

Secondary plant succession and vegetation recovery in two western Great Basin Desert ghost towns

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Abstract:

Secondary succession and vegetation recovery patterns are examined in two abandoned mining camps in the western Great Basin Desert. Results from ANOVA and Tukey comparison tests indicate that, after approximately 70 years of recovery, significant differences remain in total cover, therophyte cover and species richness between disturbed sites and control plots. The outstanding difference is the dominance of the introduced annual *Bromus tectorum* in the disturbed areas. Because the presence of *B. tectorum* changes the fire, grazing and competition dynamics of the desert vegetation assemblages, complete recovery or convergence of the disturbed vegetation to that of the ambient control plots is considered improbable. The dominance of *B. tectorum* at the expense of the other, native, species is viewed as a permanent change.

Article:

INTRODUCTION

Successional patterns in arid regions have been long argued. Shreve (1925, 1942) viewed vegetation in the Sonoran Desert as being in a non-dynamic condition, stating that desert plants did little to ameliorate their immediate environment for the possible establishment of successional species. Work done in the Chihuahuan Desert (Muller, 1940) and the Mojave Desert (Hunt, 1966; Rowlands & Adams, 1980) supported Shreve's conclusions. Conversely, vegetation studies done in the Mojave Desert (Wells, 1961; Vasek et al, 1975; Webb & Wilshire, 1980; Webb et al., 1987, 1988), the Sonoran Desert (Karpiscak, 1980), and the Great Basin Desert (Piemeisel, 1951; Hironaka & Tisdale, 1963; Hironaka, 1986) have suggested that succession has occurred.

Allen (1988) outlined three broad courses of succession in arid and semiarid lands. The first and second courses were based on the classic Clementsian view that, following disturbance, the original vegetation would re-establish or converge regardless of whether a monoclimate or polyclimate had occurred. The differences between the first and second courses of succession were based on the rate and patterns of succession, respectively. Different rates of succession in arid and semiarid regions have been shown to be affected by drought (Costello, 1944; Weaver, 1954), grazing (McLean & Tisdale, 1972) and the degree of disturbance (Webb & Wilshire, 1980).

Different patterns of succession are based on the idea that a variety of early successional species may colonize a disturbed site, but the later seral stages are dominated by the same species. This pattern is partially substantiated by the work of Collins and Adams (1983), who examined old-field succession in Oklahoma and found that the general pattern of successional convergence progressed from pioneer weeds to mature prairie.

The third course of succession, which is also Clementsian, predicts that, because of permanent stresses or changes in the initial conditions of an area, successional convergence is not possible. Instead, a new endpoint is envisioned where the disturbed vegetation would remain different from undisturbed areas (a disclimax). The factors preventing successional convergence would be the invasion of introduced plants species, loss or alteration of soil substrate and overgrazing by domestic animals (Westoby, 1979). For example, *Bromus tectorum*, an annual grass from the Mediterranean region, is now believed to be a successional dominant in the

Great Basin Desert where it effectively outcompetes native perennial grasses (Hironaka & Tisdale, 1963; Mack, 1981; DeFlon, 1986; Young et al., 1987; Billings, 1990).

One difficulty in examining any course of succession is that the period of observation is often insufficient to study long-term (>60 years) changes of vegetation patterns. Mining camps abandoned at the turn of the century provide an excellent opportunity to study these long-term changes. The purpose of this paper is to provide further evidence for the course of non-convergence in vegetative succession, and to illustrate how human use impacts may lead to permanent changes in the flora of a disturbed area in an arid environment. The role of changed fire regimes, introduced annuals, domestic grazing pressures and small mammal disturbances are examined to show how these factors, working together, have prevented convergence and have altered the path of recovery approximately 70 years after abandonment in two mining camps in western Nevada.

STUDY AREAS

Two abandoned mining camps, Terrill and Wonder, located in central, western Nevada were chosen for this study. Terrill (39°05'N, 118°46'W) was abandoned in c. 1915 and Wonder (39°35'N, 118°04'W) in c. 1925 (Paher, 1970; Carlson, 1974; Shamberger, 1974; S. Searcy, pers. comm.). Both camps are found on alluvially deposited volcanic sandy-loam to loamy-sand soils at the base of fault-block mountains, and are former gold/silver mining locations. The elevations of Terrill and Wonder are approximately 1305 m and 1740 m, respectively. Average annual precipitation for the two sites is primarily received in the winter (approximately 60%) as snow originating from mid-latitude cyclones, and ranges from 130 mm for Terrill to 250 mm for Wonder (estimated from Fallon, NV). Average January and July temperatures at the sites are -0.8°C and 22.8°C, respectively, for Terrill and -3.9°C and -20.5°C, respectively, for Wonder (National Oceanic and Atmospheric Administration, 1980; National Climate Data Center, 1986; Table 1).

Table 1. General characteristics of mining camps

	Terrill	Wonder
Elevation (m)	1305	1740
Precipitation (mm/year) ^a	130	250
Length of growing season (days) ^a	125	100
Average January temperature (°C) ^a	-0.8	-3.9
Average July temperature (°C) ^a	22.8	20.5
Grazing intensity (ungulates)	Light	Heavy ^b
Soil texture	Loamy-sand to sandy-loam	Silty-loam to sandy-loam
Bulk density R/F/C ^c (g/cm ³)	1.51* 1.47 1.41**	1.48* 1.46 1.42
Settlement date ^d	1911	1906
Abandonment date	1915	1925
Maximum population	50	600
Ore mined ^e	Gold/silver	Gold/silver

^a Estimated from data for Fallon, NV.

^b Major pressure applied by feral horses.

^c R/F/C refer to abandoned road, foundation peripheries and controls, respectively

* = Significantly different from control.

** = Significantly different from foundation peripheries.

^d Source: Shamberger (1974).

^e Source: Paher (1970).

The vegetation habitat-type for Terrill is classified as salt desert shrub (Tueller, 1989) and the dominant species are the shrubs *Sarcobatus baileyi*, *Atriplex confertifolia*, and *Tetradymia* spp., the grasses *Oryzopsis hymenoides* and *Bromus tectorum*, and the forb *Sphaeralcea ambigua*. Wonder's vegetation habitat-type is classified as sagebrush/grass (Tueller, 1989), and is dominated by the shrub *Artemisia tridentata* and the grasses *B. tectorum*

and *Sitanion hystrix*.

Domestic grazing impacts at all the three land-use sites at Terrill have been light due to the lack of a nearby water source. Grazing in Wonder, however, has been significantly greater. Sheep, cattle and feral horses have all foraged the three land-use sites in the area since abandonment. Only feral horses have grazed the area in the last ten years (A. Anderson, pers. comm.).

METHODS

Two sites were chosen because of their remoteness and their long periods of abandonment. Two levels of disturbance were compared to control plots at each site. Areas that were greatly disturbed are abandoned roads where the soil and vegetation have been impacted by vehicular traffic. Moderate levels of disturbance, primarily from trampling, were considered to be those areas around and within 5 m of foundation peripheries. Data gathered at both disturbed areas were compared to control plots, which are minimally disturbed (by mining activities) areas located near (< 2 km) but outside actual townsites. The selection of these three different land-use categories was aided by the examination of historic photographs taken during active periods. Every effort was made to ensure that the three different land-use categories were similar to each other in terms of slope, aspect, elevation, soil type, and parent material, so that an effective comparison could be made. Data were gathered in June 1990.

Vegetation data for the abandoned roads and controls were gathered using a stratified unaligned random sampling method. Thirty, 5-m line-transects were set parallel to the road direction for the abandoned roads (total of 150 m) over 300 m of roadway. Data from control plots were gathered along 40 5-m line-transects (total of 200 m) in a 400 m² area. Forty, 5-m line-transects (total of 200 m) were sampled perpendicular to the foundation peripheries, with consideration given to ensure that vegetation was equally sampled on all side of the former buildings.

Bulk density measurements were gathered at random points along each of the line-transects for the abandoned road (30 samples), foundation peripheries and control sites (40 samples each). A soil sampler was used to extract 2 × 2 inch (5.08 × 5.08 cm) cylindrical cores from the surface. The cores were oven-dried overnight and weighed to determine bulk density (Blake, 1965).

Variables used for analysis were cover percent-ages based on Raunkiaer's life-form classifications for phanerophytes, tall (> 50 cm) perennials with renewal buds on upright shoots; chamaephytes, perennials with renewal buds on the ground surface; hemicryptophytes, tussock plants with buds in or just below surface; geophytes, perennial earth plants with buds below surface as bulb or rhizome; and therophytes, annuals that overwinter as seeds (Goldsmith et al., 1986; Tables 2 and 3). At all sites, understory was counted in addition to overstory, so total cover could theoretically exceed 100%. Additional variables used were species richness, measured by the average number of species per 5-m transect, and total cover.

Statistical differences between abandoned roads, foundation peripheries and control plots data were examined using ANOVA and Tukey multiple comparison tests (Zar, 1984; SAS, 1985). The data did not meet the assumption of equal variances, but ANOVA is robust enough a technique to work well despite this problem (Zar, 1984). The vegetation data, except species richness, were, however, arcsine-transformed before running ANOVA, because the data were not normally distributed, and the cover percentages were small, being often less than 30%. All significant Tukey tests are reported at the 0.05 level.

RESULTS

Twenty-seven species were found at Terrill (Table 2). Eight of these were found in all land-use categories: *Artemisia spinescens*, *Atriplex confertifolia*, *Bromus tectorum*, *Hymenoclea salsola*, *Lycium shockleyi*, *Oryzopsis hymenoides*, *Sarcobatus baileyi* and *Sphaeralcea ambigua*. The dominant species for the abandoned road were *B. tectorum*, *L. shockleyi*, *O. hymenoides*, and *S. baileyi*, accounting for nearly two-thirds of the relative cover. The dominant species for the foundation peripheries were *A. confertifolia*, *B. tectorum* and

Halogeton glomeratus. The control plot was dominated by *S. haileyi*, *A. confertifolia* and *B. tectorum*. *S. baileyi* accounted for over a third of the relative cover (Table 2).

Table 2. Absolute (A) and relative (R) cover % of species in Terrill

Life-form ^a	Species	Road		Foundation		Control	
		A	R	A	R	A	R
H	<i>Agropyron spicatum</i> var. <i>vaseyi</i>	—	—	—	—	+	0.2
T	<i>Arenaria douglasii</i>	—	—	+	0.1	+	0.2
C	<i>Artemisia spinescens</i>	1.1 (2.4)	8.2	0.5 (1.6)	2.4	1.1 (2.5)	5.3
P	<i>Atriplex confertifolia</i>	0.9 (3.0)	6.8	3.5 (6.0)	16.3	2.1 (3.6)	10.3
T	<i>Bromus tectorum</i>	1.5 (2.8)	10.7	4.9 (6.3)	22.9	2.4 (3.5)	11.7
C	<i>Ceratoides lanata</i>	—	—	—	—	0.4 (1.7)	2.0
T	<i>Chaenactis stevioides</i>	—	—	—	—	0.2	0.8
P	<i>Chrysothamnus nauseosus</i>	—	—	1.0 (2.8)	4.5	—	—
C	<i>Eriogonum inflatum</i>	—	—	0.1 (0.2)	0.3	—	—
C	<i>Eriogonum umbellatum</i>	—	—	0.1 (0.2)	0.2	—	—
T	<i>Halogeton glomeratus</i>	—	—	3.4 (6.8)	15.9	—	—
G	<i>Hilaria jamesii</i>	—	—	0.3 (1.7)	1.3	—	—
P	<i>Hymenoclea salsola</i>	0.5 (1.8)	3.3	0.9 (4.9)	4.4	0.1 (0.6)	0.5
C	<i>Lepidium montanum</i>	0.3 (1.2)	2.2	0.7 (2.5)	3.3	—	—
C	<i>Leptodactylon pungens</i>	—	—	0.1 (0.3)	0.2	0.1 (0.5)	0.4
P	<i>Lycium shockleyi</i>	1.5 (4.0)	10.7	0.7 (3.5)	3.3	0.4 (1.1)	1.9
T	<i>Malacothrix glabrata</i>	+	0.1	—	—	0.2 (0.7)	1.1
T	<i>Oenothera refracta</i>	—	—	+	0.2	—	—
H	<i>Oryzopsis hymenoides</i>	4.0 (6.4)	29.7	0.2 (1.2)	1.1	2.0 (3.3)	10.1
P	<i>Psoralea polydenius</i>	0.9 (3.8)	6.3	—	—	—	—
T	<i>Salsola kali</i>	+	0.2	1.2 (3.3)	5.4	—	—
P	<i>Sarcobatus baileyi</i>	1.7 (4.4)	12.7	1.4 (6.6)	6.6	7.6 (10.4)	37.2
T	<i>Sisymbrium altissimum</i>	+	0.1	—	—	—	—
C	<i>Sphaeralcea ambigua</i>	0.5 (1.6)	3.8	1.3 (2.9)	6.3	0.9 (2.0)	4.7
T	<i>Stephanomeria exigua</i>	—	—	0.3 (1.5)	1.5	0.1 (0.5)	0.4
P	<i>Tetradymia glabrata</i>	—	—	0.7 (2.7)	3.3	2.4 (6.5)	11.8
P	<i>Tetradymia spinosa</i>	0.7 (3.8)	5.1	—	—	0.3 (1.3)	1.5

^a Life-form symbols are: P, phanerophytes; C, chamaephytes; H, hemicytrophites; G, geophytes; T, therophytes. Standard deviations are shown parenthetically; —, species not present at any site; +, species present, but with insufficient cover to survive rounding.

Only seven species were observed in Wonder and four of these occurred in all land-use categories. These were *A. tridentata*, *Astragalus* spp., *B. tectorum* and *S. hystrix*. *A. tridentata* was the dominant species at both the abandoned road and control sites, representing approximately two-thirds of the relative cover. At the foundation peripheries, percentages of *A. tridentata* and *B. tectorum* together comprised approximately 90% of the relative cover (Table 3).

Total cover

In Terrill, differences in total cover were significantly greater for the foundation peripheries than for the abandoned road. Cover values ranged from 13.7% along the road to 21-3% along the foundation peripheries.

Greater cover values for the foundation peripheries and control plots could be attributed to the presence of the therophyte *B. tectorum* around the foundations, and greater phanerophyte cover (*A. confertifolia*, *S. baileyi*) on the control plots (Table 4).

Table 3. Absolute (A) and relative (R) cover (%) of species in Wonder

Life-form	Species	Road		Foundation		Control	
		A	R	A	R	A	R
P	<i>Artemisia tridentata</i>	22.7 (17.0)	73.9	24.9 (14.2)	42.1	22.4 (14.4)	65.4
G	<i>Arstragalus</i> spp.	0.2 (0.7)	0.7	—	—	+	0.1
T	<i>Bromus tectorum</i>	2.4 (3.3)	7.9	28.8 (31.3)	48.6	6.9 (8.5)	20.1
P	<i>Ephedra nevadensis</i>	0.8 (2.5)	2.6	—	—	1.3 (4.6)	3.8
H	<i>Festuca idahoensis</i>	4.0 (4.6)	12.9	5.3 (11.0)	9.0	3.0 (4.3)	8.9
C	<i>Gutierrezia sarothrae</i>	0.5 (1.5)	1.4	—	—	0.6 (2.3)	1.7
H	<i>Sitanion hystrix</i>	0.2 (0.4)	0.5	0.1 (0.5)	0.1	—	—

Symbols same as for Table 2.

Standard deviations are shown parenthetically; —, species not present at any site; +, species present, but with insufficient cover to survive rounding.

In Wonder, total cover was significantly greater around the foundation peripheries than for either the abandoned roads or the control plots. Cover around the foundation peripheries was 59.2% and nearly double that of the other two land-use categories. As in Terrill, the greater cover around the foundations could be largely attributed to *B. tectorum* (Table 4).

Table 4. Results of Tukey multiple comparison tests for Terrill and Wonder^a

Location	Cover (%)	P (%)	C (%)	H (%)	G (%)	T (%)	Species richness (%)
Terrill road	13.7* (9.1)	6.6 (7.3)	1.9 (3.8)	4.0* (6.4)	0.0 (0)	1.5* (2.8)	2.2** (1.0)
Terrill foundation	21.3 (12.8)	8.2 (10.9)	2.6 (3.9)	0.2 (1.2)	0.3 (1.7)	9.9 (10.2)	2.7 (1.3)
Terrill control	20.3 (13.7)	12.8 (14.0)	2.5 (3.4)	2.0 (3.4)	0.0 (0)	2.9* (3.7)	3.0 (1.2)
Wonder road	30.7* (18.3)	23.5 (16.6)	0.5 (1.5)	4.1 (4.6)	0.2 (0.7)	2.4* (3.1)	2.9** (0.9)
Wonder foundation	59.2 (31.9)	24.9 (14.2)	0.0 (0)	5.4 (10.9)	0.1 (0.2)	28.8 (31.3)	2.7 (0.6)
Wonder control	34.3* (18.3)	23.7 (15.1)	0.6 (2.3)	3.0 (4.3)	0.1 (0.3)	6.9* (8.5)	2.5 (0.7)

^a Significant differences from Tukey tests based on arcsine transformations although data are shown in non-transformed state.

* Significantly different ($p < 0.05$) from foundation peripheries.

** Significantly different ($p < 0.05$) from control.

Standard deviations are shown parenthetically; —, species not present at any site; +, species present, but with insufficient cover to survive rounding.

Therophytes

The life-form category that was significantly different between the land-use groups was the therophytes. Therophyte cover was highest in the moderately disturbed areas (foundation peripheries). In Terrill, therophyte cover around the foundations was 9.9%, and only 1.5% and 2.9% (A) around the abandoned roads and control plots, respectively. While therophytes represent all annuals, the major contributing species were three exotics, *Salsola kali*, *Halogeton glomeratus* and *B. tectorum* (Table 4). Therophyte cover in Wonder was similar to Terrill although it consisted entirely of *B. tectorum*. Around the foundations, therophytes comprised 28.8% of the cover, while only 2.4% and 6.9% were found on the roads and control plots, respectively (Table 4).

Species richness

Species richness was significantly greater on the Terrill control plot than on the abandoned road and increased with decreasing disturbance. Along the roads 2.2 species were sampled per 5 m while 3.0 species were sampled on the control plots (Table 4). Species richness on abandoned roads in Wonder was significantly greater than for control plots, and decreased with decreasing disturbance. An average of 2.9 species were found on abandoned

roads while control plots had 2.5 species (Table 4).

Bulk density

Bulk density measurements were significantly greater for abandoned roads (1.51 g/cm^3) and foundation peripheries (1.47 g/cm^3) than for the control plot (1.41 g/cm^3) in Terrill. The abandoned roads in Wonder also had significantly greater bulk density (1.48 g/cm^3) than the Wonder control plot (1.42 g/cm^3) (Table 1).

DISCUSSION

Vegetation recovery may be assessed by both non-specific and specific parameters (Webb et al., 1988). Non-specific parameters include comparing, for example, either total cover or shrub cover of disturbed sites to adjacent control plots. This type of comparison gives an excellent general descriptive feel for the data, yet is insufficient in that sites of equal cover may be comprised of entirely different species. Some specific parameters are composition, diversity, life-history characteristics and life-form classifications. Recovery for this paper is based on criteria defined by Pielou (1988) and is dependent upon comparing two measures between disturbed and control plots: (1) equal diversity for this study, species richness; and (2) similarity of physiognomy as defined by Raunkiaer's (1934) system.

Introduced annuals

The dominant feature of recovery in both towns is the significantly higher percentage of therophytes around foundation peripheries than in the two other land-use categories. In both towns, this is a reflection of the high percentage of *B. tectorum* and, additionally in Terrill, *Halogeton glomeratus* and *Salsola kali*. All three species are non-native. A curious situation is created in that the moderately disturbed land-use category, foundation peripheries, has significantly greater cover of these species than do the more severely disturbed land-use category, abandoned roads.

Soil bulk density values have been shown to exhibit a significant negative correlation to the establishment of *B. tectorum* seedlings (Thill et al., 1989). Bulk density values at the three land-use categories at both towns were greater along the abandoned roads than the foundation peripheries and may be responsible for the differences of therophyte establishment between abandoned roads and foundation peripheries (Table 1). In addition, *S. kali* and *B. tectorum* both have extensive root systems that penetrate approximately 25-30 cm (Allen & Knight, 1984), that may be limited by the higher soil compaction. Finally, the loss of topsoil in the road ruts may have decreased levels of phosphorus and nitrogen to the point where the establishment of *B. tectorum* is critically influenced (Charley & Cowling, 1968; Morrow & Stahlman, 1984).

Role of fire

Once established, *B. tectorum* greatly increases the chance of fire in both *A. tridentata* and *Atriplex confertifolia*/*S. baileyi* communities because it provides a fine-textured, inflammable fuel that is dry by mid-July (Stewart & Hull, 1949). Further, *B. tectorum* burns more readily than the grasses that it replaces, and may accumulate over several years because the dry environment and low humidity prevents rapid decomposition (Stewart & Hull, 1949; Young et al., 1987). Fire is a natural component of the *A. tridentata* landscape, although historically occurring no more often than every few decades (Vale, 1982; Trimble, 1989). Increased dominance of *B. tectorum*, however, changed both the intensity and frequency of the fires, and because *A. tridentata* does not resprout after fire and can only reproduce by seed (West, 1983; Mozingo, 1987). Consequently, burned areas of *A. tridentata* are dominated indefinitely by *B. tectorum*. Litter accumulations at Wonder are sufficient to fuel a fire although no fires have been reported in the area since abandonment and no charcoal was found in the soil analyses. Lack of fire may be a result of insufficient fuel loads during the period when cattle and sheep heavily grazed the area (up to ten years ago).

Atriplex/Sarcobatus communities found at Terrill are characterized by infrequent fires due to the lack of sufficient fuel to carry the fire from shrub to shrub (Young et al., 1987). Invasion of *B. tectorum* provides the fine fuel and consequently alters the vegetation dynamics in a community that has not evolved with fire. Similar to the *A. tridentata* community, fire promoted by *B. tectorum* should lead to the further dominance of *B.*

tectorum.

Grazing pressures

Once established, *B. tectorum* often will continue to dominate because small mammals such as jackrabbits and field mice influence the dynamics of the grass (Stewart & Hull, 1949). Pyke (1986, 1987) examined influence of *Microtus montanus* grazing on the population dynamics of *B. tectorum* and the native grass *Agropyron spicatum* in eastern Washington. He found that *A. spicatum* was more susceptible than *B. tectorum* to the intensity and time of grazing, and growth response following grazing, which contributed to a shift in dominance to *B. tectorum*. Similarly, Hironaka (1986) found that plant succession in a southern Idaho livestock enclosure could be delayed or stopped at the *B. tectorum* stage because of pressure applied by jackrabbits and rodents. Dominance of *B. tectorum* was attributed to the heavy litter accumulation produced by the plant that provides favorable habitats for the small mammals, whose foraging patterns diminish the competitive ability of other species (Hironaka, 1986).

Feral horses provided additional grazing pressure, further diminishing the competitive ability of native perennial grasses and favoring dominance by *B. tectorum*. Grazing pressure in Wonder is classified as heavy to severe because the feral horses utilize approximately 61-100°A of the grasses in the area (A. Anderson, pers. comm.). Such heavy foraging again favors *B. tectorum* because it can quickly establish following disturbance because of a large seedbed and high inter-specific competitive ability (Allen & Knight, 1984). Similar results have been observed on the *A. tridentata* range of central Utah, where cattle grazing has led to the loss of native perennial grasses and the increased cover of introduced annuals (Brotherson & Brotherson, 1981).

Succession and the role of recovery

Recovery at Terrill and Wonder has followed similar paths in that the disturbed land-use categories show few differences among the variables with the exception of species richness and the therophyte and phanerophyte life-form categories. Characteristic of the land-use categories is the additional cover of therophytes, principally introduced annuals, but not the decrease of other species. At Terrill, the percentage of phanerophytes was significantly less along roads than in controls because the long-lived phanerophytes were destroyed by the vehicular movement and have been slow to recover. Phanerophytes have yet to establish in the road ruts at Terrill although this is likely a function of still highly compacted soils (Webb & Wilshire, 1980; Lathrop, 1983; Prose et al., 1987). Future soil loosening on the road ruts should facilitate the establishment of the phanerophytes.

Species richness was significantly less along the Terrill road than the control and is a reflection of fewer phanerophytes occurring along the road than on the controls. Conversely, species richness was significantly greater along the Wonder road than in the control and is a result of more grass species establishing along the road because *B. tectorum* could not exclude other species in the more compacted soil.

Vegetative convergence has occurred to a point given the slow growth of perennials, but complete recovery of the disturbed sites appears unlikely. The continued dominance of therophytes, particularly *B. tectorum*, is much more likely because of its superior ability to excel in disturbed areas (Sparks et al., 1990). Following disturbance, *B. tectorum* can easily establish, creating a self-enhancing cycle by promoting the development of additional disturbances such as increased fire frequency/intensity, or increased small mammal grazing. Attempts to control *B. tectorum* in arid lands through either prescribed burns and subsequent seeding of native perennial grasses or the construction of grazing exclosures have been met with limited success, and often cost more than the appraised value of the land (Young et al., 1987; Roberts, 1990).

CONCLUSIONS

Results from similar studies done in the Mojave Desert vary considerably in their estimation of recovery times. Carpenter et al. (1986) examined secondary succession in abandoned homestead fields and suggested that convergence requires approximately 65 years for *Larrea tridentata* scrub and perhaps 130 years for *Artemisia tridentata*-*Juniperus osteosperma* scrub. Results from other studies done on *L. tridentata* scrub (Webb &

Wilshire, 1980; Prose & Metzger, 1985; Webb et al., 1987) state that the recovery of long-lived perennials may take several centuries to millennia, and that the rates of vegetation change are so slow that recovery periods may be influenced more by geomorphic and climatic variability than by biological factors. In the western Great Basin Desert, however, the recovery of disturbed sites may not be possible regardless of time if current environmental conditions remain fairly constant, because the dominance of *B. tectorum* has so altered the biological interactions (e.g. fire frequencies and grazing regimes) of the region.

After approximately 70 years of recovery, there are significant differences in total cover, therophyte cover and species richness within and between land-use sites, and these differences appear to be long-term if not permanent. They may be caused by either differential impacts during mining settlement, differential impacts due to grazing after settlement, differential responses of species at each site, altered fire regimes, non-native species invasion, or all of the above. Clearly, more detailed and controlled experiments are necessary to understand fully the mechanisms causing the observed differences.

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