietz, D.R. 1965.** Deer nutrition research in range management. Trans. Amer. Wildl. and Natur. Resour. Conf. 30:274-285.
- Dietz, D.R. 1972.** Nutritive value of shrubs, p. 289-302. *In:* C.M. McKell (ed.), *Wildland shrubs—their biology and utilization*. U.S. Forest Serv. Gen. Tech. Rep. INT-1.
- French, C.E., L.C. McEwen, N.D. Magruder, R.H. Ingram, and R.W. Swift. 1956.** Nutrient requirements for growth and antler development in the white-tailed deer. *J. Wildl. Manage.* 20:221-232.
- Herbel, C.H. 1986.** Seeding shrubs in the field. *Reclam. and Reveg. Res.* 5:377-385.
- Holt, E.C., and M.R. Haferkamp. 1987.** Growth of introduced temperate legumes in the Edwards Plateau and South Texas Plains. *J. Range Manage.* 40:132-135.
- Johnson, M.K., B.W. Delany, S.P. Lynch, J.A. Zeno, S.R. Schultz, T.W. Keegan, and B.D. Nelson. 1987.** Effects of cool-season agronomic forages on white-tailed deer. *Wildl. Soc. Bull.* 15:330-339.
- Koos, W.M., and M.L. Dixon. 1964.** Soil survey of Foard County, Texas. U.S. Soil Conserv. Serv. Tex. Agr. Exp. Sta. Serv. 1961, 14.
- Leffel, R.C. 1973.** Other legumes, p. 208-220. *In:* M.E. Heath, D.S. Metcalf, and R.F. Barnes (eds.), *Forages: the science of grassland agriculture*. Third ed. Iowa State Univ. Press, Ames.
- Moen, A.N. 1973.** *Wildlife ecology: An analytical approach*. W.H. Freeman and Co., San Francisco, Calif.
- Petersen, J.L., D.N. Ueckert, R.L. Potter, and J.E. Huston. 1987.** Ecotypic variation in selected four-wing saltbush populations in western Texas. *J. Range Manage.* 40:361-366.
- Smith, G.R., R.D. Randel, and C. Bradshaw. 1986.** Influence of harvest date, cultivar, and sample storage method on concentration of isoflavones in subterranean clover. *Crop Sci.* 26:1013-1016.
- Smith, M. 1986.** *Wildlife in Texas: Probable and profitable*. The Cattleman 72:88-102.
- Sowell, B.F., B.H. Koerth, and F.C. Bryant. 1985.** Seasonal nutrient estimates of mule deer diets in the Texas Panhandle. *J. Range Manage.* 38:163-167.
- Steele, R.D.G., and J.H. Torrie. 1980.** *Principles and procedures of statistics*. Second ed. McGraw-Hill Book Co., New York, N.Y.
- Steinbach, D.W., and C.W. Ramsey. 1988.** The Texas "Lease System": History and future, p. 54-68. *In:* D. Rollins (ed.), *Recreation on rangelands: Promise, problems, projections*. Symp. Proc. Soc. Range Manage., Corpus Christi, Tex.
- Stidham, N.D., J. Powell, F. Gray, and P.L. Claypool. 1982.** Establishment, growth, utilization, and chemical composition of introduced shrubs on Oklahoma tallgrass prairie. *J. Range Manage.* 35:301-304.
- Teer, J.G., J.W. Thomas, and E.A. Walker. 1965.** Ecology and management of white-tailed deer in the Llano Basin of Texas. *Wildl. Monogr.* 15.
- USDA Forest Serv. 1974.** Seeds of woody plants in the United States. USDA Handb. 450.
- Vanderhoof, R.E., and H.A. Jacobson. 1989.** Effects of agronomic plantings on white-tailed deer antler characteristics. Paper presented at Twelfth Annu. Meeting Southeast Deer Study Group, Oklahoma City, Okla., Feb. 19-22.
- Verme, L.J., and D.E. Ullrey. 1984.** Physiology and nutrition, p. 91-118. *In:* L.K. Hall (ed.), *White-tailed deer ecology and management*. Stackpole Books, Harrisburg, Penn.
- Vines, R.A. 1960.** *Trees, shrubs, and woody vines of the Southwest*. Univ. Texas Press, Austin.
- Whisenant, S.G., D.N. Ueckert, and J.E. Huston. 1985.** Evaluation of selected shrubs for arid and semi-arid game ranges. *J. Wildl. Manage.* 49:524-527.
- Wiggers, E.P., D.D. Wilcox, and F.C. Bryant. 1984.** Cultivated cereal grains as supplemental forages for mule deer in the Texas Panhandle. *Wildl. Soc. Bull.* 12:240-245.

Interactions of pinyon and juniper trees with tebuthiuron applications at 2 matched rein-vaded sites in Utah

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Abstract

Tebuthiuron [N-[5-(1,1-dimethylethyl)-1,3,4-thiadiazol-2-yl]-N,N'-dimethylurea] controls small trees in regrown pinyon-juniper woodland chainings. Precise applications by hand minimize cost and damage to forage plants. Little information exists on the applicability of local trials to varying Intermountain sites slated for reinvestment. We conducted 2 balanced factorial experiments at well-separated sites in the Great Basin and Colorado Plateau. Four rates (0.12, 0.25, 0.50, and 0.75 gm a.i. per 130 dm³ crown volume) of tebuthiuron boluses were applied to tree stembase, mid-crown, or dripline placements. Four size classes (12-99, 100-299, 300-599, and 600-1099 dm³ crown volume) of Utah juniper [*Juniperus osteosperma* (Torr.) Little] and single needle (*Pinus monophylla* Torr. and Frem.), and Rocky Mountain (*P. edulis* Engelm.) pinyon trees were treated in September 1985. Defoliation and mortality levels were estimated 24 and 36 months following treatments. Both sites received highly similar amounts of herbicide and cumulative precipitation. Analysis of variance showed that the presence and strength of main effects and first order interactions was largely site-specific. Pinyon was more susceptible than juniper at either site. Medium-sized and large saplings were apparently more readily defoliated than seedlings and small saplings. Dosage effects were generally nonlinear for both species. The highest, most rapid and most uniform defoliation and mortality of trees resulted from application of tebuthiuron at the stem bases. This placement option has strong operational advantages and minimizes damage to forage plants beneath trees.

Key Words: pinyon, juniper, tebuthiuron, plant control, herbicidal trials

A location-by-treatment, species-by-treatment, or other kind of interaction results from many wildland experiments involving herbicides, fertilizers, soil amendments, or controlled burning (e.g., Clary, et al. 1985, Cook and Stubbendieck 1986, Herbel et al. 1985). For this reason, trials involving new species or sites, or both,

should be undertaken to test earlier results further, confirm generality, or adjust for interregional differences. Unfortunately, most field experiments are not designed to assess geographical generality (Pearce and Brown 1983). A related issue in applied ecology is "considering how spatial and temporal variability may influence the results of short-term experiments" (Gross 1989:264). Herbicidal trials generally last 1 to 3 years, and the prevailing weather may be atypical.

The efficacy of a herbicide depends on several factors (Scifres 1977), yet manufacturers and range managers often recommend and apply a uniform rate. An experimental procedure should be duplicated at numerous sites to determine whether the results of a treatment can be replicated in space or time. Expectations may be formulated as *a priori* hypotheses based on earlier research findings or long experience with a rangeland system (Provenza 1991).

Our study involved immature, noncommercial woodlands that are steadily reoccupying sites cleared to increase forage production. Our goal was the identification of optimal stand and treatment factors to defoliate trees. Saplings and seedlings invade or recover following 1960s-era chainings or cablings of pinyon-juniper woodlands. Economical control could extend the benefits of previous investments on the land (West 1984). One option involves solid tebuthiuron [N-[5-(1,1-dimethylethyl)-1,3,4-thiadiazol-2-yl]-N,N'-dimethylurea]. Hand application of this herbicide to individual trees reduces costs of materials per unit area (Van Pelt and West 1990). It also permits very precise placement on the ground surface. We compared the amount of defoliation from several combinations of placement and dosages of herbicide applied to an array of tree sizes. Two matched experiments were employed, 1 at each of 2 geographically and statistically independent sites. Analysis of variance (ANOVA), coupled with orthogonal contrasts, was the statistical tool used in conjunction with a fully balanced factorial design (Steel and Torrie 1980, Yang 1983).

Our hypotheses were that (i) tree species and size would strongly affect defoliation and mortality; (ii) defoliation and mortality would be linear for the application of equi-spaced tebuthiuron rates; (iii) placement-based differences would be very large, an expectation derived from the results of a related study (Van Pelt and West 1989); and (iv) differences evident early in the experiment among factor levels would later become insignificant. A supplementary, nonstatistical objective was to compare informally the response patterns between the 2 sites. This was done to derive an hypothesis for future resolution by chemical ecologists.

Methods

Study Sites

Our experimental locales clearly expressed salient climatic and soil characteristics of their ecoregions (Omernik 1987). Both have climatically variable, low-productivity rangelands. They chiefly

Approved as journal paper 4186 of the Utah Agricultural Experiment Station. This publication reports research involving pesticides. It does not contain recommendations for this use, nor does it imply that the uses discussed herein have been registered. All uses of pesticides must be registered by appropriate state and/or federal agencies.

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differ in moisture regime, soil family, and flora. Utah juniper [*Juniperus osteosperma* (Torr.) Little] and Rocky Mountain (*Pinus edulis* Engelm.) or singleneedle pinyon (*P. monophylla* Torr. & Frem.) were the only tree species present, but each site had only 1 pinyon species. The experiments were 169 airline km apart.

Treatments at the 2 typical sites involved highly similar sets of trees, identical setup procedures, application dates separated by 1 day, and nearly simultaneous response recordings. Each had an abundance of small trees from which matched sets of experimental units were selected. In addition, the sites occurred on similar topography (bajadas on the western sides of steep, moderately high mountain ranges). They differed in soil type and summer rainfall. We scrutinized records from stations in the vicinity of each and isopleth maps for Utah, to confirm that the sites had historically similar average annual precipitation.

The Black Rock Canyon (BRC) site, 58 km southwest of Provo, Ut., is on gentle, planar slopes at 1,830 m on the northwest side of the East Tintic Mountains. It was chained in 1964. Annual precipitation averaged between 300 and 350 mm, with most falling between December and March. Sporadic, generally low-intensity thunderstorms occurred in July, August, and September, but the climate was mainly summer-dry (Van Pelt and West 1989). The site was underlain by the Abela very gravelly loam, a fine-loamy over loamy-skeletal, mixed, mesic aridic calcic argixeroll (Van Pelt 1988). The site had a calcrete restricting layer and low production of herbaceous plants, constraining operational-scale retreatment. Singleneedle pinyon was the only pinyon species present.

The eastern site, East Carbon (EC), lies 33 km east-southeast of Price, Ut., at the base of the West Tavaputs Plateau. It was chained in 1966. The plots were at 1,980-m elevation, on west and west-southwest aspects. The long-term average precipitation was 300 to 350 mm, with about half falling during December through April and the remainder often as intense summer thundershowers. Study plots were underlain by the Mivida and Strych soils, occupying about 80% and 20% of the treated area, respectively. The deeper Mivida soil was a coarse-loamy, mixed, mesic ustollic calciorthid (Soil Conservation Service 1988). In contrast to BRC, EC had appreciable grass. It would thus show a greater rejuvenation of forage following operational-scale tree removal.

Selection of Trees, Dosage Rates, and Placements

We conducted 4 plant-dimensional, soils, and herbicidal-outlay comparisons among the sets of trees and between the sites prior to and immediately following tebuthiuron applications. The factorial arrangement included 2 levels of species, 4 of tree canopy size, 4 dosage rates plus 2 untreated controls, and 3 sub-canopy placements. With the exception of pinyon species, the factors and levels were identical at the 2 sites.

Four ranges of estimated crown volume (dm^3) were selected from the tree sizes available at the sites. Nearly all trees were less than 200 cm high, a suggested upper limit for control operations (Clary et al. 1985), even though height was not an explicit variable. The classes were: 12–99, 100–299, 300–599, and 600–1,099 dm^3 of crown volume. Crown volumes was chosen as the relevant size variable because it was easily and accurately estimated from linear stem and crown measurements. It also bears a strong correspondence to foliage mass and area (Tausch and Tueller 1988). Crown volume is the variable probably most related to tebuthiuron toxicity. Trees were all upright and vigorous, with no dead foliage. They were individually marked with aluminum tags, and were entirely independent in both the biological and statistical senses.

Four stepped rates of tebuthiuron were applied to evaluate the hypothesis of linear defoliation response. The rates were termed low (L), medium-low (ML), medium-high (MH), and high (H). Actual dosages within a rate were carefully scaled, in 9 increments, to estimated crown volume. Levels of placement were nominal,

reflecting obvious choices for particle application beneath a tree. The stembase, a mid-crown, and “dripline” (canopy-margin) options also correspond to precipitation collection or herbicide placements used by others (Young et al. 1984, Van Pelt and West 1989).

We maintained 3 temporary and 2 permanent (over-winter) rain gauges at both sites from June 1985 to October 1987. The gauges were emptied at irregular intervals during the summers of 1985, 1986, 1987, and 1988 as well as at the beginning and end of the water year. This procedure allowed comparison of cumulative precipitation at intermediate recordings as the experiment progressed.

Twenty-four surface (0–5 cm deep) soil samples were obtained at each of the 3 placements, beneath both juniper and pinyon at BRC and EC. The samples were submitted for laboratory analyses of texture and organic matter (Klute 1986). These characteristics influence the effectiveness and variability of tebuthiuron applications (Duncan and Scifres 1983).

Experimental Design, Field Layout, and Application Procedure

The 432 experimental units (individual trees) at each site were incorporated in a 2 (species) \times 4 (canopy sizes) \times 6 (rates plus controls) \times 3 (placements) balanced factorial, with 3 replications. However, the ANOVAs reported here include 4 factors of rate since the controls showed no defoliation and were not integral to the analysis. Controls were used in case a damaging natural agent caused discoloration of untreated trees during the course of the experiment. Therefore, the ANOVAs presented are for a 2 \times 4 \times 4 \times 3 factorial with 3 replications. We completely randomized the field design, to avoid any restriction of a factor level to a particular plot with possible soil anomalies (Hurlbert 1984). All terms in the ANOVA were tested against error because that is the most conservative approach to take regarding real effects (Hurst 1986).

We located, measured, and tagged trees within rectangular, carefully diagrammed plots. Plots were generally aligned on the contour. Our selection of qualifying trees observed Hurlbert's (1984) advocacy of randomization with interspersions of treatments, coupled with independence of experimental units.

We applied tebuthiuron formulated as Spike 250 “Brush Bullets”[®] (0.25 g active ingredient) by hand on 24 and 25 September 1985 at EC and BRC respectively. Particles (boluses) were gently hand-placed concentrically at 1 of 3 placements (dripline, midcrown, or at the stem base). The rates were structured in the ratios of 0.5: 1.0: 1.5 or 2.0, with ML as the base (1.0x) rate of 0.25 g a.i. per 130 dm^3 of crown volume.

Recording Procedure and Data Analysis

We recorded tree responses to the tebuthiuron treatments 12, 20, 24, 32, and 36 months post-treatment. Response was considered as either dead, adhering needles, or defoliation. Defoliation was estimated as a percentage of the live crown volume. Two estimators reconciled their estimates at the 20-month recording, and were in close agreement. Otherwise, the senior author performed all estimates. Almost all analyses below refer to the 24- and 36-month data. No control trees exhibited natural discoloration.

Tests of hypotheses followed standard ANOVA using an arc sin transformation (Hurst 1986), then linear contrasts among back-transformed means of simple and main effects. This transformation is appropriate for distributions of percentages with a wide range of values and/or with many very low and very high values. Contrasts are appropriate for analyzing experiments containing a variety of types and levels of treatment (Warren 1979, Steel and Torrie 1980, Mize and Schultz 1985).

Results

The assignment of trees to factor-level combinations, coupled

with nearly identical total dosages at both sites, afforded a strong basis for minimizing error relative to hypothesized site, species, size, and treatment (rate and placement) differences. East Carbon was the drier site for the first 2 years post-application, but by 36 months the 2 locales were highly similar in accumulated precipitation amounts (Table 1). The quantities shown are of both rainfall and snowfall.

Table 1. Cumulative precipitation (mm) at the sites, measured at each defoliation recording occasion.

Site	Months since application				
	8	12	20	24	32
	(mm)				
Black Rock Canyon	373	443	688	755	990
East Carbon	226	305	533	689	959

The primary means of demonstrating the careful matching of sets of trees was a supplemental ANOVA using estimated crown volume as the dependent variable (Van Pelt 1988:77-79). Size, which was deliberately varied among sets when trees were chosen, accounted for over 90% of total variance. After tebuthiuron applications, we also summed and compared numbers of boluses allocated to each site. The trees at EC received 6% less herbicide than those at BRC.

Soil textural analyses for BRC samples showed slightly more organic matter (OM) at the stem base than at other placements, and averaged 6% for all trees. No differences in textural proportions were evident among placements. Clay content averages 12%. At East Carbon, the least OM occurred at the dripline placement, and averaged 5% for all trees. As at BRC, no differences in texture occurred among placements. Clay content averaged 9%, with no differences detected among placements. Organic matter and texture probably did not influence overall response (Duncan and Scifres 1983) or placement differences.

At Black Rock Canyon, all main effects (factors) were significant after 24 months. However, they varied greatly in their importance and their participation in interactions (Table 2). Species was

Table 2. Comparative analyses of variance for percent defoliation response from the intermediate (24-month) and final (36-month) recordings at both sites.

BRC = Black Rock Canyon = EC = East Carbon.

Source of variation	df	24 months		36 months	
		BRC	EC	BRC	EC
		F value			
Species	1	55.0**	19.9**	30.6**	4.2**
Size	3	6.3**	6.1**	4.5**	6.4**
Species × Size	3	<1.0	<1.0	1.8	2.5
Rate	3	9.3**	9.8**	5.9**	7.0**
Species × Rate	3	3.3*	<1.0	3.9**	0.8
Size × Rate	9	1.3	3.2**	0.7	0.8
Species × Size × Rate	9	2.4*	3.2**	0.7	0.2
Placement	2	6.8**	3.6*	2.4	1.6
Species × Placement	2	3.7*	2.5	1.9	2.7
Size × Placement	6	2.3*	1.7	1.2	1.4
Species × Size × Placement	6	1.4	<1.0	1.8	1.5
Rate × Placement	6	<1.0	1.4	0.8	0.6
Species × Size × Rate × Placement	6	1.8	1.8	1.7	0.5
Size × Rate × Placement	18	1.6	<1.0	1.0	0.3
Species × Size × Rate × Placement	18	1.3	1.5	1.3	0.8
Replications	2	6.1**	<1.0	2.9	0.7
Error	190	—	—	—	—

*,** means significant at the 0.05 and 0.01 probability levels, respectively.

clearly the most important effect, followed by rate. Size and placement were nearly equal in effect. Second-order (3-term) interactions were rarely significant after 24 and 36 months (Table 2). At either occasion, species differences contributed more to variance than any other source. Placement differences were at first highly significant, but became insignificant after 36 months.

At East Carbon after 36 months, species differences were not nearly as pronounced as at 24 months. Rate and size were the most influential factors. Differences between pinyon and Utah juniper response were not as great as at BRC. Never a strong source of variation, placement had become insignificant after 36 months.

Inasmuch as size and rate within a species were most important at 36 months (at both sites), means are arrayed correspondingly in Table 3. Defoliation usually increased with higher herbicide application rates for any species and rate combination (Table 3). Many

Table 3. Defoliation (%) in 1988 at the East Carbon site (36-month recording). Within rows, means denoted by the same letter are not significantly different ($p > 0.05$).

Species and size	Rate			
	Low	Med. Low	Med. High	High
	(%)			
Utah Juniper				
Size 1 ¹	72b	48a	83b	64ab
Size 2	65a	60a	88b	100b
Size 3	77a	82a	98a	89a
Size 4 ²	88a	89a	100a	99a
Overall Mean	75ab	70a	93b	88b
Rocky Mountain Pinyon				
Size 1 ¹	73a	82ab	94ab	100b
Size 2	71ab	54a	89b	86b
Size 3	79a	95a	100a	90a
Size 4 ²	89a	100a	100a	100a
Overall Mean	78a	83a	96a	94a

¹Smallest size (12-99 dm³).

²Largest size (600-1099 dm³).

adjacent rates (for example, L and ML) were not significantly different, however. Other responses were nonmonotonic, or did not increase with a higher rate. Pinyon were slightly more susceptible than juniper of the same size class within a rate, but the difference was often not significant.

With few exceptions, trees of the 2 largest sizes were moderately more susceptible than smaller ones, a pattern true of both species here and at BRC (Table 3). Incidences of relatively low defoliation (about 70% or less) were confined to the 2 lowest rates and the smallest 2 tree sizes. Nearly all pinyon, but only the larger juniper, subjected to the highest 2 rates were dead within 3 years of application.

If tree sizes are ignored, the treatment (rate plus placement) choices available to an applicator comprise a two-way matrix for each species at each site (Fig. 1). A more divergent response of juniper is generally evident, as in the nearly complete defoliation achieved with singleneedle pinyon at the shallow-soil site (BRC). Highly effective and, conversely, ineffective options are readily seen (Fig. 1). The stembase in conjunction with 1 of the 2 higher rates is generally most effective. However, in several site-species-treatment instances the choice of rate and placement is immaterial if 36 months are allowed to elapse.

Discussion and Conclusions

Evaluation of Species and Size Differences

Utah juniper was consistently more tolerant of tebuthiuron than the associated pinyon species, but no correlative explanation was evident. The difference is greatest at earlier recordings and becomes

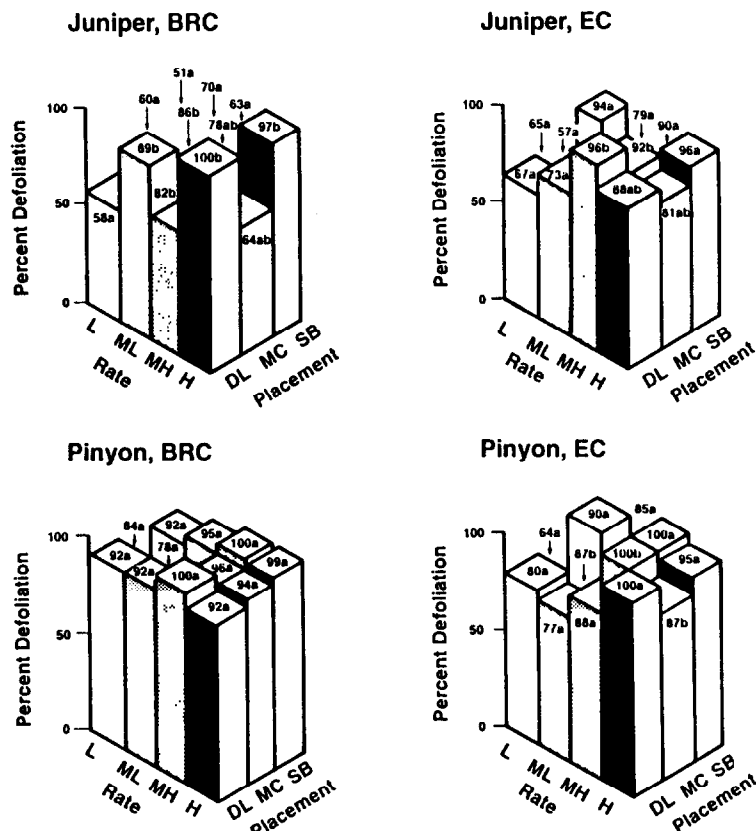


Fig. 1. Percent defoliation for dosage-rate and placement combinations, within each of the 4 species and site combinations. Within a placement, means denoted by the same letter are not significantly different ($p > 0.05$). BRC = Black Rock Canyon site; EC = East Carbon site; DL = dripline; SB = stembase; MC = midcrown; L = low; ML = medium-low; MH = medium-high; H = high.

less as 36 months are approached. Neither a particular size nor a placement were responsible. Placing herbicide at the dripline, where interception of rainfall by the crown would be immaterial, did not override the difference. It is known that juniper has greater foliar mass (Tausch and Tueller 1988), but translocation physiology, metabolism, and leaf anatomy could also account for response differences. These were untested here.

Species-and-size combinations had a less clear trend of increasing effect with higher tebuthiuron rates. The smallest juniper class was anomalous because 2 higher rates had less effect than their

adjacent, lower levels (Table 3). The averages for all sizes for each species had small (5–10%), often nonsignificant differences among rates. Differences between juniper (all sizes) and pinyon (all sizes) were slight for any rate, but pinyon was still more susceptible. The disparity between large and small trees might be less if dosages were scaled to height. Doing so is a much less time-consuming and tedious operational practice, anyway (Van Pelt and West 1990). Some intersize differences may have resulted from the width of the crown-volume classes that we defined. Too few trees were available to enable use of narrower class limits while still achieving experimental balance.

Effective and Robust Treatment Combinations

After 36 months, all but 1 of the rates used with the stem base placement at East Carbon yielded 90% or greater defoliation of both species (Fig. 1). The stem placement was similarly effective with pinyon at BRC. However, only the highest rate resulted in 90% or greater defoliation, with any placement, of juniper at BRC. The midcrown and dripline placements were frequently effective, but usually only when employed with the high and medium-high rates. If only 1 rate can be specified for tebuthiuron use at these and similar sites, it should be the highest (H), applied only to the stembases (SB) of all pinyon and juniper trees. For some site-and-species combinations, a lower rate will work if used at the stem base.

However, controlling juniper on Great Basin chainings with shallow-soil sites will require applying the highest rate, again at the stem base. That is, juniper on such sites will require greater inputs and will exhibit more variable response than pinyon. Damage to undergrowth plants, including forage species, is likely to be greatest on such sites. Although a single rate and placement treatment could be prescribed for pinyon in either ecoregion, juniper trees

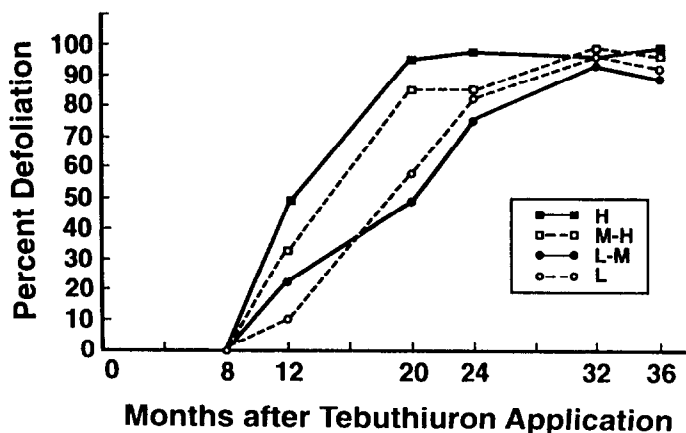


Fig. 2. Relative differences among the 4 rates of tebuthiuron application over the course of the experiment, in response to the stembase placement. Trees of all species and sizes at the East Carbon site comprise each curve. L = low; L-M = medium-low; M-H = medium-high; H = high.

still require more intrasite adjustment.

Possible causes of size-based differences include changes in growth rate with age, increases in litter depth and nonwettability beneath crowns as trees age, and greater interception and concentration of rain and snowfall beneath larger trees (Young et al. 1984, Van Pelt and West 1989). Perhaps the best explanation for the greater tolerance of the smallest 2 size classes to tebuthiuron would be that "smaller trees...generally have denser crowns with more foliage biomass per unit of crown volume or unit of basal area than...larger trees" (Tausch and Tueller 1988:43). The size-based differences evident here were too small to warrant special adjustments for large or small trees during practical retreatment operations.

Linearity of Dosage Effects

At EC, the ranking of rates by their effect depended on the levels of other factors. Application rate was involved in numerous interactions. In no instance were all the rates discrete in effect. The 2 highest ones were nearly always indistinguishable (especially with pinyon). This signified a defoliation asymptote (saturation) at some level below the maximum used. At both sites, the low rate was usually clearly distinguishable as having the least effect. Formal tests for linear, quadratic, and cubic effects on response were performed for selected factor levels (Steel and Torrie 1980). Linear effects predominated, but the stepped rates very rarely produced graphically equispaced responses. Most of the linearity is contributed by the 2 lowest rates.

Placement-Induced Differences and Convergence of Effects

Placement differences are the most explainable of the main effects at either site. The midcrown is the most sheltered from rain and snow, has considerable litter, and probably has few fine tree roots beneath it. The dripline is by far the most exposed, and lateral roots must extend to and beyond the crown margin. The stem base is about as sheltered as the midcrown, but water from stemflow concentrates there and can be intercepted by many fine roots (Young et al. 1984).

Consistent placement at the stem base over a project area will result in the lowest proportion of the area with tebuthiuron residues. Furthermore, it will best conceal relatively large particles from animals and humans. Stem base placement can be eased by dropping boluses through an inclined plastic pipe about 1-m long. Other work we reported from these sites (Van Pelt and West 1990) demonstrated the operational superiority of the stembase placement. Its use minimizes the tediousness of application, the time spent applying tebuthiuron particles to each tree, and defoliation of shrubs and grasses underneath. Our findings apply to trees of all sizes within the range of canopy volumes we treated.

Nonetheless, some placements in combination with the same rate evoked indistinguishable responses at all recording occasions. The 4 rates applied to the stem base diverged in effect throughout the intermediate (12, 20, and 24-month) occasions, but became statistically indistinguishable by 32 months (Fig. 2). Rate and placement differences probably become subdued simply because, given sufficient time, tebuthiuron suffuses through the root zones of the trees. In these climates, it takes nearly 3 years for convergence to become strongly apparent, however.

Trees of the same size, at 1 site, that are dosed identically do respond differently. This probably indicates the vagaries of rooting patterns and tebuthiuron uptake rather than inherently different tolerances. Thus, there was substantial error not attributable to the explicitly tested factors. Somewhat variable results will unavoidably afflict treatment operations, especially when Utah juniper are numerous. The advisability of followup operations would follow from economic analysis of the costs versus benefits of total tree control (Auld et al. 1987), and from derivation of a response

surface involving species, rate, and time-since-treatment (Box and Draper 1987).

Hypotheses and Ecoregional Differences

We did not devise a 'crucial' experiment to test a single hypothesis nor to discern the existence of a particular process or driving variable. Rather, we conducted it in agreement with Quinn and Dunham (1983:604):

The objective of investigation in cases of this sort is not to determine the single cause of a pattern, as no such cause exists, but rather to assign relative importances to the contributions of, and interactions between, a number of processes, all known or reasonably suspected of operating to some degree.

We confirmed that all tested factors were more or less influential, with their contributions both site- and time-dependent. If those attempting broadscale tree control wait long enough, some initial differences will "even out" (Hypothesis (iv) was retained). Treatment plans could thus concentrate on methods of getting herbicide to the trees efficiently, rather than on selection of an "ideal" rate.

We achieved our goal of specifying optimal treatment combinations, especially those manipulable by project planners. We accepted our hypothesis of strong species effects, but rejected those of linear defoliation response and large placement-based differences. The reality of size-based differences was more ambiguous.

The major difference in main effects between the 2 provinces was the greater disparity between Utah juniper and singleneedle pinyon defoliation at Black Rock Canyon than for juniper and Rocky Mountain pinyon at East Carbon. Since a different pinyon species grew at the latter site, any such difference (slight in practical terms) is not truly indicative of "ecoregion" influences (climates and soils). Pinyon species and site were unavoidably confounded. Other differences between sites after 36 months were evident in the ANOVA, but total variance was quite similar throughout.

The balanced experimental design, similar precipitation, and nearly identical amounts of herbicide applied provided a firm basis for comparing results from 2 ecoregions. The patterns of response of BRC and EC displayed some consistent similarities. Assuming the same rainfall and herbicide inputs, our proposed hypothesis is that rangeland sites mediate or apportion an equal total variance through soil differences. A site's response will reflect a greater or lesser emphasis on "natural" (species and size) versus "induced" (rate and placement) factors.

Literature Cited

- Auld, B.A., K.M. Lenz, and C.A. Tisdell. 1987. Weed control economics. Academic Press, London.
- Box, G.E.P., and N.R. Draper. 1987. Empirical model-building and response surfaces. John Wiley and Sons, New York.
- Clary, W.P., S. Goodrich, and B.M. Smith. 1985. Response to tebuthiuron by Utah juniper and mountain big sagebrush communities. *J. Range Manage.* 38:56-60.
- Cook, C.W., and J. Stubbendieck (eds.) 1986. Range research: basic problems and techniques. Soc. Range Manage., Denver, Colo.
- Duncan, K.W., and C.J. Scifres. 1983. Influence of clay and organic matter of rangeland soils on tebuthiuron effectiveness. *J. Range Manage.* 36:295-297.
- Gross, K.L. 1989. Report on "The Experimental Manipulation of Natural Plant Communities: What Have We Learned and What Remains to be Learned? A symposium organized by Jessica Gurevitch. ESA annual meeting, Toronto, Canada, August. 1989". *Bull. Ecol. Soc. Amer.* 70:263-265.
- Herbel, C.H., H.L. Morton, and R.P. Gibbens. 1985. Controlling shrubs in the arid Southwest with tebuthiuron. *J. Range Manage.* 38:391-394.
- Hurlbert, S.H. 1984. Pseudoreplication and the design of ecological field experiments. *Ecol. Monogr.* 54:187-211.
- Hurst, R.L. 1986. Factorial analysis of variance/covariance (FCT-FCTN). Department of Applied Statistics, Utah State Univ., Logan.
- Klute, A. (ed.) 1986. Methods of soil analysis. Part I. Physical and mineralogical methods. 2nd ed. Soil Sci. Soc. Amer. Madison, Wis.

- Mize, C.W., and R.C. Schultz. 1985. Comparing treatment means correctly and appropriately. *Can. J. For. Res.* 15:1142-1148.
- Omernik, J.M. 1987. Ecoregions of the United States. *Annals Assoc. Amer. Geogr.* 77:118-125.
- Pearce, S.C., and A.H.F. Brown. 1983. The agricultural field experiment: a statistical examination of theory and practice. John Wiley and Sons, Chichester, U.K.
- Provenza, F.D. 1991. Viewpoint: Range science and range management are complementary but distinct endeavors. *J. Range Manage.* 44:181-183.
- Quinn, J.F., and A.E. Dunham. 1983. On hypothesis testing in ecology and evolution. *Amer. Natur.* 122:602-617.
- Scifres, C.J. 1977. Herbicides and the range ecosystem: residues, research, and the role of rangemen. *J. Range Manage.* 30:86-91.
- Soil Conservation Service. 1988. Soil survey of Carbon County, Utah. USDA, SCS, Washington, D.C.
- Steel, R.G.D., and J.H. Torrie. 1980. Principles and procedures of statistics. McGraw-Hill Book Co., N.Y.
- Tausch, R.J., and P.T. Tueller. 1988. Comparison of regression methods for predicting singleleaf pinyon phytomass. *Great Basin Natur.* 48:39-45.
- Van Pelt, N.S. 1988. Dynamics and control of tree regrowth on previously cleared pinyon-juniper woodland sites in Utah. Ph.D. Diss., Utah State Univ., Logan.
- Van Pelt, N.S., and N.E. West. 1989. Aboveground dissolution of tebuthiuron particles placed beneath the crowns of small pinyon trees. *J. Environ. Qual.* 18:281-284.
- Van Pelt, N.S., and N.E. West. 1990. Effects of manual application method on application time, thoroughness and tebuthiuron outlays. *J. Range Manage.* 43:39-42.
- Warren, W.G. 1979. Analysis and interpretation of an experiment with a heterogeneous mixture of treatment types. *Biometrics* 35:869-872.
- West, N.E. 1984. Successional patterns and productivity potentials of pinyon-juniper ecosystems, p. 1301-1332. *In: Nat. Acad. Sci./Nat. Res. Council. Developing Strategies for Rangeland Management.* Westview Press, Boulder, Colo.
- Yang, R.C. 1983. Composite design versus factorial experiments in forest fertilization trials. *Can. J. Forest. Res.* 13:438-444.
- Young, J.A., R.A. Evans, and D.A. Easi. 1984. Stem flow on western juniper (*Juniperus occidentalis*) trees. *Weed Sci.* 32:320-327.

Grazing systems, pasture size, and cattle grazing behavior, distribution and gains

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Abstract

Reduced pasture size and distance to water may be responsible for the alleged benefits of intensive time-controlled rotation grazing systems. We compared cattle gains, activity, distance traveled, and forage utilization on a time-controlled rotation system with eight 24-ha pastures, on two 24-ha pastures grazed continuously (season-long), and on a 207-ha pasture grazed continuously, all stocked at the same rate. Utilization on the 207-ha pasture, but not on the 24-ha pastures, declined with distance from water. At distances greater than 3 km from water in the 207-ha pasture, utilization was significantly less than on adjacent 24-ha pastures, at distances of 1.0 to 1.6 km from water. Cows on the 207-ha pasture travelled farther (6.1 km/day) than cows on the 24-ha rotation pastures (4.2 km/day), which travelled farther than cows on the 24-ha continuously grazed pastures (3.2 km/day). Grazing system, range site, slope, and weather had minimal effects on cow activity patterns. Gains of cows and calves were less on the 207-ha pasture (0.24 and 0.77 kg/day, respectively) than on the 24-ha rotation pastures or 24-ha continuously grazed pastures (0.42 and 0.89 kg/day, respectively), with no differences between the latter. Calculated "hoof action" on the rotation pastures was less than that demonstrated to increase seed burial and seedling emergence. Intensive rotation grazing systems are unlikely to benefit animal performance unless they reduce pasture size and distance to water

below previous levels, decreasing travel distance and increasing uniformity of grazing.

Key Words: continuous grazing, grazing time, native range, season-long grazing, short-duration rotation grazing, time-controlled grazing, travel distance

Intensive time-controlled rotation grazing systems are characterized by multiple pastures, high stocking density, grazing periods short enough that regrowth is not grazed within the period, and lengths of grazing and rest periods that increase in length as forage growth rate decreases. Such systems have been called short-duration grazing (Savory 1978, Savory and Parsons 1980), Savory Grazing Method (SGM) or Holistic Resource Management (HRM) (Savory 1983), and planned grazing (Savory 1988). We have called the method time-controlled grazing (Hart et al. 1986) or short-duration rotation grazing (Hart et al. 1988a). The time-controlled rotation grazing used in this study will be called simply rotation grazing, since no other form of rotation grazing was examined.

Benefits claimed for time-controlled rotation grazing, in addition to doubled stocking rates, are those derived from concentrating animals in a small pasture to produce "hoof action". Hoof action supposedly breaks up surface crusts; aids water infiltration; incorporates litter and manure into the soil, speeding nutrient cycling; and buries seeds to help new plants become established (Savory 1983). Research has seldom confirmed the claimed bene-

fits of short-duration rotation grazing to animals, plants and soil (Bryant et al. 1989; Dormaar et al. 1989; Gillen et al. 1991; Hart et al. 1988a; Heitschmidt et al. 1982, 1985, and 1987; Taylor 1989; Weltz and Wood 1986). However, Hart et al. (1988a) confirmed that stocking rates can profitably be increased substantially above "government-prescribed stocking rates", although doubling them seems unduly risky (Hart 1991). Laycock (1983) and Lehnert (1985) pointed out that improved management is more important than rotation in achieving the benefits of grazing systems. Furthermore, subdividing large pastures to implement a rotation system may reduce distances to water and provide more uniform use of forage.

With these possibilities in mind, we designed a grazing study to separate the effects of time-controlled rotation from those of pasture size and distance to water, under uniformly good management. Our hypotheses were that (1) at the same stocking rate, cattle gains and activity and uniformity of grazing would be similar on rotation and continuously grazed pastures of similar size and shape with livestock water in similar locations, and (2) grazing would be less uniform and cattle would travel farther and gain less in a large continuously grazed pasture than in smaller rotation or continuously grazed pastures.

Materials and Methods

Pasture Layout

Layout of the experimental pastures is shown in Figure 1. The 2 continuous small pastures (CS1 and CS2) each covered 24 ha, except in 1986 and 1987 when the east fence of CS1 ran south to north rather than southwest to northeast and that pasture covered 34.4 ha. The continuous large pasture (CL) covered 207 ha, and was deliberately designed to produce a gradient of cattle distribution and forage utilization and to estimate the effects of these gradients on cattle gains. Each pasture contained a water source at one end. Maximum distances to water were 5.0 km on the continuous large pastures and 1.0 to 1.6 km on the rotation and continuous small pastures.

Forage Production and Utilization

Forage production and utilization estimates on Figure 1 indicate locations of 1.2 × 1.2-m exclosures 1986–1990. Exclosures at the

ends farthest from water of the rotation pastures and of continuous small pasture 2 were paired with exclosures across the fence in the continuous large pasture. In 1988–1990, additional exclosures were located near water in rotation paddocks 1, 3, 5, and 7 and in CL, comparable to the exclosures near water in CS1 and 2.

Peak standing crop was estimated in each exclosure in late July or early August each year. Production was estimated on two 0.18-m² quadrats within each exclosure with a capacitance meter. In every second or third exclosure, forage from 1 quadrat was clipped to ground level, dried, and weighed.

After cattle were removed from the pastures, residual herbage was estimated. Five capacitance meter readings 4 paces apart were taken, beginning at a random distance between 10 and 20 paces from each exclosure and walking in a random direction. One quadrat near every 2 or 3 exclosures was clipped, dried, and weighed. Meter readings and weights from the clipped quadrats were used to calculate a calibration equation. These equations were linear with correlation coefficients (*r*) of 0.86 to 0.93; *n* = 8 to 14.

Peak standing crop was the mean of estimates from the 2 quadrats in each exclosure or, in the case of paired exclosures, from the 4 quadrats in both exclosures. Utilization was calculated as (peak standing crop - residual)/peak standing crop × 100, from each of the 5 estimates of residual herbage near each exclosure.

To estimate small-scale heterogeneity of utilization, in 1987 2 pairs of transects were established in each small continuous pasture and in rotation pastures 4 and 5. Four pairs of transects were established in the large continuous pasture. In the small continuous pastures, 1 pair was placed 100 to 200 m from water and the other 400 to 700 m from water. Corresponding distances in the rotation pastures were 200 to 300 m and 700 to 1,000 m. In the large continuous pasture, transect pairs were 700, 900, 2,600, and 3,200 m from water. Transects consisted of 128 segments, each 8 cm long, for a total length of 10.24 m. Paired transects were parallel and about 20 m apart.

In July and September of 1987, aerial cover, fraction of segment showing grazing, and utilization was estimated for each segment of each transect. Utilization was scored as none, no herbage removed; light, up to 25% removed; moderate, 26 to 50% removed; heavy, 51 to 85% removed; and over-utilized, over 85% utilization. A grazed patch consisted of 1 or more grazed segments with ungrazed segment(s) on each side.

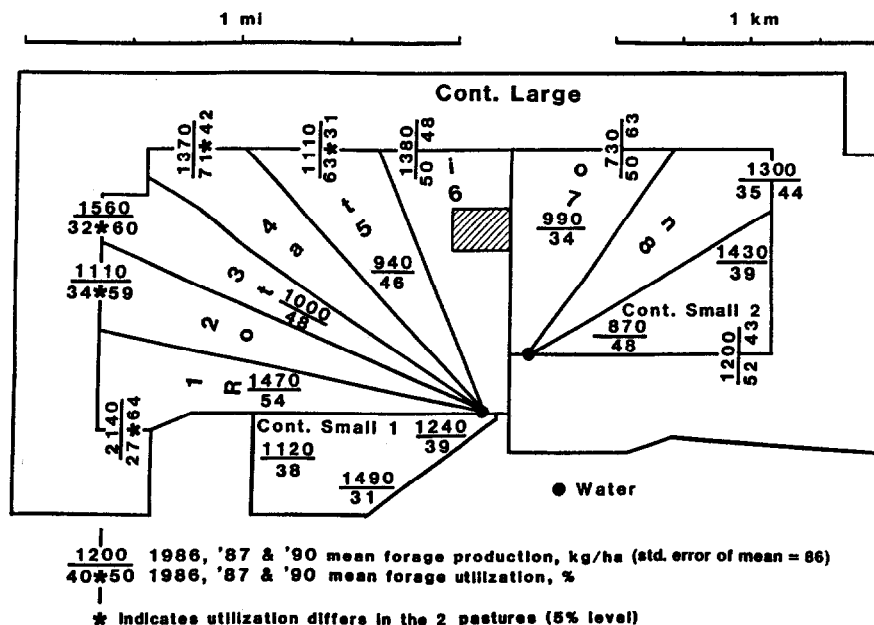


Fig. 1. Pasture layout, location of exclosures 1986–1990, and forage production and utilization.

Table 1. Forage production, days grazed, stocking rate, and grazing pressure on large and small continuously-grazed and small rotationally grazed pastures.

Year	Forage mean (std error)	Days grazed	----- Pasture -----		Pairs (1 AU)	Dry cows (0.9 AU)	Heifers (0.75 AU)	Animal units	----- AU days -----	
			Name	Size					/ha	/Mg
(kg/ha)				(ha)	(No.)	(No.)	(No.)			
1986	960 (82)	124	Continuous Large	207.3	36	6	13	51.15	30.6	31.9
			Continuous Small 1	31.8	7	0	3	9.25	36.1	37.6
			Continuous Small 2	24.3	6	0	0	6.00	30.6	31.9
			Rotation	194.3	40	5	11	53.25	34.0	35.4
1987	1190 (131)	148	Continuous Large	207.3	26	16	16	52.40	37.4	31.4
			Continuous Small 1	31.8	7	1	3	10.15	47.2	39.7
			Continuous Small 2	24.3	4	1	3	7.15	43.5	36.6
			Rotation	194.3	23	19	14	50.60	38.5	32.4
1988	1150 (118)	141	Continuous Large	207.3	19	27	10	50.80	34.6	30.0
			Continuous Small 1	24.3	3	3	0	5.70	33.1	28.8
			Continuous Small 2	24.3	3	0	5.70	33.1	28.8	
			Rotation	194.3	19	24	7	45.85	33.3	28.9
1989	530 (62)	77	Continuous Large	207.3	32	0	3	34.25	12.7	24.0
			Continuous Small 1	24.3	3	0	1	3.75	11.9	22.4
			Continuous Small 2	24.3	3	0	1	3.75	11.9	22.4
			Rotation	194.3	27	1	5	31.65	12.5	23.7
1990	1620 (199)	149	Continuous Large	207.3	30	2	19	46.05	33.1	20.4
			Continuous Small 1	24.3	3	1	1	4.65	28.5	17.6
			Continuous Small 2	24.3	4	0	1	4.75	29.1	18.0
			Rotation	194.3	27	5	10	39.00	29.9	18.5

Livestock Behavior and Gains

Pastures were stocked each spring with cow-calf pairs and, in most cases, dry cows and yearling heifers (Table 1). We tried to maintain the same stocking rates in all pastures, but this was not always possible with the available livestock and pasture sizes. If an animal became ill or died, it was replaced if a similar replacement was available, except in 1987. Cattle were weighed every 4 weeks, after feed and water were withheld overnight.

In 1989 and 1990, 6 nursing cows in the rotation herd, 6 in the herd on CL, and 3 in each herd on CS1 and CS2 were fitted with plastic neck chains. Each chain in a herd was a different color. On selected days, herds were observed from dawn to dusk and the activity and location (on a 100-m grid) of chained cows was recorded at 15-min intervals. The rotation herd was observed in pastures R4 and R5. Occasionally a cow would lose her neck chain during or just before an observation day so it was not possible to observe the planned number of cows on every day.

Distance travelled was calculated by summing straight line distances between centers of grid squares occupied at successive observations. This provides a more reliable estimate than use of pedometers (Anderson and Kothmann 1980, Walker et al. 1985), which essentially count steps and multiply by a constant. While stride length may be relatively constant during non-grazing travel, it varies greatly during grazing (Test 1984).

Grazing events per hectare were calculated as total number of observations per ha/number of cow-days observed \times number of animal-days per pasture per year. Each event was considered to represent 15 minutes of activity, so hours of grazing was calculated as events/4. Percent of time spent grazing, nursing, resting, or travelling was calculated as observations in that activity/total number of observations \times 100.

Data Analysis

Average daily gain of nursing cows, calves, dry cows, and heifers was subjected to analysis of variance in each year, with animals as experimental units (Conniff 1976). Distance travelled and percent of time spent in each activity were similarly analysed. Because no activity ever occupied 0 or 100% of the day, data were approximately normally distributed in spite of being percentage data. Differences were considered significant if probability of Type I error was less than 0.05. Analysis indicated no significant differ-

ence in gain of any class of cattle or in activity of nursing cows between CS1 and CS2, so data were pooled across these pastures. The same was true of rotation pastures R4 and R5.

Within each year, utilization was subjected to analysis of variance, with the 5 utilization estimates per exclosure treated as samples. Again, percentage data were normally distributed. The pooled sampling error mean square was used to test for differences between paired exclosures, for differences among exclosures within a pasture, and for differences among pastures. Mean utilization for 1986, 1987, and 1990 was analysed with years as main plots, exclosures as subplots, and utilization estimates as samples.

Utilization data for 1988 were eliminated from this analysis because herd composition was substantially different from that in other years. Distribution and forage utilization patterns of highly mobile dry cows are not the same as those of more sedentary nursing cows. No differences in use among pastures or locations within pastures were detected in 1988; mean use was 50%. Data from 1989 were eliminated because little forage was produced and cattle had to be removed when forage utilization reached only 17%.

Simple and multiple regression equations, using percent utilization or grazing time per hectare as the dependent variable and distance to water, percent slope and range site as independent variables, were calculated.

Grazed patches per transect, patch length, and segments overutilized as a percentage of all segments grazed were analyzed with each transect of a pair as samples within distances and within pastures. Heterogeneity chi-square was used to test variability in utilization among transects.

Results and Discussion

Cattle Activity

In 1989, cows in the continuous small pastures (CS1 and CS2) spent a smaller percentage of time grazing than cows in the continuous large pasture (CL; Table 2). No other differences in grazing time were observed. Cows spent about 93% of the time grazing and resting.

Inevitably, resting time was negatively correlated with grazing time. Cows on CS spent more time resting than cows on the rotation pastures (R4 and R5) or CL in 1989. Cows spent about 3% of the time nursing calves, with no differences among treatments in either year.

Table 2. Time spent in various activities by nursing cows on large or small continuously-grazed or small rotationally-grazed pastures.

Year	Pasture	Grazing	Resting	Travelling	Nursing
-----% of time observed-----					
1989	Continuous Large	60 a	30 a	6 a	4 a
	Continuous Small	51 b	43 b	3 b	3 a
	Rotation	57 ab	37 b	3 b	3 a
1990	Continuous Large	56 a	35 a	6 a	3 a
	Continuous Small	58 a	37 a	3 a	3 a
	Rotation	54 a	40 a	4 a	3 a

a, b Percentages within year and activity, followed by different letters, are different ($P \leq 0.05$).

Cows spent an average of 56% of daylight hours grazing, equivalent to 9.0 hr/day during the 16 hours of daylight at summer solstice in June and 7.8 hr/day during the 14 hours of daylight in late August and early September. Walker and Heitschmidt (1989) reported that cows grazed 11.3 hr/day in May and June and 9.5 hr/day in August and September. However these grazing times included night-time grazing, which was not observed in our study. Total grazing time did not differ among continuous grazing and rotational grazing in 14 or 42 paddocks. Hepworth et al. (1991) reported that steers grazed an average of 8.5 hr/day during daylight hours. In the third year of their study, steers at heavy stocking grazed longer under continuous than under rotation grazing; in no other case were differences found between systems.

Cows spent about twice as much time travelling on CL as on R or CS, but the difference was significant only in 1989. Cows travelled about 6.1, 3.2, and 4.2 km/day on CL, CS, and R, respectively (Table 3). They travelled farther on R than on CS, and about 70% farther on the 207-ha CL than on the 24-ha R or CS.

Table 3. Distance traveled by nursing cows on large or small continuously-grazed or small rotationally-grazed pastures.

Type of travel	Pasture	1989	1990	Mean
-----km/day-----				
Total	Continuous Large	6.4 a	5.8 a	6.1 a
	Continuous Small	3.2 c	3.2 c	3.2 c
	Rotation	4.4 b	4.0 b	4.2 b
Grazing	Continuous Large	4.0 a	3.6 a	3.8 a
	Continuous Small	2.6 b	2.8 b	2.7 c
	Rotation	3.2 ab	2.9 b	3.1 b
Non-grazing	Continuous Large	2.4 a	2.2 a	2.3 a
	Continuous Small	0.6 b	0.4 b	0.5 c
	Rotation	1.2 ab	1.1 ab	1.2 b

a, b Distances within type of travel and years or mean, followed by different letters, are different ($P \leq 0.05$).

Travel was divided into grazing and nongrazing travel. Nongrazing travel included travel to and from water or travel of over 100 m without stopping to graze; the latter contributed very little. Grazing travel was 3.8, 2.7, and 3.1 km/day on CL, CS, and R, respectively (Table 3), and was less in the small CS and R pastures than in the large CL pasture, with no difference between CS and R. Most of the differences in travel among systems occurred when the cows were not grazing. Nongrazing travel was 2.3, 0.5, and 1.2 km on CL, CS, and R; all differences were significant.

Walker and Heitschmidt (1989) reported that cows travelled 5.8 km/day in a 248-ha continuous pasture and 6.5 and 8.2 km/day in 27-ha and 10-ha rotation pastures, respectively. Pastures in this study were similar in size to ours, yet differences between sizes were small, probably because maximum distance to water varied only from 1.3 to 1.4 km among treatments, vs. 1.0 to 5.0 km in our study. Heifers travelled 5.5 km/day in 4-ha rotation pastures vs. 6.1

km/day in a 20-ha continuous pasture (Anderson and Kothmann 1980). Hepworth et al. (1991) estimated that steers travelled 2.7 km/day in pastures where maximum distance to water was 640 m vs. 1.9 km/day where distance was 240 m. Distance to water, not pasture size or grazing system, appears to be the major factor controlling distance travelled.

Nongrazing travel on R and CS was *positively* correlated with the number of times cows went to water; r values were 0.76 and 0.78, respectively. On CL, nongrazing travel was *negatively* correlated with times at water; $r = -0.62$. As cows grazed farther from water on the large pasture, they went to water less often, regardless of weather. Fewer trips compensated for the greater distance to water.

Cows did not go to water at all on cool damp days such as 20 July 1990 (maximum temperature 14° C, 7 mm of rain), and usually stopped grazing when it was actually raining. No other relationships were detected between weather and time in activity or distance travelled, although Anderson and Kothmann (1980) found distance travelled was correlated with precipitation, temperature, and the ratio of water vapor to dry air in the atmosphere.

Hoof Action

An estimate of "hoof action" can be calculated from travel distance and stocking density. Nongrazing travel contributed little to hoof action, because nearly all of it was done on established paths. Therefore our calculations were based on grazing travel.

Test (1984) spent many hours observing grazing cattle and concluded that, while length of stride varied greatly depending on what the animal was doing, average stride lengths of mature cows were about 45 and 90 cm, respectively, while grazing and travelling. Thus for each kilometer traveled by a cow while grazing, each leg took about 2,222 steps or approximately 9,000 steps for all 4 legs. Measurements of hoofprints showed an average area of about 100 cm², so about 90 m² was trampled for each kilometer traveled per animal. Heifers would take slightly shorter steps and therefore more steps per km, but hoofprints would be smaller so the same area per km traveled will be used.

Heaviest stocking on R occurred in 1987, with 56 cows and heifers on the system. With all 56 head in a single 24-ha pasture, each travelling 3.1 km/day while grazing, the maximum area trampled if no overlap occurred would be 15,624 m² or 1.56 ha. Some steps almost certainly overlapped but it is not possible to calculate how many. In a 144-day grazing season on an 8-pasture system, 18 days per pasture, the area trampled would be about 28 ha or 1.17 times the area of the pasture. Adding trampling by calves would slightly increase this figure, but allowing for overlap of hoofprints would decrease it. Dividing the area into more pastures, assuming that travel distance was unchanged, would not increase the total area trampled but would concentrate the effect in fewer days.

Trampling densities may influence seedling establishment. Under a 10-paddock intensive rotation grazing system, nearly all crested wheatgrass seedlings were destroyed by trampling (Salihi and Norton 1987). Zero, light, or heavy trampling, followed by rain, buried 20, 28, and 45% respectively of seeds of 4 grass species within the "biological limit" for emergence (Winkel et al. 1991). Differences between no and light trampling were not significant; differences between light and heavy trampling were significant in 2 of 8 year \times species combinations. Heavy trampling increased seedling emergence of 1 of 4 grasses in a wet year, all 4 in a moderately wet year, and none of the 4 in a dry year (Winkel and Roundy 1991). Heavily trampled soils took up water faster than untrampled soils immediately after trampling, but took up less water later in the season (Roundy et al. 1992). Light trampling was approximately 10 hoofprints per m², or 0.1 of the area trampled as calculated by the method outlined above. The area affected by heavy trampling is

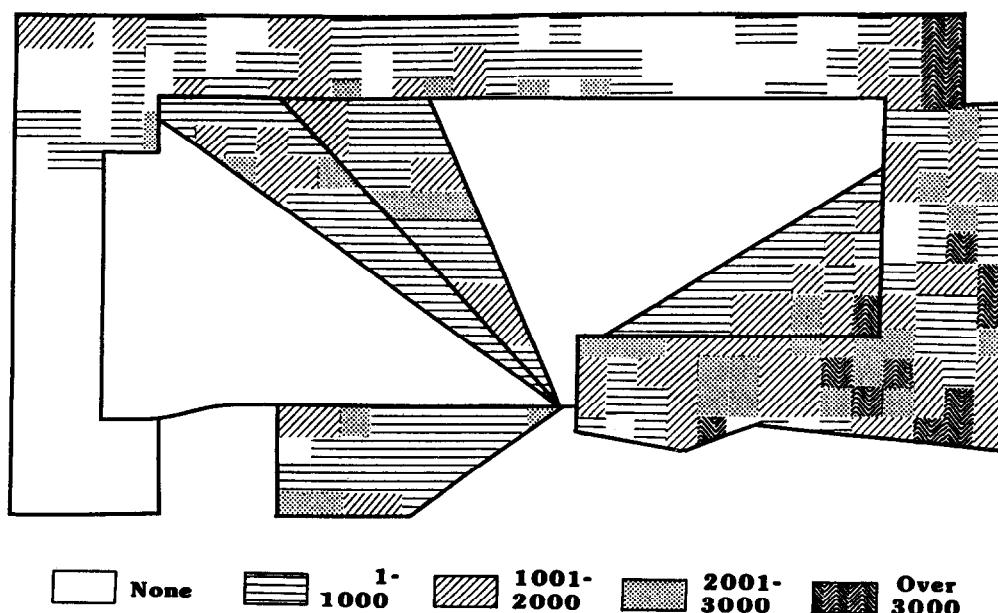


Fig. 2. Cattle hours of grazing per hectare, 1990. Activity was estimated in pastures 4 and 5 of the rotation treatment, and in both small and the large pastures of the continuous grazing treatments.

harder to calculate. Five cattle were herded inside a 6×6 m enclosure for 20 min. If they constantly walked at 6 km/hour, all 5 cattle travelled a total of 10 km and trampled an area of 900 m² or 25 times the area of the plot. Thus trampling at about 20 times the density observed in our study only sometimes increased emergence and only temporarily increased water infiltration. Abdel-Magid et al. (1987) and Taylor (1989) calculated that trampling at this density would reduce water infiltration and increase soil bulk density.

Uniformity of Use

Percent use (Fig. 1) did not differ significantly among locations within the 24-ha continuously grazed pastures (CS) or rotation paddocks (R). Use declined with distance from water in CL, the 207-ha continuously grazed pasture; $Use = 0.60 - 0.0059 D$, when D = distance to water in m; $r^2 = 0.55$. At distance greater than 3 km from water in CL, use was less than in adjacent rotation pastures with distances to water of 1.0 to 1.6 km. Webb (1931) noted that by the end of the 19th century, government publications stated that cattle should not walk more than 2 1/2 miles (4 km) to water. Nevertheless, some cattle producers are astonished to discover an increase in uniformity of use and livestock production when enormous pastures are subdivided and new water sources provided as part of a grazing system (Laycock 1983, Bryant et al. 1989, Taylor 1989). They credit the improvement to rotation grazing, not to reduced distances to water.

It should be emphasized that fencing to divide a larger pasture with a single pre-existing water source into a number of smaller pastures radiating out from that source does not reduce the distance to water on any part of that pasture, regardless of pasture size or location of the water source. On the other hand, providing new water sources can reduce the distance to water on large areas of the pasture without any pasture subdivision whatsoever. Everhart (1991) points out that locating water sources so pastures can be subdivided into smaller square rather than wedge-shaped pastures greatly reduces the amount of fence needed and more efficiently decreases the average distance to water. Walker and Heitschmidt (1986) found that dividing a pasture into 14 wedge-shaped pastures increased the number of cattle trails, and further dividing one of these pastures into 3 increased the number of trails even more.

At a finer scale along the paired transects, there were few differences among systems. In July, significantly more of the grazed segments were over-utilized under R (15%) than under CS (2%); CL was intermediate at 7%. By September, more grazed segments were overused under CS (46%) than under R (30%); CL was again intermediate at 37%. The rotation schedule was such that more animal-days of grazing had occurred on the R paddocks measured than on CL or CS in July; the reverse was true in September. By September all systems averaged 2.4 grazed patches per meter of transect, and average grazed patch length was 21 cm on R and CS and 17 cm on CL; differences were not significant.

Estimated cow-hours of grazing per hectare in 1990 (Fig. 2) also were related to D or distance to water in meters; $Cow-hour/ha = 1726 - 0.35 D$; $r^2 = 0.17$. No significant correlations between grazing time and slope or range site were detected, although cows appeared to spend more time grazing along the intermittent streams in R4, R5, and CL than in adjacent uplands. Senft et al. (1983, 1985) and Launchbaugh et al. (1990) found much greater differences in preference among sites, but vegetation also differed more among sites in their study. Our sites were quite similar, although loamy sites produced more total forage and blue grama (56% vs. 50% by weight) and less needleandthread (4% vs. 12%) than gravelly loamy sites, and supported somewhat different forb populations. However, forbs produced only 8% of total production (Hart and Samuel 1985).

DeYoung et al. (1988) found that cattle made more uniform use of vegetation types under rotation than under continuous grazing, but distribution was more influenced by soil series and distance to water under rotation grazing. Stuth et al. (1987) found cattle spent more time grazing preferred sites under rotational than under continuous grazing. Walker et al. (1989) found that cattle were more selective for plant communities under rotation than under continuous grazing, and more selective at the beginning of a rotation grazing period than at the end. They concluded that selectivity declined with decreasing forage and increasing grazing pressure, and grazing system effects were indirect through the impact of systems on grazing pressure.

Grazing time in CL dwindled with distance from water, just as percent use did. Cattle were never observed grazing at the end of

CL farthest from water, although dungpats indicated they sometimes went there. The average of 27% use observed at the far end also indicates some grazing, but some of this might be credited to insects and rodents.

Cattle Gains

Heavy use of forage near water and little use far from water, plus increased travel time and distance, reduced cow and calf gains on the large continuously grazed pasture (CL). Average daily gains of nursing cows were lower on CL than on the rotation pastures (R) or on the small continuously grazed pastures (CS) in all 5 years studied (Table 4). Only in 1988 were gains of nursing cows less on CS than on R. Calf gains did not differ between R and CS in any

Table 4. Gains of nursing and dry cows, calves, and yearling heifers on large and small continuously-grazed and small rotationally grazed pastures.

Year	Pasture	Nursing cows	Dry cows	Calves	Heifers
		Gain, km/day			
1986	Continuous Large	0.08 b	0.46 a	0.68 b	0.51 a
	Continuous Small	0.24 a	—	0.78 a	0.65 a
	Rotation	0.21 a	0.51 a	0.77 a	0.59 a
1987	Continuous Large	0.31 b	0.69 a	0.73 b	0.62 a
	Continuous Small	0.54 a	—	0.85 a	0.71 a
	Rotation	0.44 a	0.71 a	0.83 a	0.71 a
1988	Continuous Large	0.17 c	0.66 b	0.73 b	0.62 a
	Continuous Small	0.23 b	0.68 b	0.95 a	—
	Rotation	0.34 a	0.73 a	0.83 ab	0.74 a
1989	Continuous Large	0.23 b	—	0.83 b	0.79 a
	Continuous Small	0.42 a	—	1.06 a	0.94 a
	Rotation	0.49 a	0.58	0.94 ab	0.76 a
1990	Continuous Large	0.42 b	0.85 a	0.89 a	0.74 b
	Continuous Small	0.61 a	0.79 a	0.97 a	0.78 ab
	Rotation	0.61 a	0.84 a	0.94 a	0.87 a
Mean	Continuous Large	0.24 b	0.66 a ¹	0.77 b	0.66 a
	Continuous Small	0.41 a	—	0.92 a	—
	Rotation	0.42 a	0.70 a	0.86 a	0.73 a

a, b Gains of the same class of cattle in the same year, followed by different letters, are different ($P \leq 0.05$).

¹Mean gains of dry cows exclude 1989 when no dry cows grazed the large continuous pasture.

year. Calf gains on CL were lower than on CS in all but 1990, and lower than on R in 1986 and 1987.

Reducing pasture size from 207 to 24 ha usually produced marked improvements in cow and calf gains, regardless of grazing system. On 24-ha pastures, grazing system seldom affected cow or calf gains. Cow and calf gains may increase under rotation grazing systems, but because of reduced pasture size, resulting in reduced distance travelled and more uniform grazing, not because of rotation *per se*.

Cows on CL travelled about 2.9 km/day farther than cows on CS and 1.9 kg farther than cows on R (Table 3). Hepworth et al. (1991), using data of Brody (1945) and Clapperton (1964), calculated an energy requirement of 51 kilocalories/km of travel/100 kg of body weight. Ribiero et al. (1977) calculated 48 kilocalories/km of travel/100 kg of body weight. Average weight of cows observed in the current study was about 500 kg. At an energy requirement of 4,200 kilocalories/kg of gain (Garrett et al. 1959) and 50 kilocalories/km of travel/100 kg of body weight, CL cows should have gained 0.17 kg/day less than CS and 0.11 kg less than R cows. The observed differences in average daily gain were similar, 0.17 kg less than CS (0.24 vs. 0.41 kg) and 0.18 kg less than R (0.24 vs 0.42 kg; Table 4).

Gains of heifers and dry cows showed little response to grazing

system or pasture size. Heifer gains in 1990 were lower on CL than on R. No other significant differences in heifer gains were detected among treatments, partly because of the small number of heifers per pasture. Dry cows on CS and CL gained less than those on R in 1988. Pasture size or distance to water made little difference to dry cows; free of maternal responsibilities, they ranged widely.

Conclusions

Results of this study emphasize the importance of 2 requirements of sound range management, proper stocking rate and even livestock distribution. They also demonstrate that these requirements can be achieved independently of grazing system. Uniformity of grazing use; time spent grazing, resting and traveling; distance travelled by cows; and cattle gains were similar under continuous and time-controlled rotation grazing when pastures on the 2 systems were similar in size, shape, and maximum distance to water. But under continuous grazing with longer maximum distance to water, travel distance increased, grazing (as measured by both utilization and time spent grazing) was much heavier near water than at distances greater than 3 km, and gains of nursing cows and calves decreased. Installation of a rotation grazing system is unlikely to produce higher cattle gains, greater stocking rate, or more uniform grazing unless it is coupled with pasture subdivision and the provision of additional water sources. Subdivision and water may be provided more economically without implementing rotation grazing.

Literature Cited

- Abdel-Magid, A.H., M.J. Trlica, and R.H. Hart. 1987. Soil and vegetation responses to simulated trampling. *J. Range Manage.* 40:303-306.
- Anderson, D.M., and M.M. Kothmann. 1980. Relationship of distance traveled with diet and weather for Hereford heifers. *J. Range Manage.* 33:217-220.
- Brody, S. 1945. *Bioenergetics and Growth*. Reinhold Publ. Co., N.Y.
- Bryant, F.C., B.E. Dahl, R.D. Pettit, and C.M. Britton. 1989. Does short-duration grazing work in arid and semiarid regions? *J. Soil Water Conserv.* 44:290-296.
- Clapperton, J.L. 1964. The energy metabolism of sheep walking on the level and on gradients. *Brit. J. Nutr.* 18:47-54.
- Conniff, D. 1976. A comparison of between and within herd variance in grazing experiments. *Irish J. Agr. Res.* 15:39-46.
- DeYoung, C.A., A. Garza Jr., T.F. Kohl, and S.L. Beasom. 1988. Site preference by cattle under short duration and continuous grazing management. *Tex. J. Agr. Natur. Res.* 2:35-36.
- Dormaer, J.F., S. Smollak, and W.D. Willms. 1989. Vegetation and soil responses to short duration grazing on fescue grasslands. *J. Range Manage.* 42:252-256.
- Everhart, M.E. 1991. The preferred grazing system. *Rangelands* 13:266-270.
- Garrett, W.N., J.H. Meyer, and G.P. Lofgreen. 1959. The comparative energy requirements of sheep and cattle for maintenance and gain. *J. Anim. Sci.* 18:528-546.
- Gillen, R.L., F.T. McCollum, J.E. Hodges, J.E. Brummer, and K.W. Tate. 1991. Plant community responses to short duration grazing in tallgrass prairie. *J. Range Manage.* 44:124-128.
- Hart, R.H. 1989. SMART: a simple model to assess range technology. *J. Range Manage.* 42:421-424.
- Hart, R.H. 1991. Managing livestock grazing for risk; the STEERISK spread sheet. *J. Range Manage.* 44:227-231.
- Hart, R.H., and M.J. Samuel. 1985. Precipitation, soils and herbage production on southeast Wyoming range sites. *J. Range Manage.* 38:522-525.
- Hart, R.H., M.J. Samuel, P.S. Test, and M.A. Smith. 1988a. Cattle, vegetation, and economic responses to grazing systems and grazing pressure. *J. Range Manage.* 41:282-286.
- Hart, R.H., G.E. Schuman, M.J. Samuel, and M.A. Smith. 1986. Time-controlled grazing research in Wyoming, p. 73-82. *In: J.A. Tiedeman (ed.), Short duration grazing*. Wash. State Univ., Pullman.
- Hart, R.H., J.W. Waggoner, Jr., T.G. Dunn, C.C. Kaltenbach, and L.D. Adams. 1988b. Optimal stocking rate for cow-calf enterprises on native range and complementary improved pastures. *J. Range Manage.* 41:435-441.

- Heitschmidt, R.K., S.L. Dowhower, R.A. Gordon, and D.L. Price. 1985. Response of vegetation to livestock grazing at the Texas Experimental Ranch. *Tex. Agr. Exp. Sta. B-1515*.
- Heitschmidt, R.K., S.L. Dowhower, and J.W. Walker. 1987. 14- vs. 42-paddock rotational grazing: aboveground biomass dynamics, forage production, and harvest efficiency. *J. Range Manage.* 40:216-223.
- Heitschmidt, R.K., J.R. Frasure, D.L. Price, and L.R. Rittenhouse. 1982. Short duration grazing at the Texas Experimental Ranch: weight gain of growing heifers. *J. Range Manage.* 35:375-379.
- Hepworth, K.W., P.S. Test, R.H. Hart, M.A. Smith, and J.W. Waggoner Jr. 1991. Grazing systems, stocking rates, and cattle behavior in southeastern Wyoming. *J. Range Manage.* 44:259-262.
- Launchbaugh, K.L., J.W. Stuth, and J.W. Holloway. 1990. Influence of range site on diet selection and nutrient intake of cattle. *J. Range Manage.* 43:109-116.
- Laycock, W.A. 1983. Evaluation of management as a factor in the success of grazing systems. USDA Forest Serv. Gen. Tech. Rep. INT-157.
- Lehnert, E.R. 1985. Seven years of planned grazing systems data on twelve Central and North Central Nebraska ranches. *Soc. Range Manage. Abstr.* 38:24.
- Ribiero, C.R. de, J.M. Brockway, and A.J.F. Webster. 1977. A note on the energy cost of walking cattle. *Anim. Prod.* 25:107-110.
- Roundy, B.A., V.K. Winkel, H. Khalifa, and A.D. Matthias. 1992. Soil water availability and temperature dynamics after one-time heavy cattle trampling and land imprinting. *Arid Soil Res. Rehab.* 6:53-69.
- Salih, D.O., and B.E. Norton. 1987. Survival of perennial grass seedlings under intensive grazing in semi-arid rangelands. *J. Appl. Ecol.* 24:145-151.
- Savory, A. 1978. A holistic approach to ranch management using short-duration grazing, p. 555-557. *In*: D.N. Hyder (ed.), *Proc. 1st Int. Range Congr.*, Soc. Range Manage., Denver, Colo.
- Savory, A. 1983. The Savory grazing method or holistic resource management. *Rangelands* 5:155-159.
- Savory, A. 1988. *Holistic Resource Management*. Island Press, Covelo, Calif.
- Savory, A., and S.D. Parsons. 1980. The Savory grazing method. *Rangelands*. 2:234-237.
- Senft, R.L., L.R. Rittenhouse, and R.G. Woodmansee. 1983. The use of regression models to predict spatial patterns of cattle behavior. *J. Range Manage.* 36:553-557.
- Senft, R.L., L.R. Rittenhouse, and R.G. Woodmansee. 1985. Factors influencing patterns of cattle grazing behavior on short-grass steppe. *J. Range Manage.* 38:82-87.
- Stuth, J.W., P.S. Grose, and L.R. Roath. 1987. Grazing dynamics of cattle stocked at heavy rates in a continuous and rotational grazed system. *Appl. Anim. Beh. Sci.* 19:1-9.
- Taylor, C.A. Jr. 1989. Short-duration grazing: experience from the Edwards Plateau region of Texas. *J. Soil Water Conserv.* 44:297-302.
- Test, P.S. 1984. Vegetation and livestock response to three grazing systems—continuous, rotationally deferred and short-duration rotation. Ph.D. Diss., Univ. Wyoming. Laramie Diss. Abst. 8508879.
- Walker, J.W., and R.K. Heitschmidt. 1986. Effect of various grazing systems on type and density of cattle trails. *J. Range Manage.* 39:428-431.
- Walker, J.W., and R.K. Heitschmidt. 1989. Some effects of a rotational grazing treatment on cattle grazing behavior. *J. Range Manage.* 42:337-342.
- Walker, J.W., R.K. Heitschmidt, and S.L. Dowhower. 1985. Evaluation of pedometers for measuring distance traveled by cattle on two grazing systems. *J. Range Manage.* 38:90-93.
- Walker, J.W., R.K. Heitschmidt, and S.L. Dowhower. 1989. Some effects of a rotational grazing treatment on cattle preference for plant communities. *J. Range Manage.* 42:143-148.
- Webb, W.P. 1931. *The Great Plains*. Ginn & Co., N.Y.
- Weltz, M., and M.K. Wood. 1986. Short duration grazing in central New Mexico: effects on infiltration rates. *J. Range Manage.* 39:365-368.
- Winkel, V.K., and B.A. Roundy. 1991. Effects of cattle trampling and mechanical seedbed preparation on grass seedling emergence. *J. Range Manage.* 44:176-180.
- Winkel, V.K., B.A. Roundy, and D.K. Blough. 1991. Effects of seedbed preparation and cattle trampling on burial of grass seeds. *J. Range Manage.* 44:171-175.

Technical Notes: Double Sampling Revisited

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Abstract

The decision to use double sampling with a regression or ratio estimator is not a simple task. This study was conducted to determine whether a ratio or regression estimator should be used to estimate aboveground biomass of stands dominated by blue grama (*Bouteloua gracilis* (H.B.K.) Lag. ex Steud.) in eastern Colorado. One hundred 0.25-m² circular plots were systematically located in a homogeneous stand of blue grama, and on each plot biomass was estimated visually and then clipped. Three methods (classical, jackknife, and bootstrap) of estimating the variance for double sampling with regression and ratio estimator were compared in a simulation study using sample sizes 10, 20, 30, 40, and 50 clipped plots. The ratio estimator consistently had smaller bias and should be used for estimating average clipped weight of blue grama. For $n = 10$ clipped plots, the jackknife variance estimator is recommended for constructing confidence intervals. For $n \geq 20$ clipped plots, the classical variance estimate should be used to obtain reliable estimates of the population variance and in estimating confidence intervals.

Key Words: biomass, blue grama, bootstrap, double sampling, jackknife, monte carlo simulation

Pechanec and Pickford (1937) were among the first to estimate weight of vegetation by a combination of guessing and clipping. This method became known as double sampling. It has been described by several authors, and Francis et al. (1979) provided an extensive review of literature on the method. One of 2 statistical procedures can be used to express the relationship between clipped (y) and estimated (x) values: (1) linear regression, or (2) ratio estimation. Francis et al. (1979) compared the 2 methods, but their study was limited to analysis of variance estimates.

A major problem encountered in the combined use of clipping and estimating biomass is determination of the proportion of clipped plots to estimated plots. Of special interest is the ratio needed to obtain the desired accuracy of estimated biomass for the

greatest economy. It is well known that efficiency of double sampling depends on precision of the ratio or regression estimator, and on the relative cost of clipping compared to that of estimation. The objective of this study was to evaluate the use of ratio and regression estimators for determining the aboveground biomass of stands dominated by blue grama [*Bouteloua gracilis*, (H.B.K.) Lag. ex Steud.] in eastern Colorado.

Methods

One-hundred 0.25-m² circular plots were systematically located in a homogeneous stand of blue grama at 3-m spacings along twenty 15-m transects, which were spaced 5 m apart. Aboveground biomass of blue grama in each plot was estimated visually using 1 observer with 2 years of experience to eliminate any possible bias in estimated weights. Biomass in each plot was then clipped and weighed in the field. Samples were oven-dried in the laboratory at 60° C to a constant weight. The assumption that biomass of blue grama was normally distributed was evaluated with the Anderson-Darling test statistic at $\alpha = 0.05$ (Stevens 1974).

A Monte-Carlo simulation of 10,000 samples of size $n = 10, 20, 30, 40,$ and 50 clipped plots were drawn from the $n = 100$ plots by simple random sampling without replacement. For each sample, estimates of the average clipped biomass ($g\ 0.25\ m^{-2}$) were computed using regression and ratio estimation.

In regression, the estimate of the clipped mean over all plots, \bar{y}_r is given by

$$\bar{y}_r = \bar{y} + B(\bar{x}' - \bar{x}) \quad (1)$$

where \bar{y} is the mean of clipped subsample plots, \bar{x} is the mean of the visually estimated subsample, and \bar{x}' is the mean of visually estimated weights in all 100 plots.

In ratio estimation,

$$\bar{y}_r = R\bar{x}' \quad (2)$$

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Table 1. Descriptive statistics for blue grama biomass ($g \cdot 0.25\ m^{-2}$) on 100 sample plots.

Aboveground biomass	Mean	Median	Minimum	Maximum	Variance	Anderson-Darling test statistic for normality
----- (g/0.25 m ²) -----						
Visually estimated (fresh wt.)	4.29	3.0	0	19.0	17.28	7.059*
Clipped (fresh wt.)	4.95	4.0	0	23.0	23.62	6.162*
Oven-dry weight	3.36	3.0	0	15.5	9.40	2.864*

*Significant at $\alpha = 0.05$.

where the ratio R is obtained from

$$R = \frac{\bar{y}}{\bar{x}} \quad (3)$$

The decision to use the ratio or regression estimator depends on the relationship between clipped and visually estimated biomass. If a line fitting the data does not pass through the origin, and the distribution of clipped biomass is about the same throughout the range of visually estimated biomass, the regression estimator is most appropriate. If a line fitting the data passes through or very close to the origin, and the distribution of clipped biomass is proportional to visually estimated biomass, the ratio estimator is more appropriate (Cochran 1977).

The bias for the 2 double-sample estimators was computed as the difference between the average of the 10,000 estimates of clipped biomass and the average clipped biomass on the 100, 0.25-m² circular plots. Classical estimates of variance for both procedures, given by Cochran (1977) and Francis et al. (1979), were computed. In addition, variance estimates for (1) and (2) were computed using jackknife and bootstrap procedures (Schreuder and Ouyan 1992).

The jackknife procedure begins by removing one of the observations (i.e., plot) from the sample data. The desired statistic is then computed each time, with one of the observations eliminated (Smith and van Belle 1984). Standard error is then computed from the variability among these estimated values (Sokal and Rohlf 1981).

The bootstrap procedure requires random observations to be generated from the sample data. A subsample size of K is randomly selected with replacements from the n sample plots. This creates the bootstrap sample. The desired statistic is estimated based on the bootstrap sample. This is repeated N times to obtain N estimates of the average clipped biomass (Smith and van Belle 1984). The variance of the mean is then computed as the variance among the N estimates of an average clipped biomass.

The average variance of the 2 estimation methods was computed by averaging the variance estimates associated with the classical, jackknife, and bootstrap procedures. In addition, the variance of the mean (i.e., simulation variance) was computed as the variance among the 10,000 estimates of average clipped biomass using the 2 double-sample estimators. By definition, the variance of the mean is the best estimate of variability and can be used to evaluate whether variance formulae provide unbiased estimates. A 95% confidence interval for each estimate with the respective appropriate standard error of estimate was computed, and the proportion of confidence intervals enclosing the true population mean was determined.

Results and Discussion

Basic statistics for the data are given in Table 1. Tests for equality of variances (F-test) indicated that estimated and clipped weights had similar variances, while the variance of oven-dry weights was significantly smaller ($\alpha=0.05$). The Anderson-Darling test statistic confirmed that all 3 biomass distributions (estimated, clipped, oven-dried) departed significantly from normality ($\alpha=0.05$) (Table 1). The relationship between clipped and visually estimated biomass was linear through the origin. However, the ratio of clipped to visually estimated biomass varies somewhat instead of being constant over the range of estimated biomass, indicating that either the ratio or regression estimates could be used for estimating average aboveground biomass.

Both ratio and regression estimators underestimated clipped biomass of blue grama. The regression estimator consistently had a larger bias than the ratio estimator (Table 2). However, the bias for

Table 2. Effects of sample size on percent bias of estimates of aboveground biomass of blue grama for double sampling with ratio and regression estimators.

Clipped sample size	Ratio	Regression
10	-2.7	-4.4
20	-3.2	-3.6
30	-2.7	-3.0
40	-2.4	-2.6
50	-1.9	-1.9

¹Population mean = 4.95 g • 0.25 m⁻².

the regression estimator decreased linearly as the clipped sample size increased, while the bias for the ratio estimator was non-linear over sample size.

Because both ratio and regression estimators were biased, one way to compare the variance of 2 estimators with different amounts of bias is to use the mean squared error (MSE), which is defined as the "variance + bias²" (Cochran 1977). Thus, an estimator with a smaller MSE is considered more precise than one with a larger MSE, even though the latter may have a smaller variance.

In terms of the MSE, the ratio estimator was always a better estimator than the regression estimator (Table 3). Both the jack-

Table 3. Mean squared error of the classical, jackknife (J), and bootstrap (B) variance estimates of aboveground biomass of blue grama.

Estimator	Clipped sample size				
	10	20	30	40	50
Ratio	0.923	0.573	0.442	0.368	0.311
Ratio-J	1.221	0.709	0.545	0.472	0.413
Ratio-B	1.085	0.705	0.524	0.467	0.410
Regression	1.038	0.598	0.447	0.379	0.314
Regression-J	2.025	0.771	0.566	0.485	0.418
Regression-B	1.335	0.756	0.542	0.480	0.414

knife and bootstrap estimates of the sample variance yielded larger MSE than the classical method for variance estimates. The jackknife and bootstrap estimates of the variance had a smaller MSE for the ratio method than for their regression counterparts.

Another method of evaluating the effect of bias on the statistical properties of the ratio and regression estimator is to examine the relative error, which is defined as the bias divided by the root mean squared error (bias/ $\sqrt{\text{MSE}}$). A large relative error has the effect of distorting confidence probabilities (Cochran 1977). For example, with a bias/ $\sqrt{\text{MSE}} = 0.20$, the actual confidence level associated with a nominal 95% confidence interval is about 0.9454.

The relative error was less than 0.15 in 5 out of 15 cases for the ratio estimator (Table 4), and less than 0.20 in 13 out of 15 cases (Table 4). The regression estimator had only 2 out of 15 variance

Table 4. Sample bias relative to the root mean square error of the classical, jackknife (J), and bootstrap (B) variance estimates of aboveground biomass of blue grama.

Estimator	Clipped sample size				
	10	20	30	40	50
Ratio	0.138	0.208	0.204	0.199	0.168
Ratio-J	0.120	0.187	0.184	0.176	0.146
Ratio-B	0.127	0.187	0.187	0.176	0.140
Regression	0.215	0.232	0.224	0.208	0.169
Regression-J	0.154	0.205	0.199	0.184	0.146
Regression-B	0.190	0.207	0.204	0.185	0.147

estimates with a relative error less than 0.15, and 8 out of 15 less than 0.20. The bias relative to the root mean square error also approximates a curvilinear relationship with respect to the number of clipped plots. In addition, the jackknife and bootstrap variance estimates had a smaller relative error across all sample sizes compared to the classical variance estimators (Table 4). The lower relative error results from variance overestimation by the jackknife and bootstrap estimates, thus reducing the effect of the bias on the width of the confidence interval. Similarly, the lower relative error associated with the ratio estimator results from the smaller bias and MSE, compared to the regression estimator.

Ratio of classical, jackknife, and bootstrap variance estimates to the simulation variance are given in Table 5. A ratio less than 1

Table 5. Ratio of mean variance of the classical, jackknife (J), and bootstrap (B) variance estimates to variance of the mean for the ratio and regression estimator.

Estimator	Clipped sample size				
	10	20	30	40	50
Ratio	0.674	0.982	0.916	0.936	1.202
Ratio-J	0.898	1.233	1.144	1.216	1.611
Ratio-B	0.796	1.224	1.097	1.210	1.601
Regression	0.624	0.991	0.896	1.015	1.164
Regression-J	1.265	1.303	1.154	1.320	1.569
Regression-B	0.817	1.277	1.103	1.307	1.553

indicates an underestimation of the variance, while a ratio greater than 1 indicates an overestimation. Using this as a guideline, one can see that the classical regression estimator provides the best estimate of the variance across all sample sizes tested. Next best estimates were provided by the classical ratio estimator. In particular, for sample sizes of $n \geq 20$, the jackknife and bootstrap variance estimate overestimated the true variance for both the ratio and regression estimators. On the other hand, for a sample size, $n = 10$, the jackknife ratio estimate provided the best estimate of the variance, while the bootstrap regression estimate provided the second best estimate.

Francis et al. (1979) also used simulation to compare sample variances of regression and ratio estimators to theoretical variances and found that the regression estimate provided the minimum variance. Their finding is the opposite of what we observed in our study. This difference is due to the relationship between clipped and estimated biomass. In our study, this relationship was linear through the origin, and the variance of clipped biomass tended to be proportional to estimated biomass. In their study, the relationship between clipped and estimated biomass was also linear, but not necessarily one that passed through the origin, which resulted in the regression estimator being more efficient than the ratio estimator (Cochran 1977). The previous authors also noted that if bias is present in the estimation process, the difference between the theoretical and sample variance is more pronounced when the ratio estimate is used than when the regression estimate is used.

Furthermore, these authors noted that when the variance of the weight estimate is constant over the range of clipped values, double sampling significantly reduces the variance of the estimate over that of clipping only if the cost is constant. In their study, for sample sizes $n \leq 30$, double sampling reduced the variance by a factor of 2 over that of clipping only, and by a factor of 1.5 for sample sizes $n > 30$.

Table 6. Relative coverage of population mean for 95% confidence limits for the classical, jackknife (J), and bootstrap (B) variance estimates of aboveground biomass of blue grama.

Estimator	Clipped sample size				
	10	20	30	40	50
Ratio	0.946	0.942	0.893	0.930	0.946
Ratio-J	0.946	0.969	0.955	0.953	0.986
Ratio-B	0.865	0.969	0.933	0.953	0.986
Regression	0.919	0.942	0.899	0.953	0.933
Regression-J	0.946	0.960	0.955	1.000	0.986
Regression-B	0.892	0.973	0.944	0.977	0.986

The actual coverage rates of nominal 95% confidence intervals for the ratio and regression estimator, using classical, jackknife, and bootstrap variance estimates, are given in Table 6. All variance estimates provided reasonable coverage rates, especially for small sample sizes ($n = 10$). The classical variance estimates consistently had coverage rates less than 0.95, while the jackknife estimator had coverage rates exceeding 0.95. The bootstrap estimate had the lowest coverage rate for both the ratio and regression estimator at $n = 10$.

Finally, it should be mentioned that the results of this study may not be applicable in stands in which blue grama is a minor component of the species mix. In such instances, the regression estimator may be more appropriate.

Recommendations

1. Fresh weights, instead of oven-dry weights, should be used with visually estimated weights in double sampling because of the similarities in their distribution. If needed, average fresh weights can be converted to oven-dry weights.
2. The ratio estimator should be used for estimating average clipped weight in stands dominated by blue grama in eastern Colorado in preference to the regression estimator.
3. For $n = 10$ clipped plots, the jackknife variance estimate is recommended for estimated confidence intervals. For $n \geq 20$ clipped plots, the classical variance estimates should be used to obtain reliable estimates of the population variance and to estimate confidence intervals.

Literature Cited

- Cochran, W.G. 1977. Sampling techniques. John Wiley & Sons, Inc., N.Y.
- Francis, R.C., G.M. Van Dyne, and B.K. Williams. 1979. An evaluation of weight estimation double sampling as a method of botanical analysis. *J. Environ. Manage.* 8:55-72.
- Pechanec, J.F., and G.D. Pickford. 1937. A weight estimate method for determination of range or pasture production. *J. Amer. Soc. Agron.* 29:894-904.
- Smith, E.P., and G. van Belle. 1984. Nonparametric estimation of species richness. *Biometrics* 40:119-129.
- Sokal, R.R., and F.J. Rohlf. 1981. Biometry. Freeman, San Francisco, p. 795.
- Stevens, M.A. 1974. EDF statistics for goodness of fit and some comparisons. *J. Amer. Sta. Assoc.* 69:730-737.
- Scheaffer, R.L., W. Mendenhall, and L. Ott. 1990. Elementary survey sampling. DWS-Keet Pub. Co., Boston, Mass.
- Schreuder, H.T., and Z. Ouyang. 1992. Optimal sampling strategies for weighted linear regression estimation. *Can. J. Forest. Res.* (In press).

Thin layer measurement of soil bulk density

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Abstract

Measurement of soil bulk densities is difficult if there are gravel, stones, or other materials present in the soil profile. A technique is offered for estimating the soil bulk density in thin layers (1.0 cm) in loose, nonuniform soils with low moisture levels. The technique consists of the removal of the soil in shallow layers. As each layer is removed, the hole is filled with a molten paraffin wax to obtain a casting of the excavated volume. Measured bulk densities values using this procedure compare well to results obtained with other techniques.

Key Words: infiltration, paraffin wax, soil texture

Bulk density is an important soil characteristic which affects many biological and physical processes, such as root penetrations and water infiltration. Soil bulk density of field sites is frequently estimated by driving a cylinder of known volume into the ground to obtain a soil core. Other techniques utilize a balloon or other flexible lining placed in the excavated hole and filled with a measured volume of water or dry sand. The removed soil is dried and weighed (Blake 1965). These techniques provide a measure of the average bulk density for a volume of soil several centimeters deep. The procedures are reasonably simple and easy, giving reproducible results in homogeneous soils if: (1) the soil water content is in the range where there is no change in soil core volume as a sampler is inserted; (2) the soil texture allows insertion of the sampler, i.e., no rocks or stones in the profile to hit the cutting edge of the sampler; and (3) the balloon or lining material does not dislodge soil particles when being inserted and conforms to the minor surface irregularities. These techniques are not satisfactory on many soils where there are gravel, stones, or other material in the profile or for sampling at the soil surface when the soil is dry and loose. A technique is described that allows the estimation of soil bulk density in thin layers (1.0 cm) in loose soils which may also have imbedded material such as gravel and rocks or is composed of other non-uniform material. The accuracy of the technique is compared to other bulk density techniques.

Methods and Materials

The technique involves the removal of the soil in shallow layers. As each layer is removed, the hole is filled to the original soil level of the layer with a molten paraffin wax. After the wax has solidified, the wax casting is removed and weighed. The volume of the wax is then calculated based on the wax specific density. The soil removed from the hole is dried and weighed.

Procedure

The soil in the sampling area is removed to the required depth

These studies were conducted in the summer of 1992. Shortly after acceptance of the paper for publication the authors became aware of the studies by Muller and Hamilton (1992) which uses an expanding urethane foam in a similar conceptual approach. We do not wish to detract from the earlier published studies and believe the techniques are complementary.

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and stored in a plastic bag for later drying and weighing. After the hole has been excavated, molten paraffin wax is carefully poured into the depression, filling to the original soil surface of the layer. It is critical that the temperature of the wax be as close to the solidifying temperature as possible. If the wax is too hot, pouring may disturb the soil particles resulting in incorporation of soil into the wax casting. Also, cooler wax will not flow into the pores of the surrounding soil, which can occur if the wax temperature is too hot. At the optimum temperature, the wax will begin to solidify and change from colorless to white immediately upon contact with the soil surface. The wax temperature can be checked by pouring a small amount into a test soil depression in the area. It is best to use wax with a melting point of 53° C or higher. On summer days when soil temperatures are hot, low melting point waxes are slow to solidify and the castings are soft and may deform when being removed from the soil or transported.

After the wax has solidified, the casting is removed from the hole. Soil particles clinging to the wax casting are rinsed off with cool water (18–22° C) and brushed away. Wax shrinks as it cools which leaves a depression in the top surface. Molten wax is poured into the depression in sufficient quantity to "bead" above the surface. After the wax has cooled, the excess wax above the sides of the original sample is removed using a sharp knife (Fig. 1). This can be done at a later time.

The volume of the wax casting is determined by dividing the weight of the wax casting by the density of the paraffin wax. The specific density for paraffin wax varies from 0.87 to 0.91 g/cm³ (Weast et al. 1965). The density of the wax used can easily be determined by filling a known volume container with molten wax. The specific density is the weight of the cooled wax per unit volume.

The soil removed from the hole is dried at 105° C for 24 hours and then weighed. The soil bulk density is the weight of the soil per unit volume of wax. The procedure is repeated as necessary for deeper layers in the soil profile.

Evaluation Procedure:

Laboratory—The procedure was initially compared to the cor-

Table 1. Comparison of bulk density measurements of laboratory prepared samples using wet 70 mesh silica sand with a known bulk density.

	Tray		
	1	2	3
	----- g/cm ³ -----		
Average bulk density of tray	1.50	1.32	1.44
Core	1.59	1.50	1.55
Wax—Subset A	1.18	1.18	1.20
B	1.45	1.28	1.33
C	1.32	1.23	1.31
Mean (wax)	1.32	1.23	1.28
Std. Dev. (wax)	.14	.05	.07



ORIGINAL
WAX
CASTING



WAX CASTING
WITH
BEAD



SHAVED
WAX
BEAD CASTING

Fig. 1. Sketch of wax castings.

Table 2. Comparison of bulk density measurements of laboratory prepared samples using dry 70 mesh silica sand with a known bulk density.

	Tray					
	1	2	3	4	5	6
	(g/cm ³)					
Average bulk density of tray	1.32	1.32	1.36	1.36	1.34	1.37
Wax—Subset A	1.46	1.46	1.26	1.59	1.28	1.17
B	1.35	1.21	1.25	1.21	1.17	1.12
C	1.31	1.28	1.42	1.20	1.19	1.29
D	1.23	1.31	1.41	1.17	1.33	1.28
Mean (wax)	1.34	1.32	1.34	1.29	1.24	1.22
Std. Dev. (wax)	0.10	0.11	0.09	0.20	0.08	0.08

ing method on 70 mesh silica sand dampened with sufficient water to make the sand particles cohesive. Small plastic trays, 12.5- × 13.5- × 3.5-cm were filled with silica sand and wetted with approximately 100 grams of water. The average bulk density of the sand in the tray was based on the total dry sand per tray. In each tray, the bulk density of the top 1 cm layer was measured at 3 locations using the paraffin wax technique. Also, a 4.9-cm diameter × 3.0-cm long core of wetted sand was extracted with a thin wall core sampler. The size of the trays limited the number of core samples that could be taken. Both sets of measurements were repeated in 3 separate trays.

The paraffin wax technique was then tested on dry silica sand that was not wetted. The bulk density of the top 1 cm layer was measured at 4 locations within each tray. These measurements were replicated in 6 separate trays. The absence of soil moisture made it impossible to use the core sampler.

Field—The technique was field evaluated on a test area at the Central Plains Experimental Range near Nunn, Colo. The area had previously been plowed and leveled into a bench terrace (6.5-m wide and 175-m long) in preparation for another project. The soil is an Ascalon series gravelly loamy sand that had been thoroughly mixed in the top 60-cm approximately 6 months prior to sampling.

Four areas, 3-m × 3-m, were randomly located on the terrace. Two of the areas were sampled on 1 day and the remaining 2 sampled 7 days later. The soil surface of the area was dry and hard. To facilitate sampling, the soil was sprinkled with sufficient tap water to wet to a 3-cm depth. A 4.9-cm diameter × 3.0-cm long core sample was taken with a thin wall sampler on the prewetted soil surface in each test area. Three adjacent locations were sampled using the wax technique. The wax sample areas are identified as small-wet, large-wet, and large-dry. Areas designated as wet were prewetted by spraying sufficient water on the soil surface to penetrate 2–3 cm. The “small” samples were 4-cm diameter and the “large” samples were 8-cm diameter. Samples were initially taken on the 0–1 cm depth range. After removal of 0–1 cm depth layer wax casting, a second measurement representing the 1–3 cm depth was taken using the bottom of the hole from the first sample as the top of the second layer. Only the surface 0–1 cm layer was sampled in the dry state.

Results and Discussion

Laboratory

The bulk density values determined by both the wax casting

Table 3. Comparison of bulk density measurements on a sandy loam field site.

Method	Size	Depth	Replicate			
			1	2	3	4
		cm	(g/cm ³)			
Core		0–3	1.24	1.32	1.28	1.23
Wax	Small ¹ Wet	0–1	1.30	1.10	1.28	1.05
	Large ² Wet	0–1	1.65	1.06	1.19	1.20
	Large Dry	0–1	0.77	1.10	1.31	1.01
Mean (wax)		0–1	1.24	1.09	1.26	1.09
Std. Dev. (wax)		0–1	0.44	0.02	0.06	0.10
Wax	Small Wet	1–3	1.36	1.22	1.14	1.64
	Large Wet	1–3	1.34	1.21	1.29	1.21
Mean (wax)		1–3	1.35	1.22	1.23	1.43
Std. Dev. (wax)		1–3	0.01	0.01	0.11	0.30

¹Small = 4 cm diameter samples

²Large = 8 cm diameter samples

technique and the thin wall core sampler on the prewetted sand were within 2 standard deviations of the average bulk density of the tray (Table 1). Core measured bulk densities were expected to be higher than the bulk densities determined using the wax method due to compaction of the sand from insertion of the coring tool. It is also possible that the surface layer had a lower bulk density than the "average" for the entire depth.

Bulk density measurements of dry sand were within 1 to 2 standard deviations of the mean bulk density of the tray (Table 2). Bulk density measurements with the wax technique were often lower than the average bulk density of the tray because of a tendency not to remove enough of the excess wax and possibly because the top 1 cm of the sand had a lower bulk density.

Field

As expected for field sites, all techniques resulted in variations in the measured bulk densities. Soil surface (0–1 cm) bulk densities with the wax technique were approximately within 1 to 2 standard deviations of the core sampling technique bulk densities (Table 3). It was expected that the wax measured bulk densities would be lower than the core measured bulk densities because the core measurements represented the 0–3 cm depth and the wax technique measured the upper layer which is normally less dense. In the 1–3 cm layer, the wax measured bulk densities were similar to the core sampled measurements. The variability in the measured bulk den-

sity values were similar to the variability reported in other studies (Fritton and Olson 1972).

Summary

The wax casting technique is an effective way of obtaining a measure of soil bulk density in loose and dry soils where other techniques are limited or ineffective. The wax technique is also suitable for obtaining bulk density values at micro locations within the soil profile. This is important in applications such as evaluating the effect of soil layering on water infiltrations or plant root penetration.

Literature Cited

- Blake, G.R. 1965.** Bulk density. p. 374–390. *In*: C.A. Black (ed.) Methods of soil analysis, Part 1, Physical and mineralogical properties including statistics of measurement and sampling. No. 9, Agronomy. Amer. Soc. of Agron., Madison, Wisc.
- Fritton, D.D., and G.W. Olson. 1972.** Bulk density of a fragipan soil in natural and disturbed profiles. *Soil Sci. Soc. Amer. Proc.* 36:686–688.
- Muller, Robert N., and Millinda E. Hamilton. 1992.** A simple, effective method for determining the bulk density of stoney soils. *Commun. Soil Sci. Plant Anal.* 34:313–319.
- Weast, Robert C., Samuel M. Selby, and Charles D. Hodgman. (eds.) 1965.** Handbook of Chemistry and Physics. 46th Ed. The Chemical Rubber Co., Cleveland, Ohio.

Book Reviews

Primary Productivity of Grass Ecosystems of the Tropics and Sub-Tropics. Edited by S.P. Long, M.B. Jones and M.J. Roberts. 1992. Chapman and Hall, London. 267 p. US\$85.00 hardbound. ISBN 0-412-41020.

This book arises from research conducted under the United Nations Environment Programme Project: The Primary Productivity and Photosynthesis of Semi-Natural Ecosystems of the Tropics and Sub-Tropics. Five regional centres, mainly tropical grasslands and a sub-tropical bamboo forest, were selected, where the dominant or co-dominant species are graminaceous, and a uniform approach was agreed on prior to the commencement of these studies. The methods developed between the centres during this project are discussed fully. The success of the project in promoting new and more detailed information on carbon flow, photosynthesis and productivity, and remote sensing in tropical grasslands is underlined by the many new findings and detailed data, now housed in a central data base at the University of Essex, presented in this book on the potential of the tropical grasslands as a major sink of CO₂. It was not the objective of this work to duplicate the data collected in the IBP but to give attention to factors which had received little attention in the previous studies such as below-ground biomass, rates of decomposition, and year to year variation in relation to climatic factors. One of the most valuable outcomes of these studies was the development of practical procedures for the measurement of vegetation dynamics above and below ground, including methods designed to allow direct estimation of the quantities of vegetation lost each month through mortality so that the flow of material could be directly assessed. The common use of this methodology in each study maximised comparability.

The study sites which reflected the diverse nature of grasslands were as follows: dry savanna grassland in the Nairobi National Park, Kenya; humid grassland in southern Thailand; saline grassland in Mexico and a grassland in the floodplain in the central Amazon; and a bamboo forest site in China, to provide baseline data. The bamboo site was included by the editors to illustrate the differences between relatively closely related species of grasses with different life forms. As the editors point out in their introduction, most of the knowledge of natural grassland systems in the tropics is based mainly on the IBP studies. However, at the majority of the IBP sites, production was estimated from summing positive increments in biomass or estimation of production from peak biomass. These methods may have led to a serious and variable underestimation of production and turnover in plant biomass related to the seasonality of growth and death in these communities. Generally, below-ground production was not considered and above-ground net primary production of tropical grasslands was estimated by plot analysis of live and standing material. The results presented in these studies indicate that the below-ground portion can contribute more than 1/2 of the total net primary production (about 65% in the saline grassland). Furthermore the studies did not take the rate of different cycles of growth and death in different species into account and there was no allowance for the possibility of negative production. Grossman (1982, Proc. Grassld. Soc. Sth. Afr. 17:76-78.) pointed out that as most harvest studies are confined to above-ground biomass and no account is taken of losses due to translocation of metabolites to unsampled below-ground organs or below-ground respiration, exudation, leaching and consumption by insects, the result of most harvest studies is an underestimation of net annual above-ground primary production. The studies

discussed in this book go some way to dealing with some but not all of these issues. Root exudation or pest attack as well as consumption by grazers have not been investigated. Consequently, the effect of grazing on primary production is not known even though it has played a major role in the development of many of the grassland areas. Unfortunately, none of the primary production studies which have been carried out in the dry savannas of southern Africa have been quoted in this book.

The information collected at each of the study sites has been presented in a logical, standardized format which is easy to follow and enables easy comparison. An index of abbreviations and symbols has also been provided. Summaries, in point form, at the end of each chapter were very useful. However, the standard of production of some of the figures was disappointing. In the chapter on the Kenyan savanna, several errors have crept in, such as figures 2(a) and (b) being transposed. The use of a single figure for a group of figures in some of the chapters was annoying and frustrating and necessitated having to page back and forth frequently. In some figures different units used for above- and below-ground components made comparisons difficult and the failure to use the same symbols in all the figures concerning *T. triandra* and *P. mezianum* also led to confusion. In some cases reduction of figures led to reduced clarity. Furthermore *Rhynchelytrum repens* was renamed *Melinis repens* (Willd) Zizka subsp. *repens* some time ago.

In the discussion on the use of remote sensing for determining grassland primary production it was pointed out that the presence of dead matter in canopies tends to reduce the estimates of biomass by increasing red reflectance. Therefore, any advances in remote sensing of light interception efficiency of canopies requires the development of methods of distinguishing light interception by live and dead matter. It was also pointed out that remote sensing of plant production requires long-term, intensive 'ground truth' validation studies such as those discussed in this book, to help elucidate some of these problems.

Overall results of the studies indicated that primary production is linked to rainfall with water stress proving to be a major factor affecting production. Recovery after fire depends on moisture availability and the rate of recovery differs for different species, as corroborated by several studies already carried out in the southern African semi-arid grasslands.

Because of the heterogeneous nature of the grasslands, it is difficult to make global estimates of primary production as extensive sampling is required over longer periods. The editors have discussed these difficulties clearly and constructively. However, perhaps scientists are chasing rainbows by attempting to allocate actual production figures to specific grasslands in view of the problems highlighted above. If the difficulties with remote sensing can be resolved, it may be possible to set up a range of production indices for grasslands under different environmental conditions. However, as the editors so rightly point out, these indices would still require ground verification. All researchers who need to determine primary productivity of specific areas should find this book helpful.—*Maureen M. Wolfson*, Roodeplaat Grassland Institute, Pvt Bag X05, Lynn East 0039, South Africa.

Forage Resources of China. By Shing Tsung (Peter) Hu, David B. Hannaway and Harold W. Youngberg. 1992. Center for Agricultural Publishing and Documentation (Pudoc), Wageningen, Netherlands. 327 p. US \$80.00 hardbound. ISBN 90-220-1063-5.

Range scientists and managers, forage agronomists, livestock specialists, plant ecologists, geographers, and China buffs will be interested in *Forage Resources in China*, a new ecological survey of the rangeland and pastures of China, including Tibet. The book fills a void in the English language literature on Chinese rangelands by assembling in one place the dispersed research of a wide array of researchers, both Chinese and international. The book will likely be a major resource for Chinese students of ecology and range and pasture management looking to study abroad.

The volume, organized into 11 chapters includes a brief but interesting introduction, a short chapter surveying all of the grazing lands of China, 7 chapters on important regional rangelands and grasslands, a brief Chapter 10 covering saline, alkaline and hydrophytic communities and a concluding chapter of discussion and recommendations. Scientific plant names are used throughout. A list of cited references follows each individual chapter. Following Chapter 11 is a bibliography, a number of entries in which date back to the early part of this century. Detailed indices of geographic names and scientific plant names follow the bibliography. A general index is lacking. A few but sufficient maps and figures are included.

Both recent and historical black-and-white photographs are numerous and welcome. The clarity of the reproductions, some apparently made from newsprint, is often disappointing, and is not highly correlated with the age of the photographs. The content of the photographs, however, does not disappoint, and covers not only plant communities but subjects of livestock production, cultural practices and geological scenery. Even with their sometimes poor clarity, the photographs add much to the book.

The text is simple in style and invigoratingly free of much of the weighty jargon characteristic of many recent writings in plant community ecology. Even the most purely ecological discussions rarely stray far from applied concerns such as livestock production, erosion control, or cultural practices. The reader is never put under the ether of phytosocial complexity and jargon. Instead, a thoroughly ecological but pleasingly pragmatic perspective is maintained throughout the book. Some may attach cultural significance. Others will feel nostalgia.

The greatest general impression of the book is of the magnificent natural resources of a country known more for its culture and people. Many of the lands discussed may not be unspoiled, and may not have always been well managed, but they remain magnificent, and deserve thoughtful management and careful preservation.—*David L. Scarneccchia*, Washington State University, Pullman, Washington.

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