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Restoring riparian meadows currently dominated by Artemisia using alternative state concepts - the establishment component

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Abstract. We evaluated the potential for restoring riparian grass and sedge meadows currently dominated by *Artemisia tridentata* var. *tridentata* with an experiment in which we burned sites with low, intermediate, and high water tables, i.e., dry, intermediate, and wet sites. To define the alternative states and thresholds for these ecosystems, we examined burning and water table effects on both abiotic variables and establishment of grasses adapted to relatively high (*Poa secunda* ssp. *juncifolia*), intermediate (*Leymus triticoides*), or low (*L. cinereus*) water tables. Wet sites had lower soil temperatures and higher soil water contents than dry sites. Burning increased soil temperatures on all sites. Undershrub microsites on control plots had the lowest temperatures, while former undershrub microsites on burn plots had the highest temperatures. Surface soil water was low on burn plots early in the growing season due to desiccation, but higher at deeper depths after active plant growth began. Emergence was generally greater on wet sites, but survival was microsite- and species-specific. Undershrub microsites on control plots facilitated emergence and first-year survival, but seedlings that survived initially harsh conditions on burn plots had similar numbers alive at the end. In general, favorable environments and establishment of species adapted to mesic conditions indicate that wet sites represent an alternative state of the naturally occurring dry meadow ecosystem type, and can be restored to grass and sedge meadows. Harsh environments and lack of establishment of species adapted to mesic conditions indicate that dry sites have crossed a threshold and may represent a new ecosystem type. Understory vegetation and seed banks on dry sites have been depleted, and restoration will require burning and reseeding with species adapted to more xeric conditions.

Keywords: Dry meadow; Great Basin; Restoration; Seedling emergence; Seedling survival; Soil temperature; Soil water; Thresholds.

Nomenclature: Hickman (1993); Cronquist et al. Vol. 5 (1994).

Introduction

Riparian corridors in the western US are often severely degraded due to overgrazing by livestock, stream channel incision, and lowered water tables (Belsky et al. 1999). A consequence of this degradation is the expansion and dominance of the shrub *Artemisia tridentata* var. *tridentata*, and loss of grass- and sedge-dominated meadows. Because riparian meadows increase water storage, decrease sedimentation, and provide forage for both wildlife and livestock, their restoration is of considerable interest to land managers.

Restoration of riparian ecosystems currently dominated by *Artemisia* to grass and sedge meadows requires understanding the alternative states and thresholds that exist for these systems. Classical succession models are often inadequate for assessing range condition or restoration potential in semi-arid ecosystems because they imply that the system will recover to the original condition when the stress is removed (Westoby et al. 1989; Laycock 1991; Chambers 2000c). For degraded rangelands, there is often no change in condition, or the change that occurs is not a reversion to the former condition. Transition and alternative state models have been used to explain these divergent and non-linear pathways both theoretically (Rietkerk & van de Koppel 1997; Rietkerk et al. 1997) and experimentally (Allen-Diaz & Bartolome 1998; Augustine et al. 1998). These studies have tended to focus on vegetation changes following specific human disturbances or management actions. More ecologically relevant is determining the underlying factors that influence resource changes and plant responses because the emphasis is on feedback mechanisms (Rietkerk et al. 1997).

In a restoration context, it is necessary to both identify the alternative states that exist for degraded ecosystems and define the thresholds between states in terms of quantifiable ecosystem characteristics (Hobbs & Norton 1996; Whisenant 1999). One method of identifying alternative states and defining thresholds is to use restoration experiments that examine the abiotic and

biotic responses of sites representing an appropriate range of environmental conditions. Restoration of *Artemisia*-dominated riparian areas to grass and sedge meadows requires removal of *Artemisia*. Fire plays an important role in structuring *Artemisia* ecosystems (Miller & Rose 1999), and can be used in experiments to remove *Artemisia* and restore riparian areas.

The ability of plants to establish over a range of environmental conditions may be a sensitive indicator of threshold crossings and, thus, site potential to support a given state. Little information exists on relationships between plant establishment and environmental variables for native meadows in the western US. However, a primary determinant of seedling establishment in mesic meadow (Martin & Chambers 2001) and *Artemisia* steppe ecosystems (Chambers 2000a) is soil water availability. In *Artemisia*-dominated riparian areas, soil water may vary as a function of water table depth and *Artemisia* competition (Campbell & Harris 1977). Fire can alter not only soil water availability, but also soil temperatures and soil nutrients (Whelan 1995). Soil chemical and physical properties often differ between undershrub and interspace microsites in *Artemisia* ecosystems (Burkhardt & Tisdale 1976; Chambers 2000b), and these differences persist even though burning eliminates the *Artemisia* plant (Blank et al. 1994,1998). Microsites may vary sufficiently to cause differential seedling establishment both before and after burning.

We identified two ecological types that occur naturally in the drainages and have similar soils and landforms as models for the restoration effort: (1) the dry meadow ecosystem type which is dominated by grasses and sedges and has relatively high water tables; and (2) the *Artemisia/Leymus cinereus* type which is dominated by *Artemisia* and *L. cinereus* with lesser amounts of dry meadow species and relatively low water tables (Weixelman et al. 1996). Water tables largely control vegetation dynamics in riparian ecosystems (Martin & Chambers 2001), and we believed that the threshold for restoration to grass and sedge meadows could be defined by water table depth. We conducted a restoration experiment in which a burning treatment was applied to sites with low, intermediate, and high water tables to remove *Artemisia*.

We addressed three questions. 1. How do soil water and temperature differ among water tables and in response to burning, and what are the effects of *Artemisia*? 2. How is establishment of plant species adapted to relatively high, intermediate, and low water tables affected by water table depth, burning, and *Artemisia*? 3. Can the abiotic and establishment responses examined be used to identify alternative states and define restoration thresholds for these ecosystems?

Methods

Study site selection and experimental design

We located five *Artemisia*-dominated study sites within the Toiyabe Mountain Range in central Nevada in July 1996 that represented a gradient of modification of the dry meadow ecosystem type. Selection was based on presence of species typical of the dry meadow and *Artemisia/Leymus* ecosystem types (Weixelman et al. 1996), and depths to water table determined from hand-augered wells. Sites were on drainage ways and stream terraces at elevations ranging from 2000 to 2300 m and were characterized by haplocryoll soils. They included two replicate high water table sites (Willow and Ledbetter Creeks; – 153 to – 267 cm), one intermediate water table site (Willow Creek; – 250 to – 280 cm), and two low water table sites (Willow and Marysville Creeks; ≥ 300 cm) (referred to as wet, intermediate, and dry sites, respectively). Wet and intermediate sites had higher cover of perennial graminoids and forbs, lower densities and volumes of *Artemisia*, and were more similar to the model dry meadow type than low water table sites (Linnerooth et al. 1998). Common species included *Carex douglassii, L. cinereus, L. triticoides, Poa secunda* ssp. *juncifolia,* and *Lupinus argenteus*.

A paired plot approach was used in which one plot at each site received the restoration treatment and the other plot served as a control (referred to as the burn and control plots, respectively). Plots (740 to 900 m^2) were fenced following treatment to exclude livestock and native ungulates. Treated plots were burned on 19 through 21 October 1996 using a drip-torch to remove both woody and understory vegetation. All *Artemisia* were killed as a result of the burn.

Environmental responses

Rain gauges were located at each site to record growing season precipitation. Water table depth was measured monthly throughout the growing season in burn and control plots at each site with three or four wells (12.7 cm diameter, perforated PVC) installed in September 1996.

Soil water content data were collected on the wet and dry sites. Soil samples were taken every three weeks during the early part of the growing season and monthly thereafter. Samples were collected for each treatment/microsite from the soil surface and from depths of 5-15, 30-40, and 60-70 cm using a small diameter auger (2 treatments × 4 depths × 4 replicates per site). Samples were weighed before and after oven drying (120 °C).

Soil temperature was measured for each microsite/treatment on the wet and dry sites (2 treatments × 2

microsites × 2 depths × 2 or 4 replicates per site). Networks of three thermocouples of equal length, spaced 50 cm apart, were installed at 30 cm deep. Surface temperatures were measured with 10 cm × 10 cm perforated copper plates placed 1 cm below the soil surface. Soil temperatures were recorded hourly with Campbell Scientific CR-10 microloggers.

Seedling emergence and survival

We examined seedling establishment of grass species adapted to relatively wet, dry, and intermediate site conditions (*P. secunda* ssp. *juncifolia*, *L. cinereus*, and *L. triticoides*, respectively) on burn and control plots. We used seeds of *Poa* that were collected near the study sites in the summer prior to the burn. Seed fill was low and we used commercial seed for the other species. All seeds were tested for viability using standard tetrazolium tests. Establishment quadrats (50 cm × 50 cm) were randomly located in both past and present undershrub and interspace microsites in burn and control plots (2 treatments × 2 microsites × 3 species × 4 replications per site). Quadrats were seeded in October 1996 following the burn with 100 filled seeds (1997 cohort) using a 50 cm × 50 cm grid with 100 evenly spaced holes. The small *Poa* seeds were placed 1 cm below the surface, while larger seeds of the two *Leymus* species were placed 2 cm below the surface. Seeding was repeated one year later in October 1997 with four new locations using identical methods (1998 cohort). Seedling emergence and survival was censused every three weeks in 1997 and 1998 starting in April or May and continuing through July. During the summer dry period (Aug- Sep) survival was monitored monthly. Root sprouts and seedlings of species that were not seeded were removed at each sampling date to reduce competition.

Statistical analysis

The overall study design was a split, split plot that was completely randomized at the main plot level. Water table depth was the main plot factor, burning treatment the split plot factor, and microsite the split, split plot factor. Where present, depth was treated as a split, split, split plot and date as a repeated measure. Sites within water tables were the main plot error term. All mean comparisons were performed using least squares means (Anon. 1990).

Soil water content was analysed by year using a five-way ANOVA (water table, treatment, microsite, depth, and date) (Anon. 1990). For the soil temperature data, regression equations were generated using weekly means for each treatment/microsite. Differences in regression equations then were compared using a general linear test approach (Neter et al. 1990).

Emergence data were corrected for initial seed viability, and three-way ANOVA (water table, treatment, and microsite) (SAS 1990) was used to determine differences in emergence for each species and cohort. Seedling survival was analysed with logistic regression in PROC LIFEREG (Weibull distribution, likelihood ratio test) (Anon. 1995; see Fox 1993). The independent variables used in the models included water tables and individual treatment/ microsite combinations. Intercept variables were selected based on consistent emergence and survival for the two-year study. Differences among parameter estimates for water tables and treatment/ microsites were calculated from Wald chi-square tests using the variances and covariance for the two parameter estimates of interest (Anon. 1995).

Results

Abiotic response

There was less total precipitation for all sites in 1997 (310 to 460 mm) than 1998 (420 to 720 mm). In 1997 and 1998, the Willow intermediate site received the most precipitation and the Marysville dry site received the least. Water tables reflected precipitation differences, and were lower and more variable spatially in 1997 than 1998. In both years, most of the total precipitation came during the winter. Water tables declined during the growing season but exhibited substantial overwinter recharge. Water tables ranged from a low of – 250 cm in September 1997 to a high of – 54 cm in May 1998 on wet sites, and from – 287 cm in September 1997 to – 83 cm in May 1998 on the intermediate site. On dry sites, water tables were generally inaccessible (< – 300 cm) but were as high as – 268 cm in May and June 1998.

In 1997, soil water content was lower than in 1998 early in the growing season (Fig. 1). Wet sites had higher overall soil water content than dry sites throughout the growing season, but only at lower depths (water table × depth, $F_{3,24} = 4.78$, $P = 0.0095$). Burn plots had lower soil water contents than controls for the 0 - 5 cm depth (treatment × depth, $F_{3,24} = 3.35$, $P = 0.0358$). These plots were more exposed and had lower overall soil water contents in March and April (treatment × date, $F_{7,224} = 54.11$, $P = 0.0001$) due to snow removal and desiccation by wind. After initiation of active plant growth in early June, burn plots had higher soil water contents than control plots. Depths critical for seedling establishment, 0 to 5 cm and 5 to 15 cm, had the lowest soil water contents for most dates (depth × date, $F_{21,224} = 68.00$, $P = 0.0001$).

In 1998, soil water content was extremely high early

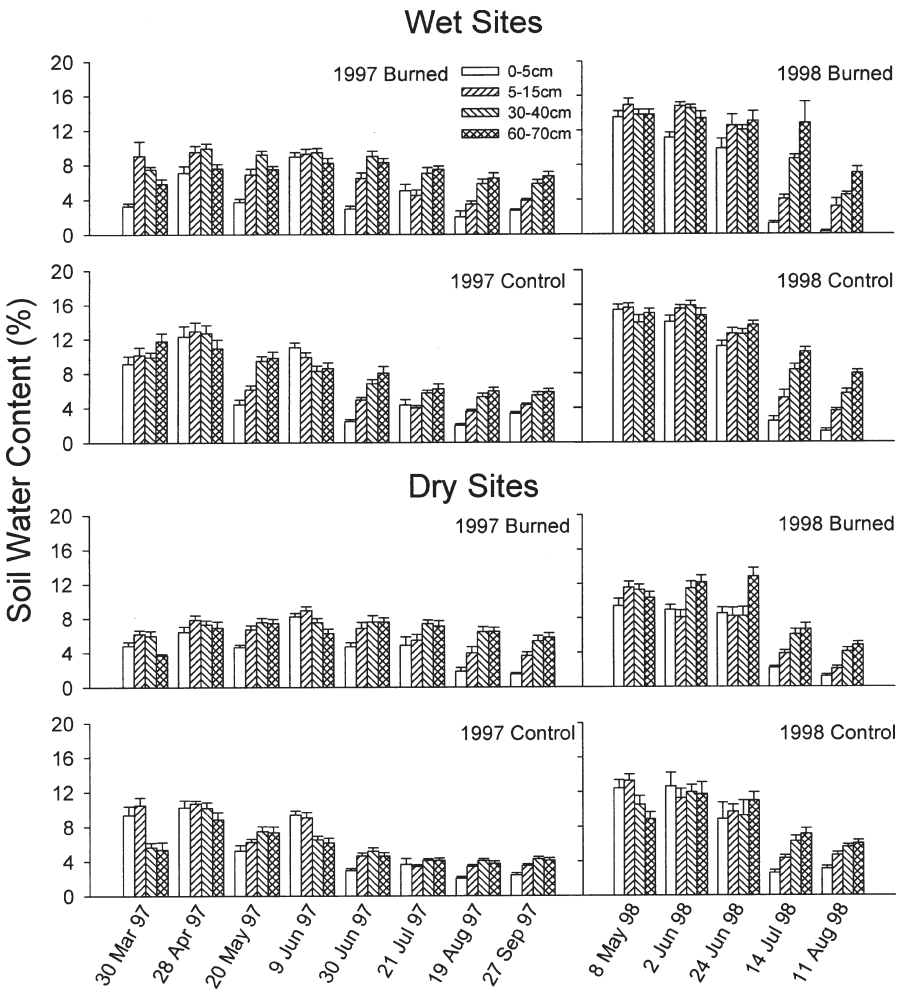

Fig. 1. Soil water content (%) by treatment and depth for the wet and dry sites during the 1997 and 1998 growing seasons. Values are mean ± S.E., $n = 8$.

Table 1. Mean soil temperatures (± S.E.) by year and month for each of the treatment/microsite combinations on the wet and dry sites.

	1997		1998	
	Wet	Dry	Wet	Dry
Burn undershrub				
1 cm	24.9 ± 0.25	25.9 ± 0.20	17.1 ± 0.14	21.9 ± 0.14
30 cm	18.9 ± 0.22	20.7 ± 0.03	13.5 ± 0.06	17.8 ± 0.05
Burn interspace				
1 cm	23.7 ± 0.22	25.5 ± 0.18	16.7 ± 0.15	20.5 ± 0.13
30 cm	18.9 ± 0.05	20.4 ± 0.03	13.5 ± 0.05	17.1 ± 0.05
Control undershrub				
1 cm	17.7 ± 0.14	19.4 ± 0.10	14.1 ± 0.01	17.6 ± 0.08
30 cm	14.7 ± 0.03	14.9 ± 0.03	11.2 ± 0.04	14.4 ± 0.04
Control interspace				
1 cm	22.5 ± 0.23	19.5 ± 0.12	17.3 ± 0.16	18.1 ± 0.10
30 cm	15.5 ± 0.03	15.5 ± 0.03	12.4 ± 0.04	14.9 ± 0.04

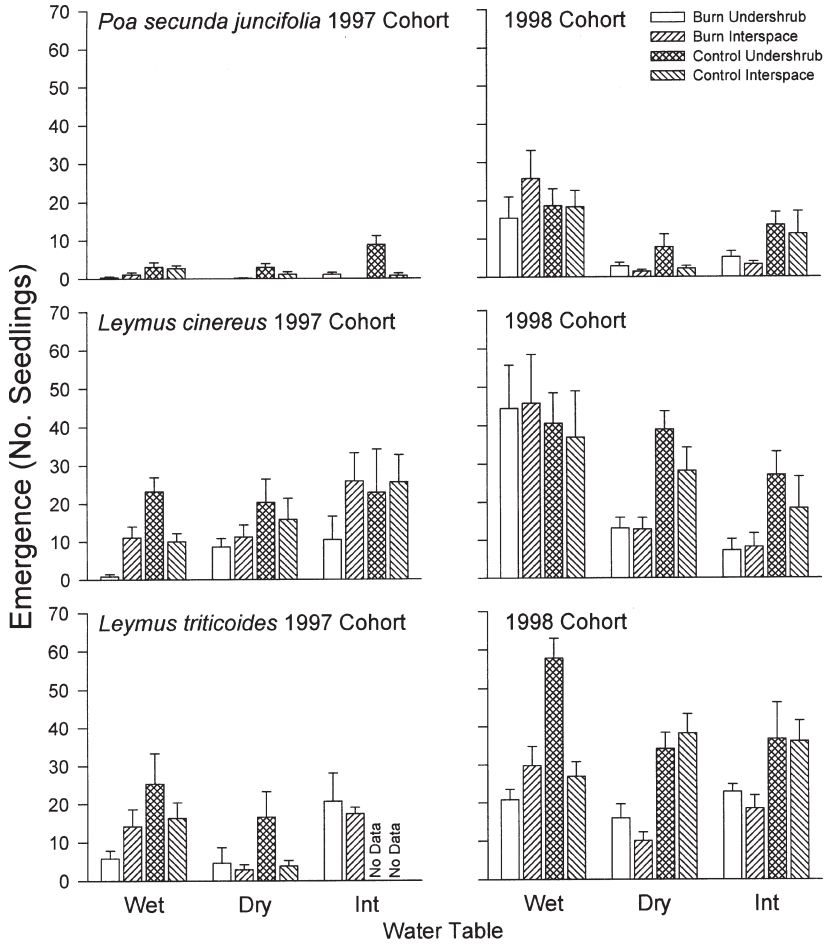

Fig. 2. Seedling emergence by species for the wet, dry and intermediate sites for the 1997 and 1998 cohorts. Values are mean ± S.E.; $n = 4$.

in the growing season, but wet sites still had higher water contents than dry sites (water table × date, $F_{4,\ 128}$ = 294.38, $P = 0.0001$) (Fig. 1). By August, water contents were similar to or lower than in August 1997. Water contents of surface depths were similar to lower depths early in the season, but were drier than lower depths by the end of June (depth × date, $F_{12,\ 128} = 24.85$, $P = 0.0001$).

Results of a separate analysis of matric potential days (Linnerooth 1999) were similar to those for soil water contents, except that differences among microsites were more evident. In general, burned undershrub and interspace microsites had higher water potentials than either microsite on control plots, especially for the dry sites.

Soil temperatures were typically higher in 1997 than 1998, and wet sites had lower soil temperatures

Table 2. Results from the lifespan analysis examining survival of the 1997 cohort during the 1997 and 1998 growing seasons and of the 1998 cohort during the 1998 growing season. Life-span estimates are derived from the natural log of the parameter estimates. Unlike letters indicate significant differences among microsites for each year (Wald χ^2, $P \leq 0.05$).

	1997 cohort		1998 cohort		
	Leymus triticoides	*Leymus cinereus*	*Poa secunda ssp. juncifolia*	*Leymus triticoides*	*Leymus cinereus*
Water table					
Dry	0.61 b	0.82 b	1.09 a	1.03 a	2.04 a
Intermediate	0.87 a	1.72 a	0.66 b	0.89 b	0.81 c
Wet	1.00 a	1.00 b	1.00 a	1.00 a	1.00 b
Treatment/microsite					
Burn interspace	0.97 a	1.09 a	0.72 b	0.60 b	0.51 c
Burn undershrub	0.94 a	1.42 a	0.68 bc	0.61 b	0.78 b
Control interspace	0.35 b	0.39 b	0.59 c	0.56 b	0.56 c
Control undershrub	1.00 a	1.00 a	1.00 a	1.00 a	1.00 a

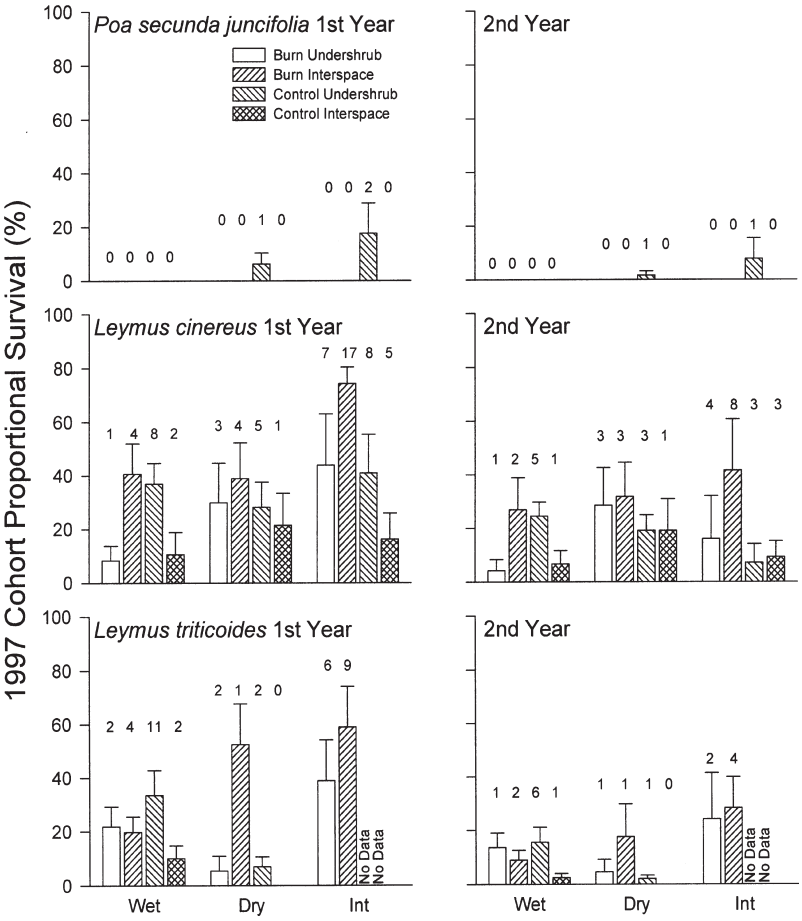

Fig. 3. Proportional survival for the 1997 cohorts of *P. secunda juncifolia, L. cinereus, and L. triticoides* during 1997 and 1998 on the wet, dry and intermediate water table sites. Values are means ± S.E.; n = 4. Numbers above individual bars indicate the numbers of seedlings alive at the end of the study.

than dry sites (Table 1). In 1997 and 1998, burn treatments had generally higher temperatures than controls for both microsites and depths (P = 0.01 to < 0.0001). Differences between undershrub and interspace microsites occurred primarily in control plots where undershrub temperatures were lower than those in interspaces for one or both depths (P = 0.0466 to P < 0.0001). The 30 cm depth had consistently lower soil temperatures than the surface (1 cm) depth for all treatment/microsites (P < 0.0001).

Seedling emergence and survival

Higher mean emergence was observed in the wetter and cooler spring of 1998 than 1997 across treatments and sites for *Poa secunda* (1.8 vs. 10.4; $F_{1,4}$ = 72.27, P = 0.0011), *Leymus cinereus* (15.5 vs. 26.8; $F_{1,4}$ = 79.09, P = 0.0009) and *L. triticoides* (12.9 vs. 30.0; $F_{1,4}$ = 28.26, P = 0.0060) (Fig. 2). In 1997, emergence of all species was higher on the wet and intermediate sites than on the dry sites. Also, emergence was higher in control undershrub microsites than in other microsites for *Poa* (microsite × treatment, $F_{1,4}$ = 16.04, P = 0.0161) and *L. cinereus* (water table × microsite × treatment, $F_{2,4}$ = 16.2, P = 0.0121) (Fig. 2).

During 1998, emergence was higher on wet than either intermediate or dry sites for *Poa* (water table × treatment, $F_{2,4}$ = 8.31, P = 0.1044) and *L. triticoides* (water table × microsite × treatment, $F_{2,4}$ = 8.86, P = 0.0339) (Fig. 2). For *Poa*, emergence tended to be higher on burn plots for wet sites, but control plots for the intermediate site. For *L. triticoides*, the highest overall emergence occurred in the control undershrub microsite on wet sites.

Seedling mortality of the 1997 cohort was lower for the second than first growing season (Fig. 3). Survival in fall 1997 vs. fall 1998 was 6.3 vs. 2.7% for *Poa,* 37.1 vs. 23.2% for *L. cinereus*, and 26.5 vs. 11.7% for *L. triticoides*. The life-span analyses for one-year and two-year survival were similar for the 1997 cohort. Thus, only the two-year analyses are included here.

For the 1997 cohort, *Poa* had inadequate numbers of seedlings emerge (0 to 9) to assess life spans. *Leymus cinereus* had the highest survival and life spans of the three species (Fig. 3, Table 2). For *L. cinereus* and *L. triticoides*, the highest life-span estimates were on the

wet or intermediate sites. For both *Leymus* species, there were few differences in life spans among microsites, except for the control interspaces which had the lowest life span.

The 1998 cohort had higher survival at the end of the first growing season than the 1997 cohort. Overall survival was 45.4% for *Poa*, 46.3% for *L. cinereus*, and 37.6% for *L. triticoides* (Fig. 4). Both *Poa* and *L. triticoides* exhibited similar life spans for wet and dry sites, but significantly lower life spans on the intermediate site (Fig. 4, Table 2). The species adapted to the driest conditions, *L. cinereus*, had the highest life spans on dry sites followed by wet and intermediate sites. For all three species, the highest life spans were generally on control undershrub microsites, while the lowest life spans were again on control interspaces.

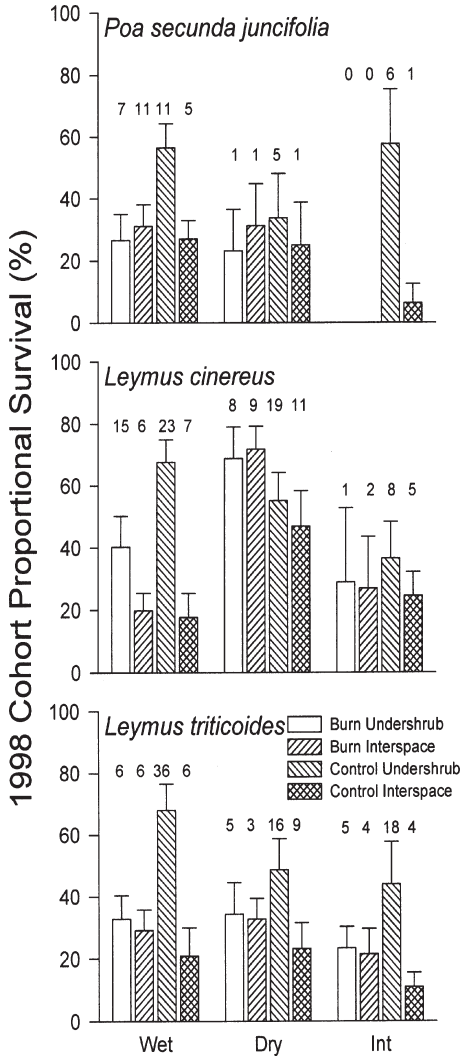

Fig. 4. Proportional survival for the 1998 cohorts of *Poa secunda* ssp. *juncifolia, Leymus cinereus,* and *L. triticoides* during 1998 on the wet, dry and intermediate water table sites. Values are means ± S.E., $n = 4$. Numbers above individual bars indicate the numbers of seedlings alive at the end of the study.

Discussion

Abiotic response

Both water table depths and pre-treatment vegetation indicated that *Artemisia* dominated sites within these riparian corridors are generally at the lower end of the water table depths required to support the dry meadow ecosystem type. Weixelman et al. (1996) described the dry meadow ecological type as having depths to saturation of –70 to –100 cm in June and July. Depth to water table reached these levels only in the spring and early summer of 1998 on the wet and intermediate sites. The relatively low water tables on the study sites are reflected in the persistence of *Artemisia* and rabbitbrush which cannot survive on sites that have high water tables or periods of prolonged soil saturation in the spring (Ganskopp 1986).

For the water tables examined in this study, wet sites generally had more favorable environmental conditions (higher soil water and lower soil temperatures) than the dry sites. Soils on wet sites had lower percentages of coarse fragments (12% for the wet sites vs. 23% on the dry sites) and higher organic carbon than soils on dry sites (Blank et al. 1998). Coarser soils and lower organic carbon on dry sites undoubtedly resulted in lower water retention and capillary movement. Also, dry sites had higher *Artemisia* densities and volumes, but lower abundances of perennials in the understory than wet sites (Linnerooth et al. 1998). *Artemisia tridentata* ssp. *wyomingensis* competes with understory species for soil water in the upper 50 cm early in the growing season, but uses water lower in the soil profile once this source is depleted (Campbell & Harris 1977). In dry years, perennial understory species on wet sites may effectively deplete soil water in near surface soils relatively early in the growing season. Higher *Artemisia* densities coupled with lower soil water retention and water tables on dry sites may contribute to generally lower water contents throughout the soil profile.

Burning had no measurable influence on water table depths on the small plots, but had significant effects on soil temperature and water. Generally higher soil temperatures following the burn was a function of eliminating the shade provided by burned vegetation, removing the insulating effect of litter, and altering the soil albedo (Whelan 1995). Initially lower soil water in surface soils was the result of snow removal, desiccation by wind, and increased soil temperatures resulting in higher evaporation (Wright & Bailey 1982). The increase in soil

water at depths ≥ 30 cm was probably a result of removing the deeper-rooted *Artemisia*. In areas sprayed with 2,4-D (2,4-Dichlorophenoxyacetic acid) to remove *Artemisia tridentata* ssp. *vaseyana*, soil water depletion during the growing season was reduced an average of 9% or 2.4 cm annually over a 20-yr period (Sturgis 1993). Most of the difference occurred at deeper depths (– 0.9 to – 1.8 m), and the greatest difference (+33%) occurred the year after spraying when herbaceous biomass was low.

Artemisia plants influenced soil temperature and water on control plots and had residual effects on burned plots. Reduced solar radiation coupled with higher humidity under *Artemisia* canopies (Burkhardt & Tisdale 1976) explain the lower soil temperatures in undershrub microsites. Higher burn temperatures under shrubs and mortality of existing vegetation may explain both higher soil temperatures and soil water in burn undershrub microsites. Fire in under canopy locations is often hot enough to burn existing vegetation and litter, and result in darker and warmer soils (Wright & Bailey 1982; Whelan 1995).

Establishment response

The abiotic conditions on the different sites had significant effects on seedling establishment. Like other semi-arid ecosystems, generally higher emergence and life spans on wet and intermediate sites can be related to higher soil water retention and availability (Chambers 2000 a, b). Higher vegetative regrowth, greater snow retention, and less extreme fluctuations in soil temperatures on wet and intermediate sites were probably also important. These results indicate that wet sites have the potential to support episodic establishment of species requiring mesic conditions, and to provide these species with adequate water once rooted.

Burn plots generally had lower emergence than control plots, except during the second year on wet sites. Near surface soils on burn plots had lower soil water availability and higher but more variable soil temperatures, and the combined soil temperature and water requirements for germination may not have been met. Also, mortality of seedlings that did emergence was probably high. Seedlings that survived the initially harsh conditions on burn plots exhibited relatively high life spans and greater growth (although not measured). The increase in soil water after *Artemisia* removal is greatest in the first year after treatment (Sturgis 1993), and this increase undoubtedly facilitates vegetative regrowth and seedling establishment.

Individual *Artemisia* plants affected seedling establishment both before and after the burn. In semi-arid ecosystems, facilitation of seedling establishment by shrubs that modify the environment can depend upon the degree of stress (resource levels and environmental conditions) experienced by seedlings (Callaway et al. 1996). In this study, emergence was highest in control undershrub microsites when other microsites had lower soil water availability or more extreme temperatures, e.g., spring and early summer 1997. Similarly, survival was highest in undershrub microsites when soil water availability was relatively high, e.g., 1998 when surface soil water rapidly declined (Figs. 1 and 4). The interaction between seedling and shrub usually switches from facilitation to competition once the life stage changes from seedling to adult (Callaway & Walker 1997). In this study, seedlings in undershrub microsites were typically smaller than in other microsites. After the burn, harsher environmental conditions resulted in lower seedling emergence. But, lack of competition from *Artemisia* and herbaceous species resulted in life spans and numbers of surviving individuals that were similar to other microsites (Figs. 4 and 5).

Species differences in emergence and survival can be related to their affinities to sites with different water tables. *Poa* occurs with high constancy in the dry meadow ecosystem type (Weixelman et al. 1996) and requires the highest water tables. It had almost no emergence in 1997, the dry year, but moderate emergence in 1998, the wet year. The species with intermediate water table requirements, *L. triticoides*, consistently had the highest emergence and survival on wet and intermediate sites. *L. cinereus* had the highest survival and life spans of the three species in the drier 1997, and it had the highest life spans on dry sites in 1998.

Alternative states and restoration thresholds

The abiotic and establishment responses can be used to identify alternative states and define restoration thresholds for *Artemisia*-dominated riparian corridors. Water table depths and shrub abundance indicated that the study sites were at the lower end of the water tables necessary to support dry meadow ecosystems. Despite this, wet sites had both higher soil water contents and lower soil temperatures than dry sites. Seedling emergence was higher on wet sites for all three species, and survival of species adapted to high and intermediate water tables, *Poa* and *L. triticoides*, was generally highest on wet sites followed by the intermediate site. In contrast, survival of the species adapted to more xeric conditions, *L. cinereus*, was clearly higher on dry sites. These results indicate that wet and intermediate sites represent an alternative state of the dry meadow ecosystem type, and have the potential to be restored to grass and sedge dominance. Due to relatively low water tables on these sites, maintaining grass and sedge dominance

probably requires proper grazing and possibly periodic burning. Dry sites appear to have crossed a threshold as reflected by a lack of environmental conditions necessary for establishment of species adapted to dry meadows. They may represent a new site type with the potential to support the *Artemisia/Leymus* ecosystem type. The understory vegetation and seed banks on dry sites have been depleted (Linnerooth et al. 1998; Chambers unpubl.), and restoration will require burning and reseeding with species adapted to more xeric conditions.

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