

Vegetation differences in desert shrublands of western Utah's Pine Valley between 1933 and 1989

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Abstract

Changes in rangeland vegetation integrate the consequences of livestock grazing intensity and possible climate change, as well as other factors. Because vegetation changes tend to be very slow in dry environments, observational time scales that exceed a human generation are needed to separate real trends from year-to-year, weather-driven variability. An exhaustive literature search for valid starting points within the Intermountain region revealed a unique quantitative study that was more than 50 years old. In 1933, vegetation along a 37-km transect in southern Pine Valley, Utah, was read from 19-m² plots located every 42 m. The only intentional, local management treatment in the interim has been moderation of domestic livestock grazing pressure. During a period climatically and phenologically similar to the original study, we re-examined representative segments of this transect by a more detailed updating of the original "square-foot-density" method. We found that vegetation type boundaries and ecotones were little changed after 56 years. However, canopy cover was dramatically greater in 1989, in some cases by more than tenfold for several perennial grasses, and less so for shrubs. Substantially greater understory cover as a relative proportion of total plant cover occurred in 1989 in all vegetation types examined. Some of the observed positive shifts of dominance/diversity are contrary to widely accepted expectations in the literature.

Key Words: *Artemisia tridentata*, sagebrush, *Chrysothamnus viscidiflorus*, rabbitbrush, *Grayia spinosa*, spiny hopsage, *Ceratoides lanata*, winterfat, *Oryzopsis hymenoides*, Indian ricegrass, *Hilaria jamesii*, galleta, livestock grazing, canopy cover

Qualitative evidence that unrestricted domestic livestock grazing had caused seriously destructive impacts on the vegetation cover of federal lands in the western United States led to the passage of the Taylor Grazing Act in 1934. Substantial reductions in domestic grazing pressure followed. Land management agency data, generalized observation, and qualitative photographic comparisons, along with extrapolations from the results of small-scale experimentation, have caused range managers to believe that improvement at the ecosystem level has occurred on many large tracts of public land since 1934, where moderation of grazing use has been the sole known treatment (Box 1990). Broad improve-

ments, however, have not been substantiated by rigorously scientific proof within the Intermountain West (Box 1990).

Analyses of Western U.S. rangelands are complicated by a high annual variability in plant production among species. Sharp et al. (1990), for example, qualitatively documented that in 1 southern Idaho valley the overall vegetation's appearance can be so different as to be almost unrecognizable from 1 year to the next, even for observations made on the same month and day during each year. This situation is largely a result of short-term fluctuations in temperature and precipitation, consequently variable inter-annual phenology, and removal of plant growth by livestock and other herbivores. Increasing the number and frequency of measurements seems to be the only long-term cure for these underlying data problems. On the other hand, a careful choice in the timing of field observations in relation to climate, phenology, and animal utilization appears to be one practical compromise between the reality of shortfalls in funding for monitoring and doing nothing at all.

Critically, data from any particular set of sites can provide only an example, not a definitive conclusion, for what may be occurring over wider areas. Results cannot be extrapolated, without further confirmation, even to neighboring tracts. Accordingly, they cannot be held per se as representative of the overall situation on federally managed lands. Further, determination of precise cause/effect relationships, including those resulting from livestock management decisions, is almost certainly beyond the scope of observational science, because there are so many uncontrollable variables.

Within these limits, the primary reason for the lack of scientific evidence for whether or not improvements have occurred (at scales which make them applicable to management) is the dearth of reliable and accurately repeatable baseline measurements made before 1934.

One older published effort (Stewart et al. 1940) proved to stand alone in its potential for effective repetition. Concern about apparent desertification on southwestern Utah's public lands had prompted a small group to survey the status of the vegetation in 2 neighboring large valleys in 1933. They did so in notable detail. The project's leaders were assisted by an influx of labor made suddenly available through the newly commissioned Civilian Conservation Corps.

One of their major efforts involved a single 40-km transect of 920 regularly spaced 19-m² plots across southern Pine Valley (Stewart et al. 1940). They estimated percentage of canopy cover by all plants (by species), and counted the absolute number of living, dead, and seedling shrubs. Subsequent administrators preserved a careful contour map that the field crew drew of the actual transect line as it was carried out in the field, including frequent references to fixed survey markers as well as topographic details. This map was of critical importance to repetition of the original study. A legible and complete copy of the original plot-by-plot field data

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sheets also survived for reuse through the intervening 55 years.

In our repeat study during 1989, potential sample size was limited by a labor pool one-fifth the size available during the original effort. Accordingly, we selected 250 plots in the major vegetation types and range sites represented. Our goal was to establish the magnitude and direction of vegetation change in these types over the intervening years. To avoid phenological or methodological differences in the expression of cover, we resurveyed the selected plots at the same time of year, and in essentially the same fashion, as was done in 1933. Similarity of climate is, of course, an important factor in vegetation examinations that seek information about other impacts. We were encouraged to initiate this repetition because preceding three-year drought conditions appeared to be similar to conditions during the original study (Table 1).

Table 1. US Weather Service mean precipitation values (\pm standard error of the mean) for Milford, Ut., with crop-year and shorter summaries as per Sneva and Britton (1983).

Year	Average precipitation for period		
	Oct-Sept	Oct-June	Jul-Sept
A: Large blocks of years			
	----- (mm) -----		
1908-1932	219 \pm 9	169 \pm 8	51 \pm 5
1933-1963	*202 \pm 9	155 \pm 9	*46 \pm 4
1964-1988	*257 \pm 12	187 \pm 10	*70 \pm 7
B: Three years immediately preceding studies			
	----- (mm) -----		
1930-1932	210 \pm 40	150 \pm 20	60 \pm 30
1986-1988	250 \pm 30	190 \pm 40	60 \pm 10

* $P < .05$ for similarity of indicated values within a column.

Study Site

Vegetation in the lower, southern part of Pine Valley is desert shrubland. Dominance along the transect line at the lowest points is by low rabbitbrush [*Chrysothamnus viscidiflorus* (Hook.) Nutt.] and winterfat [*Ceratoides lanata* (Pursh) J.T. Howell]. With rising elevation, dominance shifts to spiny hopsage [*Grayia spinosa* (Hook.) Moq. in DC.]. Above this, there is a belt of big sagebrush [*Artemisia tridentata* Nutt. spp. *tridentata*] and black sagebrush [*Artemisia nova* A. Nels.]. Each segment includes a mixed understory of grasses and forbs. [Nomenclature according to Goodrich (1976); growth form assignments for the USDA Soil Conservation Service (Anon., 1988).]

Lack of permanent water kept domestic livestock from grazing Pine Valley until about 1890, when the development of wells brought the first cattle. They were followed in about 10 years by the introduction of sheep, at high intensities of year-round use. "Seriously destructive" was the phrase of Stewart et al. (1940) used for early grazing practices. Consequent erosion was moderate to severe, and in places exceeded 15 cm, with some dune development (Murdoch and Welsh 1971).

Substantial, but undocumented, reductions in stocking rates, with restriction of grazing to winter-only at lower elevations, occurred in response to inadequate forage in the 1930's (S. Hansen and W. DiAge, Bureau of Land Management (BLM), Cedar City, Utah, personal communication). According to this same source, a further 33% reduction occurred as a part of the adjudication agreements which coincided with fence construction in the transect area in 1956. An additional 25% reduction in allotted livestock pressure occurred in 1983 for those portions converted from winter sheep to winter cattle use.

A free-ranging herd of feral horses was eliminated from the general area in the 1940's (V. Wood, permittee, personal communi-

cation). An examination of yearly grazing permits disclosed evidence that the sagebrush-dominated portions of the transect line may have been completely rested from sheep grazing during the period of 1967 to 1977. Those records are internally inconsistent, however, and so must be accepted with caution.

In 1958, the average Pine Valley stocking rate was estimated overall to be 3 ha/AUM (Murdoch and Welsh 1971). Until recently, fences have been several kilometers apart, and developed water similarly distant. More intensive fence building is currently in progress. Current management is described as "rest-rotation" (S. Hansen and W. DiAge, personal communication).

Much of the state of Utah received historically high levels of

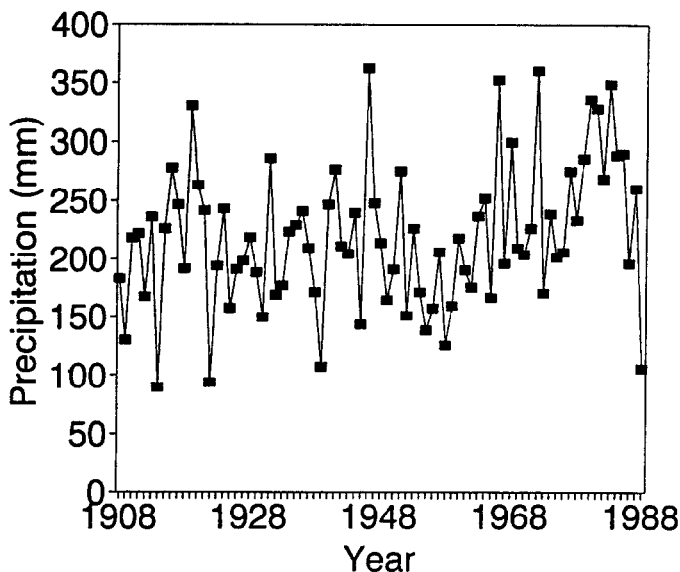


Fig. 1. Annual precipitation totals for Milford, Ut., measured on an October to September basis, 1908-1989.

rainfall during the early 1980's. Such, if present on the study area, could have had a confounding impact on vegetation response over time (Laycock 1991). Because of the highly local variability in weather during the interstudy period, this concern cannot be addressed absolutely. However, as the following will show, the best data available indicate that the likelihood of climatic impacts on this study may be particularly low.

Milford, Utah, is the nearest continuously recording U.S. weather station to Pine Valley which has a similar geographical character (i.e., located in a north-south valley). Figure 1 depicts the extant precipitation records for Milford, 65 km to the east of the center of our transect. A general upward temperature trend would parallel that found by Bradley et al. (1987) for the United States as a whole, in response to increasing atmospheric greenhouse gas concentrations. A possible upward trend in precipitation may be seen during the past 30 years, but which seems to have receded before our restudy took place (Fig. 1).

Table 1 breaks out equivalently large blocks of years, whose duration equals the period where data were available before the 1933 study. Also detailed is a block for the 3 years immediately preceding each study. From the data in Table 1A, it can be seen that an increase in both yearly and summer rainfall has occurred over the most recent 25 years in Milford, which may be statistically significant. In contrast, Table 1B indicates a smaller, and nonsignificant, spread between the 3 years immediately preceding the vegetation surveys.

Because the center of the actual study area is so far from Milford, and because rainfall can be very strongly affected by topography and distance in the Intermountain West, all precipitation data

available through the Utah State Climatologist's office for the Desert Experimental Range in Pine Valley are compared with those for Milford in Table 2. Over comparable periods, precipitation at the Desert Experimental Range was lower than might be expected from Milford's records. The most recent block of time, which parallels the increase found among the more complete records at Milford, shows that the difference in precipitation was much smaller, and not statistically significant, within Pine Valley

Table 2. US Weather Service mean precipitation values (\pm standard error of the mean) for Milford, Ut., compared with those for the Desert Experimental Range.

Year	Average precipitation for period		
	Oct-Sept	Oct-June	Jul-Sept
A: Milford			
	----- (mm) -----		
1950-1966	199 \pm 14	154 \pm 10	45 \pm 7
1967-1983	257 \pm 15	185 \pm 13	73 \pm 9
B: Desert Experimental Range			
	----- (mm) -----		
1950-1966	149 \pm 13	96 \pm 8	52 \pm 8
1967-1983	170 \pm 12	102 \pm 9	68 \pm 8

itself. This should indicate that in Pine Valley rainfall trends, if present, seems to have had less magnitude, and should therefore be less likely to have had an impact on vegetation, than elsewhere in western Utah.

Methods

Establishing the Transect

Location of plots in the 1933 study was done by visual sighting of the chosen end points of the transect line, supplemented as necessary by a hand-held compass (J. Pechanec, personal communication). The distance between regularly spaced plot centers along the line was established with a steel tape, since even spacing was the method used to provide an unbiased sampling.

To relocate the transect in 1989, we drew the theoretical northeast-to-southwest line on U.S. Geological Survey 7-1/2 minute quadrangles, connecting the original starting point of Sol's (or Red) Knoll, which is next to Utah State Highway 21, and Indian Peak, respectively. As an overlay, we copied unpublished BLM delineations of range site boundaries and vegetation categories, which the Beaver River Resource Area office had mapped in 1978-81. Along the transect line, we marked the specific plot numbers whose placement could be established from intermediate topographical details or from references on the 1933 contour maps, noting the relationship of particular plots to 1913 cadastral survey posts. From these known points and from the 42 m (132 feet) center-to-center distance between plots, we established the rest of the plot locations on our 1989 maps.

Groups of 50 or more plots that fell within a single BLM range site classification and within a single fenced grazing allotment were marked for restudy. This group size was chosen to minimize observational effects of slight inaccuracies in plot relocation (West and Hatton 1990).

The selected plots were subsequently located by distance and compass bearing from the 1913 survey markers that were still in place; by sighting along the measurement tape to the transect endpoints; and by comparing visible contours and roads to both the new and the original maps. The dates when the 1933 researchers measured individual plots were taken from their original data sheets. The 1933 field workers progressed from northeast to southwest along the transect line, and we followed their progress as

closely as possible by date.

Vegetation Measurement

Following Stewart and Hutchings (1936) and Stewart et al. (1940), we employed a central stake as a pivot on each plot and used a nonstretching rope with a 50-cm section of steel rod at its other end to mark on the ground a circle enclosing 19 m² (200 ft²). The "square-foot-density" method of the 1933 study depended upon estimating cover (to the nearest 0.023 m², i.e., one-quarter of a square foot) for each plant species. That cover, in turn, was defined as the area where the ground "could not be seen by an observer standing directly over it" (Stewart et al. 1940). Any individual plant that did not have such complete coverage (like most in practice) was to be visually compressed until it did so. Repeated circuits of the plot circle by an observer were made for each plant species. A "click counter" was used to enter quarter-of-a-square-foot units each time observed cover for individual or groups of small plants reached that threshold. The consequent sums were recorded at the end of each species-specific circuit (J. Pechanec, personal communication). Results from the 19 m² circle were then divided by 2 to give "density percent," which is now called percent cover (Bonham 1989).

Presence of a species that covered less than one-quarter square foot in total area was recorded as "trace" in 1933. These "trace" amounts were converted to an area of .0058 m² for our computerized analyses, since that is halfway between 0 and the smallest estimate that would round up to one-quarter of a square foot.

Numbers of dead, living, and seedling shrubs or trees are direct counts of these plants that had basal stem, or stem cluster, centered within the circle. The "seedling" classification was an arbitrary choice made by the observer in 1933 (J. Pechanec, personal communication). In the 1989 reexamination, a seedling was defined as a shrub whose canopy cover area was less than 5 cm in diameter, or a tree having a canopy diameter of less than 15 cm.

For better future repeatability, and to permit more types of analysis than were possible with the original method, we did not just estimate total cover for each species in 1989. Instead, we measured the diameter of the top of each plant's canopy with a meter stick. This measure was intended to be equal to the diameter of a cylindrical complete shadow cast straight down by the plant. Each plant was compressed by hand, if necessary, to estimate its complete shadow diameter. This variation on the original technique (i.e., recording individual diameters) took very little additional time when compared to simply estimating cover. Diameter classes were listed in our field notebook under species headings. Additional plants of the same size were quickly added as tick-marks alongside the original entry. Transfer of these cryptic data to a computer spread-sheet allowed for rapid calculation of combinations and totals.

From these more direct measurements, frequency, density, and cover may be determined for any desired class of plants. Our revised technique also aids in providing data to more accurately interpret dominance/diversity relationships, especially for minor species. For shrubs where only part of the plant was living, we estimated and recorded that percentage as well.

However, for mat-forming and rhizomatous species (whose individual plant boundaries are ambiguous), we continued to use visual estimates for area. Thin, wispy plants such as dried cheat-grass (*Bromus tectorum* L.), especially when widely, but densely, scattered, remains a particularly difficult-to-estimate cover problem.

Plot edges were visualized as a cylinder extending upward from the circular boundary as it was marked on the ground. Only those portions of plants whose shadow fell inside this imaginary cylinder, if illumination was at the zenith, were recorded. Plants whose basal stem center was outside this boundary, or which had more than

half their canopy cover extending beyond the cylinder, were placed into separate categories in the detailed records. This information could reduce skewing in any future plant size analyses.

All original data, photographs, and a manuscript having more methodological detail will be archived in Special Collections at Utah State University's Merrill Library.

Plant Identification Accuracy

Since the 1989 study's timing followed the mid-to-late summer schedule of the original one, the post-grazing expression of cover left few seed heads present on the grasses. Some misidentification, especially in heavily grazed patches, was inevitable. Many ephemeral forbs will have been missed in both the original and repeat studies. Extended recent drought (Fig. 1) and a consequent lack of flowering made separations especially difficult between broom snakeweed [*Gutierrezia sarothrae* (Pursh) Britt. & Rusby] and low rabbitbrush, and among these and young specimens of rubber rabbitbrush [*Chrysothamnus nauseosus* (Pallas) Britt.]. The same was true for the Mormon teas, *Ephedra nevadensis* Wats. and *Ephedra viridis* Cov., and for species separations among *Eriogonum* (Michx.) spp.. Further, on many sites, hybridization appears to be occurring between black sagebrush and big sagebrush, since many individuals seem to be intermediate between the characteristics of the more obviously differentiated specimens. Rabbitbrush, as *Chrysothamnus* spp. in Table 3, were larger in stature than those listed as *C. viscidiflorus*. Therefore, percentages and changes within these taxa must be interpreted with special caution.

The above considerations have resulted in condensations which produced minor differences between tables and figures. However, growth-form groupings aid in getting past practical errors in identification at species levels. They also have considerable utility for other reasons, not least in gaining a quick overall picture of on-site condition, and of an area's potential practical utility (Friedel et al. 1987).

Sampling Error

Five percent of the 1989 plots were re-read by the same observer, and additionally by 2 different observers, with sufficient time between such readings to dissipate mental carryover. Density (i.e., the absolute number of plants per unit area) varied at most by 10%, and was typically less than 5%, both for a single observer's repetition and between observers. Replicate readings of species cover also fell within this same quite satisfactory pattern, when made by a single observer.

However, cover estimates for the same plot, when made by different observers, varied much more widely. This is as might be expected from what seems to be an unavoidably subjective compression of plants to derive continuous shadows (West and Hatton 1990). This variation in apparent cover reached as much as 50% for more openly structured species. However, each observer did tend to have a consistent bias.

Cover estimation can be further subject to confusion because of the effects of leaf and stem utilization during grazing. Removal of early growth among canopies by grazing animals was missed in both 1933 and 1989. Any individual plant which had been heavily grazed, and accordingly down-rated in recorded size and cover, may also have lost competitive advantage from that heavy use. What we measured was essentially basal cover for the bunchgrasses, as was likely to have also been the case in 1933. Additionally, drought kept rhizomatous grasses and turf-formers from leafing out in 1989. Their cover as reported here may be taken as effectively basal.

Calculations

We have generally adopted the convention of Wheeler et al. (1989) for rounding all calculations, or have been even more con-

servative in its expressed accuracy.

The "trace" category from the 1933 data creates some difficulties in interpreting dominance/diversity because it produces what might best be described as an "artifact floor" for the relative cover of the rarest plants. The value assigned to it is an arbitrary logical choice, as described above. Our direct measurement of plant size in 1989 allows a more accurate reading of dominance/diversity for the less common taxa.

An observational study such as this one has a great many potentially confounding environmental and biological interactions (Hairston 1989). Accordingly, we have chosen a conservative statistical approach to highlight changes which are most likely to be of special interest. "Significant" is defined herein as a difference between 2 means that is greater than the sum of twice the standard error for each of those means (or 4 times that of 1 when the other is 0), which gives $P < .05$ that the difference in question would have resulted from random chance.

Other Considerations

Shadscale [*Atriplex confertifolia* (Torr. & Frem.) Wats.] is a dominant species in some portions of Pine Valley. However, the overall transect is only tangential to the locations where the plant is characteristic. The portion of the transect where shadscale is found is coincidentally at the point where the transect line crosses the state highway. This area is also now paralleled by a secondary road (and so received heavy vehicular disturbance), and has been used as a sheep bedding ground. Because of these localized impacts, which were visibly unrepresentative for the wider valley, we did not repeat measurements on that section. Although repeat readings of pinyon-juniper plots were done, they are, in the interests of brevity, to be reported elsewhere.

Results

A manyfold greater total live plant cover was found in 1989 compared to that in 1933 in all 3 of the vegetation types (Table 4). These differences are statistically significant, and are of such magnitude that they would seem to far exceed even the several previously noted sources of possible error. Furthermore, relative variability (as a percentage of absolute cover) was less in 1989. This should correlate with a more spatially consistent pattern for the increase in overall cover. Physical size of the dominant species varies with elevation, following the probable concomitant rise in average precipitation at higher altitudes. This will, by itself, bring about a likelihood of increased variability among plots at higher elevations.

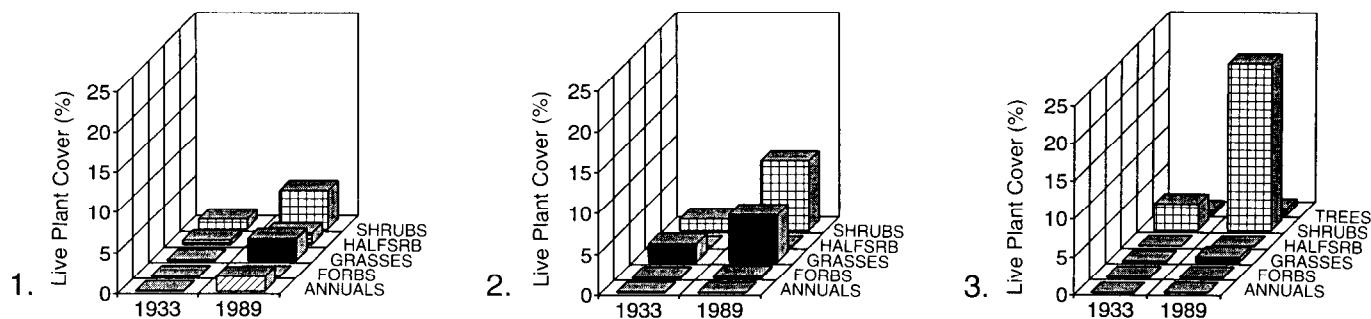
The Low Rabbitbrush Type

Figure 2-A1 gives a comparison of live plant cover, by growth form, between years. Table 3 includes details of differences in the absolute magnitude at the species level. A significant, fourfold difference in shrub cover is apparent. This was confined almost entirely to low rabbitbrush. Broom snakeweed is also more abundant in 1989, in this case threefold over the 1933 value.

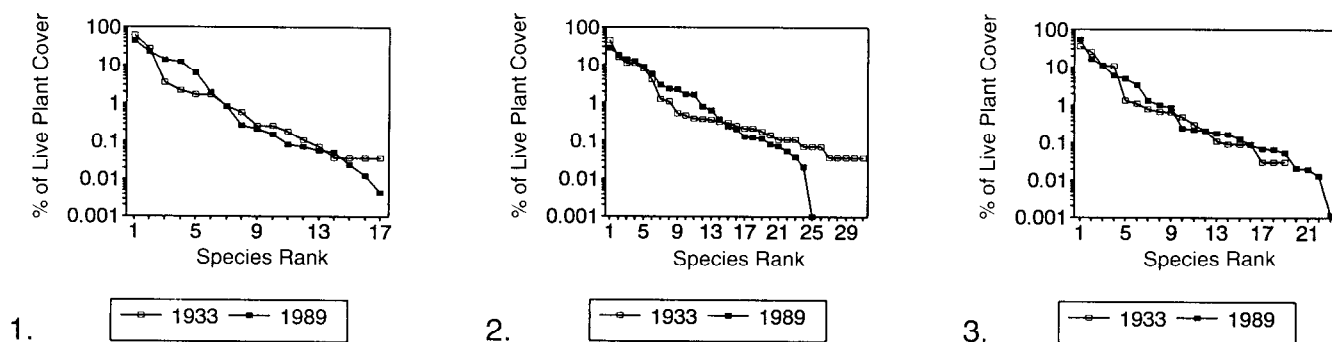
The most dramatic difference in live plant cover is found in the perennial grass category (Fig. 2-A1). This comes primarily from a twenty-fold greater amount of Indian ricegrass [*Oryzopsis hymenoides* (R. & S.) Ricker in Piper], although a second species [*Hilaria jamesii* (Torr.) Benth.] was also ten-fold greater in the low rabbitbrush vegetation type during 1989 (Table 3).

The relatively dry preceding seasons, and summer timing for data collection, in both years might be expected to reduce the cover of moisture-sensitive forbs and annuals most strongly. Nevertheless, notable differences in introduced annuals, cheatgrass, and halogeton [*Halogeton glomeratus* (Bieb.) Mey. in Ledeb.] were

A. Live plant cover, by growth form



B. Dominance/Diversity



C. Density

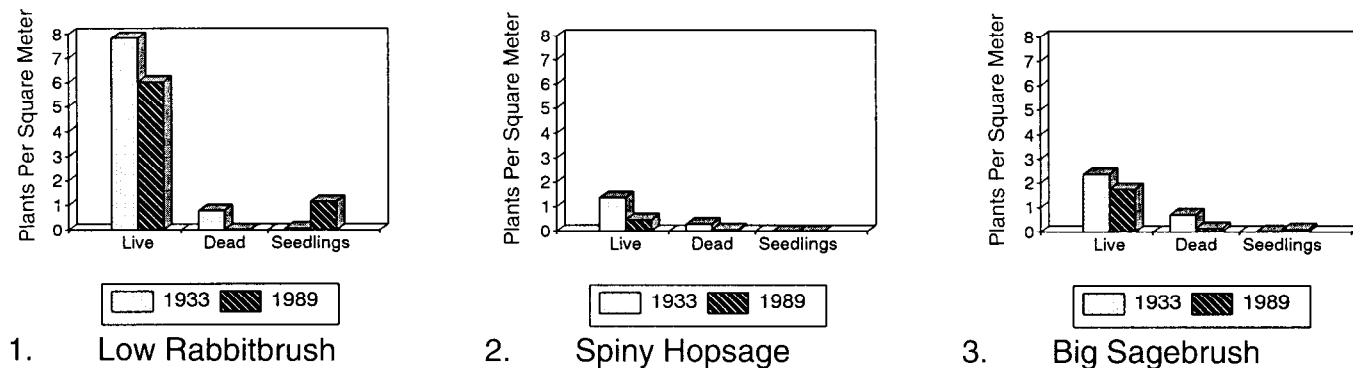


Fig. 2. Absolute cover among growth forms (A), dominance/diversity among species (B), and average shrub density (C) for rabbitbrush (1) hopsage (2), and sagebrush (3) dominated community types in Pine Valley, Ut., 1933 and 1989.

detected (Table 3), although their variability kept these differences from being statistically significant.

By examining the results of each plot displayed by position along the transect line (Fig. 3), one can readily see that these invading annuals are confined almost entirely to less than one-fifth of the locations. Most of the total comes from just 4 plots. These concentrations tend to be circular in shape and therefore appear to be the location of old anthills. Some of these may also be associated with former campsites of shepherds or other human visitors, since we tended to observe relatively high concentrations of rusted cans and older glass bottles on and near them.

The format of Figure 3 dramatizes spatial data concentrations that otherwise might be missed with simple averaging of plots within types. The difference between the highly localized appearance of the annuals' and the perennials' more consistent increase

across space becomes especially clear by comparing Figure 3 with Figure 4.

Table 3 shows the absolute magnitude of species was different between years while Figure 2-B1 illustrates that the relative number of species that have importance has also increased. Westman (1978) considers this to be one indicator of possible community "health", its potential for resilience. Five species (*Ceratoides lanata*, *Chrysothamnus viscidiflorus*, *Oryzopsis hymenoides*, *Bromus tectorum*, and *Hilaria jamesii*), instead of just 2 in 1933, now play major roles in overall cover during 1989.

Density of living shrubs (number per unit area) of all species in 1989 was not much different from 1933 (Fig. 2-C1). However, since cover increased in 1989, this result suggests that in 1933 individual plants, while somewhat more numerous, were smaller. This finding correlates with the observations by Stewart et al. (1940) of severe

Table 3. Pine Valley species comparison by vegetation type, by year (live plants, percent cover).

Growth Form Taxa	Low Rabbitbrush		Spiny Hopsage		Sagebrush	
	1933	1989	1933	1989	1933	1989
----- % -----						
Trees:						
<i>Juniperus osteosperma</i>					0.5 ± 0.5	0.8 ± 0.4
Shrubs:						
<i>Chrysothamnus nauseosus</i>	0.005 ± 0.004	0.02 ± 0.02				
<i>Artemisia spinescens</i>	T	0	0.007 ± 0.003	0		
<i>Atriplex canescens</i>	0.03 ± 0.02	0.09 ± 0.05	0	0.04 ± 0.03		
<i>Atriplex nuttallii</i>	0.011 ± 0.008	0	0.001 ± 0.001	0		
<i>Chrysothamnus viscidiflorus</i>	1.17 ± 0.04	*4.6 ± 0.3	0.61 ± 0.08	*1.8 ± 0.2	1.2 ± 0.1	*2.6 ± 0.4
<i>Ephedra nevadensis</i>	T	0	0.014 ± 0.004	0.4 ± 0.1	0.049 ± 0.005	*0.20 ± 0.07
<i>Opuntia</i> spp.	0	0.008 ± 0.004	0.008 ± 0.003	0.019 ± 0.007	0.010 ± 0.003	*0.04 ± 0.01
<i>Grayia spinosa</i>			0.43 ± 0.05	*2.5 ± 0.4	0.023 ± 0.008	*1.4 ± 0.5
<i>Artemisia tridentata</i>			0	*2.0 ± 0.4	0.5 ± 0.2	*13. ± 1
<i>Chrysothamnus</i> spp.			0.012 ± 0.006	*1.3 ± 0.3	0	*1.2 ± 0.3
<i>Ephedra viridis</i>			0	0.005 ± 0.005	0	0.02 ± 0.2
<i>Echinocereus</i> spp.			0	0.0001 ± 0.0001	0	0.0003 ± 0.0002
<i>Artemisia nova</i>					1.7 ± 0.2	*4.0 ± 0.6
Half Shrubs:						
<i>Ceratoides lanata</i>	0.51 ± 0.03	*1.5 ± 0.08	0.004 ± 0.002	0.010 ± 0.007		
<i>Gutierrezia sarothrae</i>	0.002 ± 0.001	0.001 ± 0.0003			0.005 ± 0.005	0.016 ± 0.008
<i>Eriogonum microthecum</i>			0.003 ± 0.002	0		
<i>Leptodactylon caespitosum</i>			0.006 ± 0.003	0		
Perennial Grasses:						
<i>Sporobolus</i> spp.	0.005 ± 0.002	0.006 ± 0.005	0.001 ± 0.001	0		
<i>Hilaria jamesii</i>	0.016 ± 0.008	*0.21 ± 0.09	0.41 ± 0.08	0.24 ± 0.06	0.004 ± 0.002	*0.05 ± 0.02
<i>Oryzopsis hymenoides</i>	0.041 ± 0.005	*2.4 ± 0.2	0.042 ± 0.005	*0.33 ± 0.04	0.029 ± 0.005	*0.24 ± 0.04
<i>Sitanion hystrix</i>	0.001 ± 0.001	0.003 ± 0.001	0.032 ± 0.009	*0.24 ± 0.06	0.035 ± 0.005	*0.31 ± 0.03
<i>Aristida</i> spp.	0	0.02 ± 0.01	0.02 ± 0.01	0.12 ± 0.04	0	0.004 ± 0.003
<i>Stipa comata</i>			0.15 ± 0.006	*0.8 ± 0.2	0	0.02 ± 0.01
<i>Bouteloua gracilis</i>			1.7 ± 0.2	*4.0 ± 0.6		
<i>Poa</i> spp.			0.001 ± 0.001	0		
Annual Grasses:						
<i>Bromus tectorum</i>	0	1.3 ± 0.9	0.16 ± 0.08	0.03 ± 0.01	0	0.05 ± 0.03
Perennial Forbs:						
<i>Machaeranthera canescens</i>	0	0.006 ± 0.002				
<i>Aster adscendens</i>	0.004 ± 0.002	0	0.001 ± 0.001	0		
<i>Eriogonum</i> spp.	0	0.009 ± 0.007	0.007 ± 0.002	*0.05 ± 0.02	0.001 ± 0.001	0
<i>Sphaeralcea grossulariifolia</i>	0.032 ± 0.04	*0.0014 ± 0.0008	0.049 ± 0.005	*0.012 ± 0.004	0.031 ± 0.005	0.03 ± 0.01
<i>Astragalus</i> spp.			0.015 ± 0.004	0.008 ± 0.003	0.060 ± 0.007	0.04 ± 0.01
<i>Brickellia oblongifolia</i>			0	0.02 ± 0.02		
<i>Laphamia</i> spp.			0.004 ± 0.002	0		
<i>Cryptantha</i> spp.			0.008 ± 0.003	0.09 ± 0.05		
<i>Euphorbia fendleri</i>			0.010 ± 0.003	0.003 ± 0.002	0.004 ± 0.02	0
<i>Penstemon</i> spp.			0	*0.018 ± 0.007	0.004 ± 0.002	0.005 ± 0.003
<i>Phlox</i> spp.					0	0.0009 ± 0.0006
Annual Forbs:						
<i>Iva axillaris</i>	T	0				
<i>Salsola kali</i>	0.07 ± 0.02	0.03 ± 0.02	0.001 ± 0.001	0		
<i>Halogeton glomeratus</i>	0	0.7 ± 0.3				
"Fern weed"					0.001 ± 0.001	
"Red weed"					0.01 ± 0.01	
<i>Gilia</i> spp.					0.001 ± 0.001	0

* $P < .05$ for similarity of 1989 value to that for 1933, within a vegetation type.

shrub hedging by livestock in 1933. Of related importance in Figure 2-C1 is the greater number of dead plants in 1933 and the greater number of seedlings in 1989, which should also correlate with an improving overall community health.

The Spiny Hopsage Type

By comparison of dominance/diversity in Figure 2-B2 to that of Figure 2-B1, one can observe that the spiny hopsage vegetation type had a somewhat more balanced relationship among the most dominant species present in 1933 than was apparent in the low rabbitbrush area during that year. Between 1933 and 1989, the

greatest difference in the low rabbitbrush type occurred in the most dominant portion of species cover (Fig. 2-B1). For the spiny hopsage area, however, it is the middle range of cover that exhibits the most noticeable (and positive) difference (Fig. 2-B2).

An indicated difference in total species number from 31 species in 1933 to 25 in 1989 (Fig. 2-B2) is partly due to a more complete splitting out of buckwheat species by the 1933 observers, along with the likely problem of identification among the grasses that was mentioned in the methods section. The "artifact floor" created as a result of converting 1933 trace cover to a single number is also

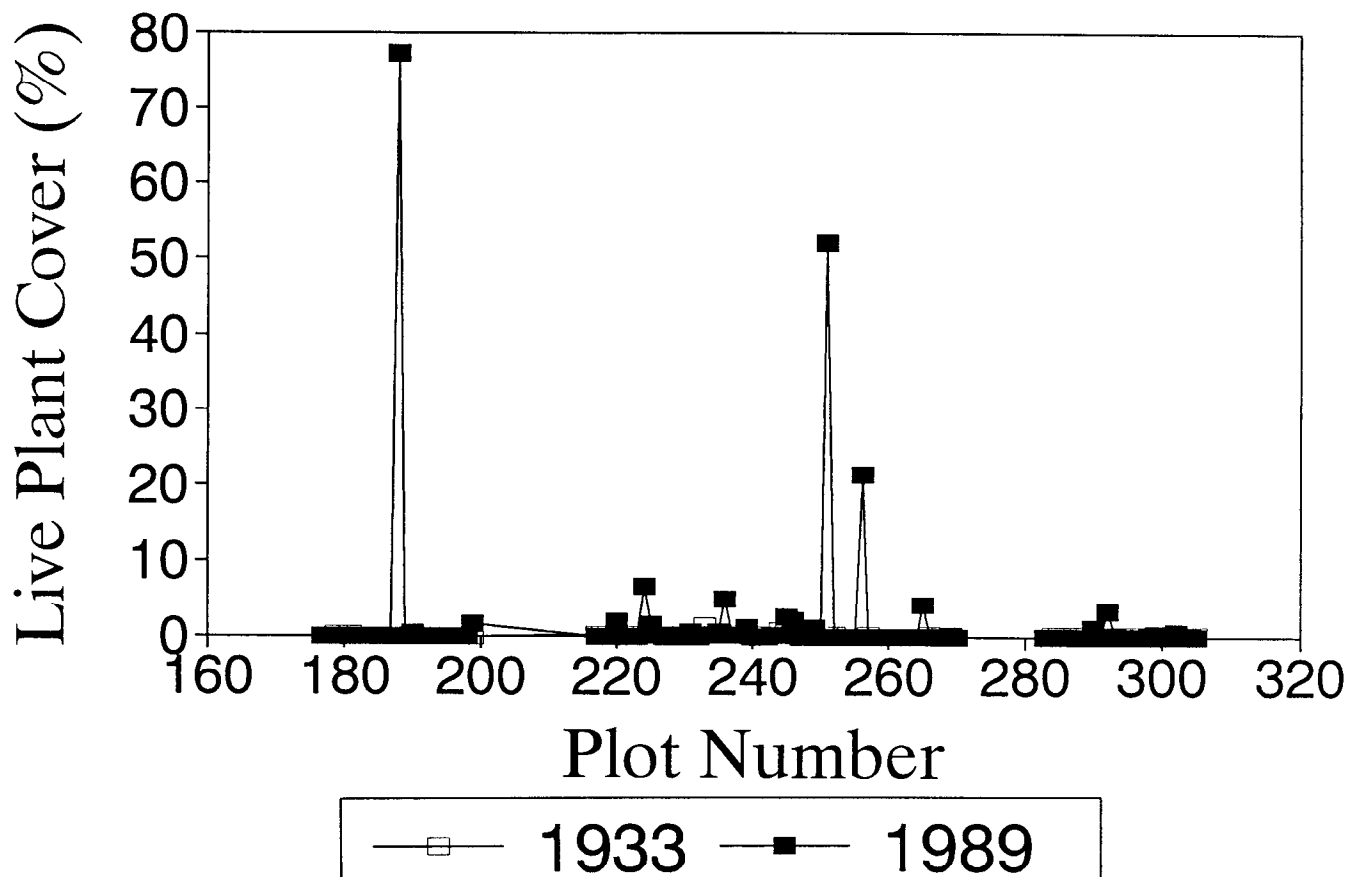


Fig. 3. Live plant cover by introduced annuals in the low rabbitbrush type, illustrating distribution along the transect line.

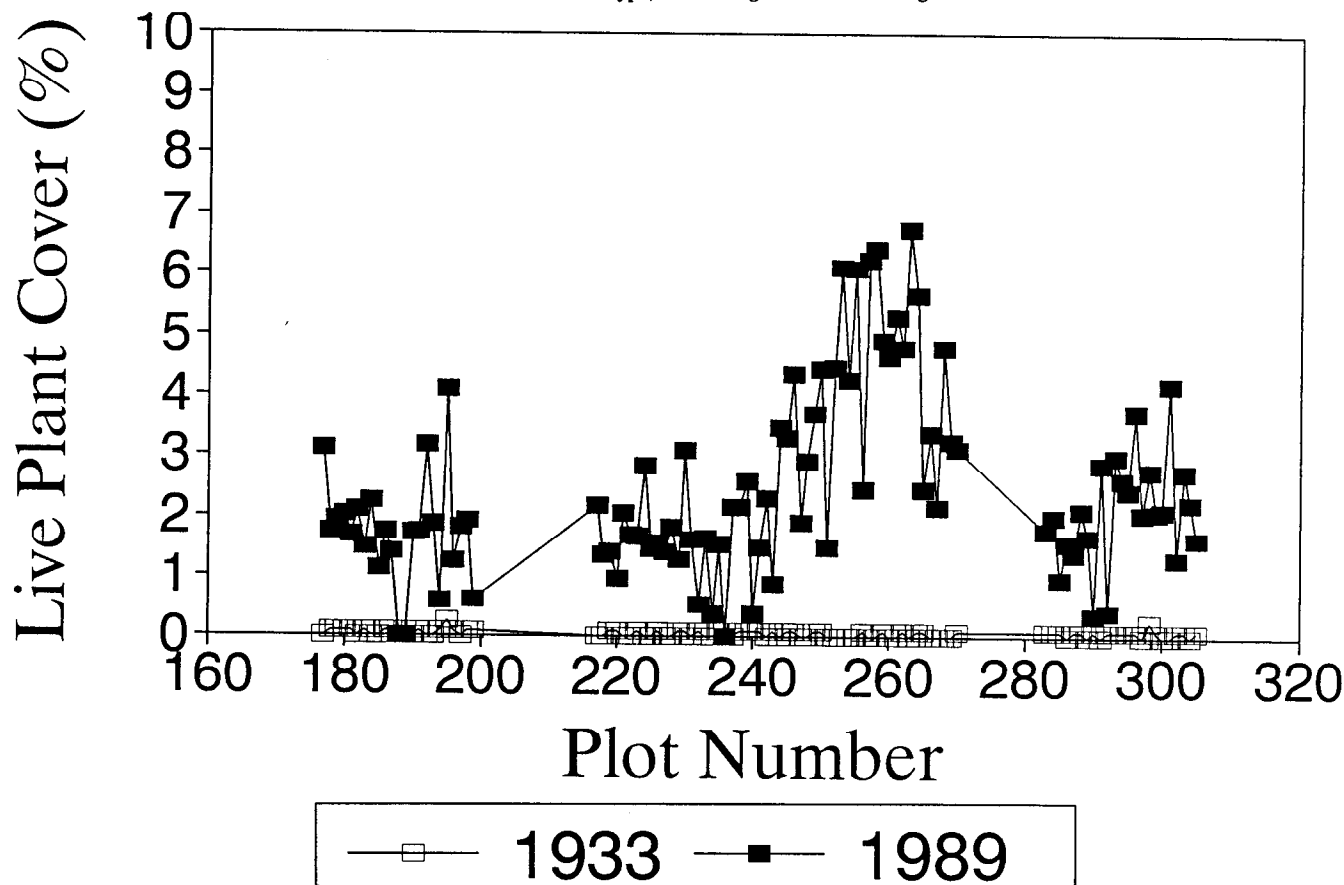


Fig. 4. Live plant cover by *Oryzopsis hymenoides* in the low rabbitbrush type, illustrating distribution along the transect line.

Table 4. Comparison of total percent live plant cover across vegetation types.

Vegetation type	No. of plots	1933 Mean live cover	1989 Mean live cover
		-----%-----	-----%-----
Low rabbitbrush	100	1.9 ± 0.04	*11 ± 0.9
Spiny hopsage	50	3.9 ± 0.2	*14 ± 0.7
Sagebrush	50	4.1 ± 0.5	*24 ± 0.9

* $P < .05$ for similarity of 1989 value to that for 1933.

apparent here.

A comparative species breakdown for the spiny hopsage type is given in Table 3, and illustrated at the growth form level of aggregation in Figure 2-A2. In this second vegetation type, a six-fold overall difference in shrub cover is spread among 4 major species, instead of being concentrated only in 1 as it was in the low rabbitbrush type. Also, the greater grass cover and species richness in 1989 are more widely distributed within the spiny hopsage type. Annuals in 1989 are far less important here than in the rabbitbrush type.

A point of special historical interest is the appearance of "*Bromus*" in 1933 in the spiny hopsage type, almost certainly an early penetration of cheatgrass. This species was not commonly recognized in the eastern Great Basin at the time (Mack 1981).

The greater diversity of shrub species in the hopsage type, as expressed in cover (Table 3), is reflected in that growth form's greater overall dominance in 1989 (Fig. 2-B2). This must be balanced, however, against the doubly greater total grass cover during 1989 (Fig. 2-A2).

Relative shrub densities in Figure 2-C2 indicate an even higher total number of living shrubs in 1933 as compared to 1989 than was the case in the rabbitbrush area (Fig. 2-C1) in 1933 as compared to 1989. Once again, this reflects smaller individuals, since a greater number of shrubs covered a more limited area in 1933 (Fig. 2-C2).

There were also many more dead shrubs reported in 1933 (Fig. 2-C2). This latter situation must be appreciated in the context of a practical difficulty, however. This arises when attempting to assess the degree of life among drought-deciduous spiny hopsage plants during a low-rainfall period. Thus some shrubs which were assessed as dead in 1933 may not actually have been so.

The Sagebrush Type

This type is found on a semidesert loam range site (BLM, unpublished), and varied in general character from a relatively open canopy to an almost completely closed one. Beginning the comparisons by dominance/diversity for the sagebrush type (Fig. 2-B3) between 1933 and 1989, one finds a rather close similarity over time. The exception is a nearly 20% greater number of species in 1989 (Table 3). These data indicate that the overall dominant, big sagebrush, has not negatively affected the proportion of other species, as it might have been expected to do from the conclusions of many other studies (Laycock 1991).

On the other hand, the increase in total shrub canopy cover expected to occur with sagebrush when there is livestock grazing, without other major treatment, is indeed happening (Fig. 2-A3). What was not anticipated from more controlled experimental studies (Laycock 1991) was a concurrent tenfold greater cover of perennial grasses in 1989 (Table 3). Not only has the absolute cover of grasses increased, but they have also made headway in the proportion of area they cover relative to other growth forms (Fig. 3A-3). While this proportional increase in perennial grass cover was not as dramatic as it was among the shrubs, the fact that it happened at all is worthy of special emphasis.

A (nonsignificant) relative increase in tree cover may be seen in

Table 5. Perennial grass growth season relationships, with season of growth for species, per Stubbendieck et al. (1982).

Classification	Vegetation type:		
	Rabbitbrush	Spiny Hopsage	Sagebrush
Perennial grasses, percent cover:			
1933 cool season (%)	0.043	0.093	0.064
1989 cool season (%)	2.4	1.4	0.56
1933 warm season (%)	0.021	2.1	0.0042
1989 warm season (%)	0.23	4.3	0.066
Ratio, cool season/ warm season:			
1933	2.0	0.33	15.
1989	11.	0.044	9.6

Figure 2-A3, and in more detail in Table 3. Since this portion of the transect does run up to near the lower edge of the valley's juniper belt, these data might suggest a slightly greater penetration into this sector by trees with the passage of time, as has been noted to be occurring in other parts of Pine Valley (Murdoch and Welsh 1971). However, any downward movement by junipers appears to be, at most, quite modest here. It was actually limited to just a single tree, which was found 200 m further downslope than those already present in 1933.

Figure 2-C3 illustrates that there were about twice the number of living individual shrubs in 1989 as compared to 1933. Fewer dead shrubs were also present in 1989. In this sagebrush-dominated community, considerably more shrub seedlings were apparent in 1989 (Fig. 2-C3), as was the case in the rabbitbrush-dominated type (Fig. 2-C1). Once again, the overall data suggest that a filling out of mature individual plants, rather than a larger number of them, primarily accounts for the greater overall canopy cover (Fig. 2-C3).

The greater 1989 shrub cover in this community type is concentrated in the 2 species of *Artemisia* (Table 3). However, there is a problem in assessing the ratios and relationships among the listings for sagebrush. As was noted in the methods section, there was difficulty in separating black and big sagebrush because of their apparent hybridization.

Cool-Season to Warm-Season Grass Relationships

Sims (1988) has argued that, over time, climate (i.e., mainly temperature and precipitation) changes may be expected to shift the relative percent of cool- to warm-season grasses (i.e., those with C₃ to C₄ photosynthetic pathways). Accordingly, in Table 5, we have summarized the totals from these 2 classes of grasses across each of the 3 vegetation types. These totals do dramatize the magnitude of the difference in grass cover along the transect between 1933 and 1989. However, no consistent seasonal growth pattern for that difference is apparent. Only in the sagebrush sector could this phenomenon possibly be said to be occurring in Pine Valley. This would seem to provide additional evidence that the differences between 1933 and 1989 which we have observed are not simply a result of trends in precipitation and/or temperature. Moreover, the warm-season grasses noted in our study represent some of the same low-growing species that have been shown to selectively increase under heavy domestic grazing pressure in the shortgrass prairie (Milchunas et al. 1988).

Discussion

The key points for this paper are evidence for an expansion of (1) overall live plant cover and (2) relative cover contributed by perennial grasses, both of which can best be described as dramatic (as

well as statistically significant) in all 3 shrubland types. These are rather unexpected trends in what was a previously more shrub-dominated community that is still primarily grazed by livestock (Blaisdell and Holmgren 1984). The dynamics are not those of the zero-sum game for grazing exclusion described in West et al. (1984) and Anderson and Holte (1981), or for the selective shrub reductions suggested by Orodho et al. (1990). The shifts are greater than the modest grass increases described for moderate grazing by Daddy et al. (1988), and are more clearly expressed than in a more intensely studied area of Pine Valley (Clary and Holmgren 1987, Whisenant and Wagstaff 1991).

While the total number of species is more or less unchanged in each vegetation type, the clear indication from examination of dominance/diversity (Fig. 2-B) is that more of the species found in all 3 community types are playing an appreciable role, and so could serve as a set of buffers against specialized environmental pressures (Archer and Smeins 1991). This should result in a heightened responsiveness to climatic variations, thus allowing overall growth to occur over a wider range of conditions (Westman 1990). Accordingly, these communities appear to have become more "robust".

While our data are insufficient to deduce even general patterns of causation, similarity in climate during and directly preceding 1933 and 1989 should minimize differences resulting from short-term vegetation responses to precipitation (Olson et al. 1985). The observed differences have been paired in time with reduced domestic livestock grazing, which has declined both in length of season and in absolute magnitude. Also, the study has been carried out on not-otherwise-specially-managed public lands in Pine Valley. Such ordinariness can be an important consideration in comparison to more controlled experiments, where subtle secondary effects can so easily be imposed inadvertently. While our observations are empirical, they do have the compensating advantage of directly reflecting federal land management as it has been done when no one is even thought to be watching.

Regarding introduced annuals, halogeton should not have reached the Pine Valley area at all by the time of the initial study (Dayton 1951), to which these data lend support. On the other hand, the early observation of cheatgrass (Table 3), even though in a small quantity, remains worthy of note. There is a smaller current presence of introduced annuals along the transect than has been reported for nearby areas (Sparks et al. 1990, Whisenant and Wagstaff 1991), especially under the droughty conditions of this study.

Greater shrub cover diversity by 1989 is reflected in that growth form's increased overall dominance with the passage of time, as is commonly expected for systems primarily grazed by domestic livestock (Archer and Smeins 1991). However, this abundance of shrubs must be balanced against the increases in total grass cover, exceeding those noted elsewhere by Daddy et al. (1988) for moderate grazing. This means, in part, that the overall dominant shrubs have neither negatively affected the proportion of other species, nor simply retained the same proportional balance, as might have more commonly been expected (West 1988, Laycock 1991).

The overall increase in perennial grasses in all 3 vegetation types is a central conclusion of our study. This is precisely the growth form shown by a great many experimental studies to be heavily stressed by domestic livestock grazing (Archer and Smeins 1991) and, in shorter-term studies, to respond most slowly to reduced grazing pressure (e.g., Wester and Wright 1987).

Such an increase in perennial grasses is encouraging in the Pine Valley area, where desertification was reported to have been in obvious process (Stewart et al. 1940, Murdoch and Welsh 1971, Blaisdell and Holmgren 1984). This change is especially notable because it occurred on land that received no substantial treatment subsequently, except reduction—not elimination—of domestic

grazing pressure. This remains true even if the increase was the partial result of threshold influences of precipitation (Laycock 1991). We could appropriately choose to use the word "recovery" at the growth form level to characterize our measured increase by assuming that accounts by early explorers depicting a notably greater grass presence before white settlement (Cottam 1961) have the breath of truth.

On the other hand, extensive citations should not be required to underline that all grazed lands in the United States do not have the same trends observed here. The differences which were observed between 1933 and 1989 in the studied portion of Pine Valley can only be held as strong evidence that in at least 1 publicly owned area, improvement in the condition of vegetation has indeed occurred, and that this is concurrent with changes in livestock management as a result of the Taylor Grazing Act. A cause-effect relationship cannot, however, be drawn from this nonexperimental study. Furthermore, our results should not be extrapolated without extreme caution, especially to areas where shorter times for recovery, wildfire, or less conservative human or animal use, in any form, has occurred.

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