COORDINATED INTERMOUNTAIN RESTORATION PROJECT – FIRE, DECOMPOSITION AND RESTORATION

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1. INTRODUCTION

The Great Basin of the U.S.A. lies between the Rocky Mountains on the east and the Sierra-Nevada and Cascade mountains on the west. The area incorporates over 40 million ha of land (MacMahon 1979) in the Columbia Plateau, High Lava and Snake River Plains of Washington, Oregon and Idaho as well as the hydrologic Great Basin of Nevada and Utah. Vegetation within this region consists of coniferous forests and woodlands in the mesic mountains grading to semi-arid shrub grasslands in xeric lower elevations (West 1988).

Before people of European decent settled this region in the 1840s, much of the shrub grasslands consisted of "open stands of shrubs with a strong component of long-lived perennial grasses and forbs in the understory" (Miller, Svejcar & West 1994). Since settlement a number of human-caused factors have led to declines in the abundance of perennial grasses and forbs in these communities. In the mesic *Artemisia tridentata* ssp. *vaseyana* (mountain big sagebrush) communities, abundance and density of sagebrush and trees (mainly *Juniperus*.) has increased, whereas in the drier *A. t.* ssp. *wyomingensis* (Wyoming big sagebrush) communities, native shrubs and perennial tussock grasses are replaced by exotic annual grasses, such as cheatgrass (*Bromus tectorum*) and medusahead (*Taeniatherum caput-medusae*) (Eddleman 1989; Laycock 1967; Miller *et al*. 1994; Winward 1991).

Fire historically shaped the ecosystems of this region, but post-settlement alterations in fire frequency have caused major changes in vegetation composition, structure and potentially ecosystem processes. Along the ecotone between shrublands and pinyon pine/juniper savannas, fire suppression has lead to the increase in tree density and a reduction in shrubs and grasses (Miller and Tausch 2001). Conversely, fire frequency has dramatically increased from 1 in 70 to 1 in 10 years in the more xeric communities, where invasive annual grasses have replaced native shrub grasslands (Whisenant 1990). When weather conditions favor fires, such as 1999 when 680,000 ha burned, invasive annual grasses become increasingly dominant in the region (Pellant 1990).

To provide science-based information necessary for managers to break the wildfire-annual grass cycle, the U.S. Geological Survey in cooperation with the Bureau of Land Management (BLM) initiated a long-term research program called the Cooperative Intermountain Restoration Project. The overall objectives of this project were (1) to understand ecosystem responses to disturbances and invasive plants, (2) to develop strategies and appropriate techniques to maintain or restore functioning ecosystems, and (3) to demonstrate and transfer scientific results and applications to land managers. We have been attempting to address these objectives through simultaneous studies that investigate ecosystem processes and develop successful restoration strategies. In this paper, we report on some preliminary findings from two initial studies. One examines the decomposition process on invaded and native sites, while the second is a retrospective study examining the native plant establishment on lands rehabilitated after wildfires.

Keywords: Invasive Plants, Native Plants, Revegetation

2. ECOSYSTEM PROCESSES

Native shrub grassland communities typically have higher total available N and P under shrubs than in interspaces (Burke *et al*. 1989, Charley & West 1977). Soils associated with shrubs within native-dominated ecosystems have higher plant cover, water infiltration, and available nutrients, plus moderated temperature fluctuations, and lower levels of direct solar radiation than those in interspaces *(*Blackburn 1975; Doescher, Miller & Winward 1984; Eckert, Peterson

& Belton 1986). With the exception of solar radiation, these ecosystem factors are influenced by the higher organic matter content and thus more microbial biomass under shrubs. Increased microbial biomass causes higher N mineralization rates under shrubs than in the interspaces.

As native shrub grasslands are invaded by exotic annual grasses, such as cheatgrass, litter biomass and chemistry may change causing alterations in nutrient cycles. Evans *et al*. (2001) speculated that invasions of exotic annual grasses may immobilize nitrogen in the short-term and through fire may volatilize nitrogen resulting in nitrogen losses in the long-term. Field studies using additions of labile sources of carbon (e.g., sucrose) to immobilize nutrients in the microbial community have shown that cheatgrass growth is inhibited when carbon is added and native plants appear to tolerate the lower nitrogen environment (McLendon & Redente 1991; Redente, Friedlander & McLendon 1992; Young, Blank & Clements 1999). Thus, reductions in nitrogen with cheatgrass invasion that were predicted by Evans *et al*. (2001) seem to contradict the species responses seen in the other studies noted above. Field studies of nitrogen in cheatgrass- and native-dominated communities in Washington state provide additional evidence against general shifts in nitrogen cycles with shifts from perennial shrub grasslands to annual grasslands (Bolton, Smith & Wildung 1990, Svejcar & Sheley 2001). Clearly, more study is necessary to examine the relationship between nutrient cycles and native plant establishment in cheatgrass dominated ecosystems.

Since available nitrogen influences the establishment and growth of native plants relative to cheatgrass, it is important to know the mechanisms involved in shifts of available nitrogen so we might be able to efficiently reduce nitrogen during restoration. Although microbial organisms are a major contributor to decomposition and nutrient cycling, little is known about their temporal and spatial diversity and how they are associate with litter in the Great Basin. Since cheatgrass and native plants in this region often have temporally separated germination (Mack & Pyke 1983), knowledge regarding any temporal differences in decomposition and in diversity of microbes associated with decomposition might aid our understanding of ecosystem processes and native plant restoration. Since locations dominated by cheatgrass may undergo shifts in the chemical composition of vegetation, microbes available for decomposition might differ on sites dominated by native vegetation and those dominated by cheatgrass. Such differences might lead to differences in nutrient cycles. We established a study near Boise Idaho, U.S.A. to evaluate the spatial and temporal relationships of decomposition and the diversity of microbes associated with litter of three native perennial grasses (Sandberg bluegrass *Poa secunda*; bottlebrush squirreltail *Elymus elymoides*; bluebunch wheatgrass *Pseudoroegneria spicata*) and the exotic annual grass, cheatgrass.

2.1 Methods

Studies were conducted at four locations within Wyoming big sagebrush plant communities in Idaho. Two locations were on loamy 8 to 10 inch (203 to 254 mm) precipitation ecological sites and two on loamy 10 to 12 inch (254 to 305 mm) precipitation ecological sites. At each location, a pair of communities was located on the same soils. One was a diverse mixture of native shrubs, grasses and forbs with less than 20 % relative canopy cover of cheatgrass. The other was a site dominated by cheatgrass (> 50% of the relative canopy cover was cheatgrass and medusahead). Although these were not necessarily adjacent sites, a pair was always located on the same soil series. The difference in the proportion of cheatgrass is difficult to determine with certainty, but we believe that recent fires (< 20 years) on cheatgrass sites and lack of recent fires (> 50 years) have contributed to the difference. Other factors in native communities, such as, biological soil crust cover, may protect the site from cheatgrass invasion (Kaltenecker 1997) and thus reduce the continuity of fuels for fires.

At the end of the 2000-2001 growing season (June 2001), just prior to seed dispersal, we collected standing live aboveground biomass of cheatgrass, Sandberg bluegrass, bottlebrush squirreltail and bluebunch wheatgrass. Inflorescences were removed and vegetation was air-dried at 70° C to a constant weight. Fiberglass litter bags (1.5mm mesh) were filled with similar amounts of dry vegetation of one of the four species, sewn shut, individually tagged and weighed. During early July 2001, 14 sets of litter bags for each species were randomly placed in contact with the soil on each of three treatment sites (cheatgrass, native interspace, or native under shrub) in a split-plot design. Random samples of each species were removed for lignin-to-nitrogen ratios (Forage Analytical Laboratory, Oregon State University, Corvallis) when litter bags were filled.

Two litter bags of each species were collected from each treatment at regular intervals between August 2001 and September 2002. One bag of each species was dried to a constant weight at 70° C and weighed to determine the percent mass remaining. The other bag was kept cold in a sterile plastic bag and transported to the lab where samples were processed for total bacterial and fungal biomass within a day (Soil Foodweb, Inc., Corvallis OR) using a litter extract technique (compost technique). Percent mass remaining was analyzed using a multiple linear regression on litter species type, treatment site, and number of days in the field. Microbial biomass data were log-transformed and then analyzed using a linear mixed-effects model in which parameters were estimated using the restricted maximum likelihood method.

2.2 Results and Discussion

Lignin-to-nitrogen ratios showed a clear gradient among the four species (6.08 bluebunch wheatgrass > 5.45 Sandberg bluegrass > 4.17 cheatgrass > 2.60 bottlebrush squirreltail). Species with lower lignin-to-nitrogen ratios are predicted to have higher overall decomposition rates (Parton *et al.* 1983). Based on the lignin-to-nitrogen ratios, the two ruderal species, cheatgrass and bottlebrush squirreltail, would be predicted to have the fastest decomposition rates. Our lignin-to-nitrogen ratios for cheatgrass were nearly 75 % less than values (approximately 15) reported by Evans *et al.* (2001). Since lignin decomposition often takes years to occur (Minderman 1968), much of the difference in cheatgrass lignin-to-nitrogen ratios between our study and theirs may be associated with time of collection. Evans *et al.* (2001) sampled standing dead material midway through the year (March), which might allow the occurrence of some initial decomposition of readily available carbon and nitrogen. Litter decomposition during the first year is often associated with the breakdown of cellulose and hemicellulose (Killham 1994) leaving recalcitrant forms of carbon, such as lignin, for decomposition later. In our study, percent mass remaining did not differ among the four species or the three treatment sites (Figure 1, $F_{5,572} = 1.8$, $P = 0.11$). Time that litter was exposed to decay processes was one factor that clearly explained the first-year decomposition ($R^2 = 0.71$).
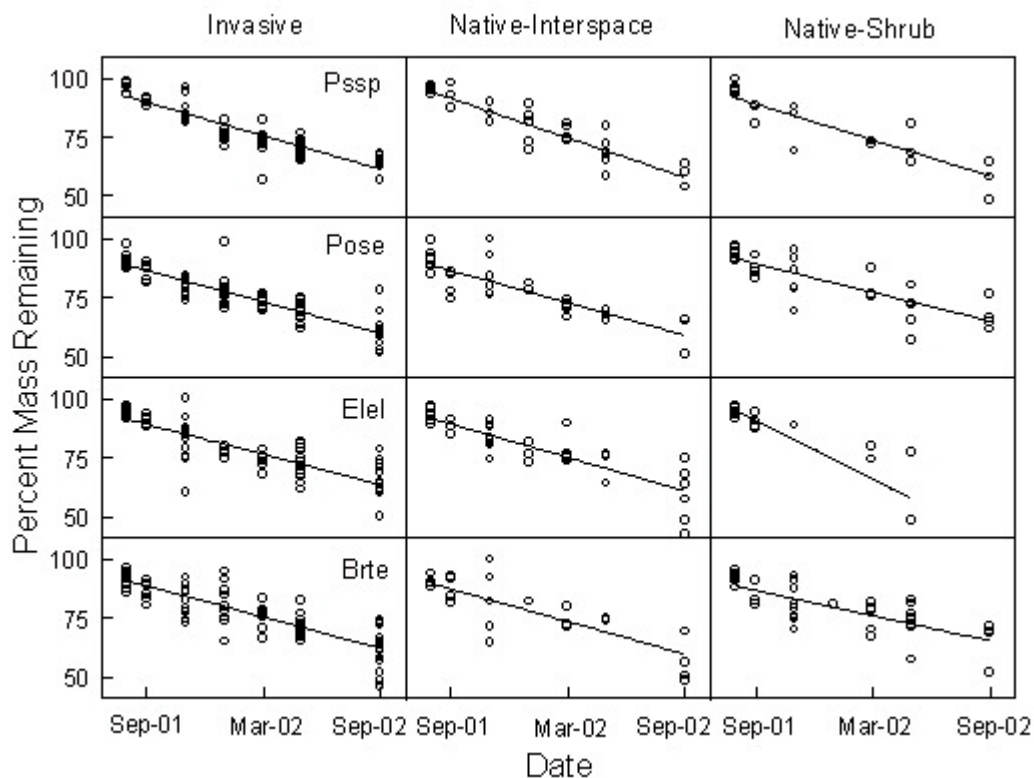

Figure 1. Litter biomass remaining (%) in litter bags for four species (Pssp – bluebunch wheatgrass; Pose – Sandberg bluegrass; Elel – bottlebrush squirreltail; Brte – cheatgrass) during the 15 months of the study.

Amounts of bacterial and fungal biomass associated with litter changed as decomposition proceeded throughout the seasons. Bacterial biomass increased slightly during autumn and winter, then declined between January and May. In contrast, fungal biomass nearly tripled from autumn to midwinter and then declined slightly during spring and summer (Figure 1). By the end of the first season of decomposition, bacterial and fungal proportions were nearly equal. We found no differences in bacterial or fungal biomass among the three microsites ($F_{2,204} \leq 2.2$, $P \geq 0.12$) and among the four species ($F_{3,204} = 2.3$, $P = 0.08$).

Figure 2. Fungal and bacterial biomass (± 1 S.E.) collected from grass litter samples during the 15 months of the study.
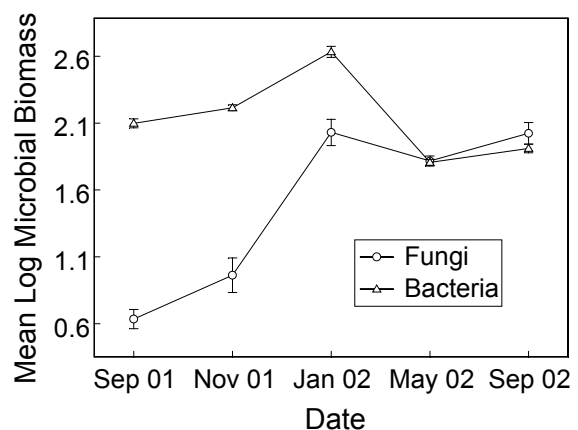
We found no evidence that microbial biomass associated with decomposing litter has changed on sites dominated by cheatgrass. Although Evans *et al.* (2001) proposed that changes in litter quality might be a mechanism for changes in nutrient availability, our study does not detect any differences in the microbial communities among the native and cheatgrass litter. Our lack of differences between native- and cheatgrass-dominated communities in decomposition rates and microbial communities aligns our study more closely with those that found similar nitrogen pools founds in native- and cheatgrass-dominated communities in the northern Intermountain West (Bolton *et al.* 1990, Svejcar & Sheley 2001), than with the differences noted by Evans *et al.* (2001). Differences in the distribution of seasonal precipitation between the Colorado Plateau (80% in spring, summer and autumn, Evans *et al.* 2001) and the northern Great Basin (80% in autumn, winter and spring, Svejcar & Sheley 2001) may contribute to the differences seen in nitrogen pools among the two studies. We propose that litter quantity and spatial distribution may be more important factors than litter quality for providing plant available nitrogen in the Great Basin. Cheatgrass does provide a more continuous cover of litter than native plants in communities where cheatgrass has not invaded. Our results did not examine decomposition beyond the initial year nor did they examine other vectors for decomposition such as microarthropods or nematodes. It is possible that these organisms change as communities become invaded and dominated by cheatgrass (Belnap and Philips 2001).

Clear differences exist between vegetation communities dominated by cheatgrass and native plants in the Great Basin, however these differences do not easily translate into differences in ecosystem processes. A more thorough understanding of nutrient cycling within invaded and native communities will help us understand degraded ecosystems and their components that require restoration. However, we cannot rule out the possibility that the invasive plants in some ecosystems may not modify ecosystem processes in significant ways, but merely compete more effectively for available resources (Svejcar & Sheley 2001). Thus, successful restoration may require direct reductions of competitor populations, for example weed control mechanisms, or modifications to the environment, such as additions of labile carbon to reduce nitrogen availability, that may give native plants with an opportunity to establish and survival when nutrient levels are low. We intend to continue our investigations of mechanisms to control invasive plants while simultaneously examining the role invasive plants play in shaping ecosystem processes within our region. We hope these efforts will one day lead to strategies for restoring native plants to Great Basin ecosystems now dominated by cheatgrass.

3. REHABILITATION SUCCESS: A RETROSPECTIVE STUDY

The conversion of perennial shrub grasslands to annual grasslands through the grass-fire cycle has been noted in several ecosystems throughout the world (D'Antonio & Vitousek 1992). Within the semiarid portions of the Great Basin, fire leaves land vulnerable to two potential threats, a reduced hydrologic capacity to safely capture and release water during precipitation episodes (rainfall or snowmelt) and the spread of exotic invasive plants. Vegetation removed by fires is no longer available to protect soil from raindrop impacts and to provide obstructions that reduce surface flows of water and enhance infiltration. Many invasive plant populations are enhanced by fires. Additionally, these populations may facilitate the establishment of other invasive species (Brooks & Pyke 2001).

Federal land management agencies within the U.S.A. may elect to rehabilitate federal lands after wildfires provided burned locations meet one of two criteria, they are threatened by potential soil erosion or by dominance and spread of invasive plants (Anonymous 2002). Although these agencies cannot use these funds to conduct complete restoration projects, they are strongly encouraged to use native plants when available to accomplish rehabilitation objectives (e.g., Executive Order 1312 signed by President Clinton, 03 February 1999). Richards, Chambers & Ross (1998) determined on average that between 1984 and 1996 one native plant was sown for every four introduced species in wildfire rehabilitation projects on Bureau of Land Management (BLM) lands within the state of Nevada. Since native species have been sown on federal rehabilitation projects, we attempted to use a retrospective study to determine how representative the findings of Richards *et al.* (1998) were for the Great Basin as a whole.

3.1 Method

BLM offices throughout the Great Basin may be referred to as Districts, Resource Areas, or Field Offices and during the time period of interest may have been consolidated or split; however, we refer to them as Field Offices in this study. The Great Basin is represented by 19 BLM field offices that cover 35.3 million ha of four states: Idaho, Nevada, Oregon, and Utah. Each field office manages various sized units of public land dominated by semiarid vegetation. Many of these plant communities are susceptible to invasion and dominance by one or more exotic annual grasses, such as cheatgrass. To provide adequate dispersion of samples through the region of inference, two field offices per state were randomly selected for study (Oregon – Burns and Vale; Idaho – Burly and Jarbidge; Utah – Cedar City and Fillmore/Richfield; Nevada – Ely and Winnemucca). The selected field offices represented approximately 51 % (18.2 million ha) of the BLM land area in the region.

Within each field office, we examined all wildfire rehabilitation plans for projects implemented between 1988 (when monitoring was first funded with fire rehabilitation funds) to 1999. Prior to 1988, monitoring was infrequently conducted due to budgetary and personnel constraints. We examined the number and composition of introduced and native plant species seeded in rehabilitation projects over time. For ease of consistent classification of plant origin, we used the PLANTS database definitions for native (naturally occurring on lands defined by the current boundaries of the U.S.A. and its territories in the late 1400's) and introduced (naturally occurring outside lands defined by the current boundaries of the U.S.A. and its territories in the late 1400's) species (USDA, NRCS 2002). Invasive species were introduced species that successfully establish themselves in, and then overcome, otherwise intact, pre-existing native ecosystems (Invasive Species Specialist Group http://www.issg.org/ 11 December 2002).

The number of species sown per project was averaged for each year within each field office. Composition of native species within a seed mixture was calculated for each project by dividing the bulk seed density of native species by the bulk seed density of all seeds in the project. BLM project plans provided the data for pounds of bulk seed of each species sown in a project. To obtain the numbers of bulk seed, we used the number of seeds per unit weight for each species found in VegSpec version 3.1 (http://plants.usda.gov and follow the link to VegSpec, 15 December 2001). We recognize that bulk seed density does not provide an accurate estimate of viable seed densities, but since many projects did not report either germinability, viability or the number of pure live seeds sown per species, bulk seed density was the best estimate of seed density composition. Within each year and field office, the mean seed density composition for native species was determined by averaging the seed density composition of the natives among all projects.

All trends over time were tested using a mixed model analysis of variance with year as a repeated measure. The twelve years of data were collapsed into six, two-year periods to reduce the number of dependent estimates made per independent field office below the number of replicated field offices. All models were initially tested on a full quadratic equation and 14 potential covariance structures. The best covariance structure was selected using Akaike's Information Criterion (AIC) using the smaller-is-better form in SAS Version 8.0 PROC MIXED (SAS 1999). If a simpler covariance structure was within 2.0 AIC units, then we selected the simpler model; otherwise, we used the model with the smallest AIC value.

To examine the establishment success of native and introduced species sown on BLM rehabilitation projects, we randomly selected three rehabilitation projects per field office for conducting additional field monitoring. Potential projects had to meet the following selection criteria: (1) that monitoring plans were implemented in 1995 or earlier; and (2) that projects had 3 or more native species in the seed mixture. When fewer than three projects met our criteria, we used all projects with monitoring and native plants. Twenty-one projects were investigated. At each monitoring location within a project, we used a line-point intercept technique (Herrick *et al.* 2002, 2003) to determine canopy coverage by species. Species were grouped into native, introduced or invasive plants. Coverage by group was averaged across all monitoring locations within a project. Plant group coverage was converted to relative cover (composition) and averaged. All composition values were transformed using the arcsine square-root transformation, and then average composition and a 5 % confidence limit for each plant group (N = 21) was calculated for the region.

3.2 Results & Discussion

Numbers of fires per unit area in the Great Basin increased over 12 years of the study from 7 to 14 per million ha. Simultaneously, the mean size of fires increased from 56 to 235 ha. The increase in fire numbers and size are thought by some to be related to the increase in the fuel loads from the spread and dominance of exotic annual grasses (Pellant 1990; Whisenant 1990). The increase in fires has lead to an increase in rehabilitation projects within the region from 180 to 720 per million ha.

The number of species sown in rehabilitation projects has remained constant between 4 and 5 per project with 1 to 2 of those species being natives. Our findings agree with those of Richards *et al.* (1998) in Nevada and extend the area of inference to include the three surrounding states in the Great Basin. The three most constantly sown native species among the rehabilitation projects were two shrubs, *Atriplex canescense* (fourwing saltbush), and *Artemisia tridentata ssp. wyomingensis* (Wyoming big sagebrush), and one forb, *Linum lewisii* (Lewis flax). For introduced species, however, they were two forbs, *Medicago sativa* (dryland alfalfa) and *Sanguisorba minor* (small burnett), and a grass, *Agropyron desertorum* (desert wheatgrass). Although the number of native species has remained constant, the proportion of the bulk seed mass that was native has increased steadily over time (Figure 3). If we assume that the seed mass of native and introduced species are similar, then this would indicate that the number of native seeds sown by the BLM has increased in rehabilitation mixtures over time. Unfortunately, we were unable to determine levels of pure live seed that might add greater support to this finding (Pellant & Monsen 1993).
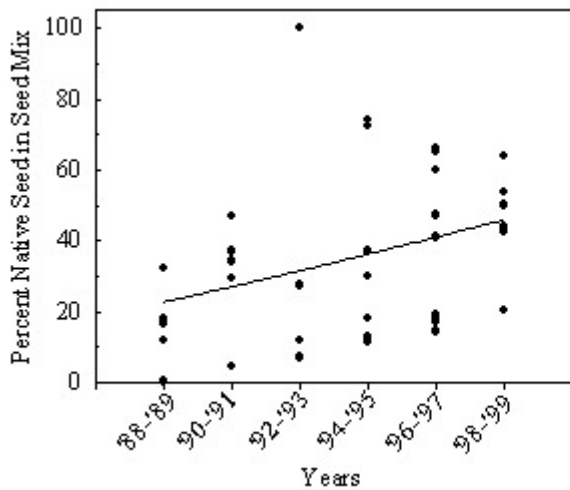
Figure 3. Percent native seed (based on bulk seed mass) used in rehabilitation seed mixtures. Each point represents the mean for a field office during a two-year period and the line represents the best-fit model for the repeated measures analysis.
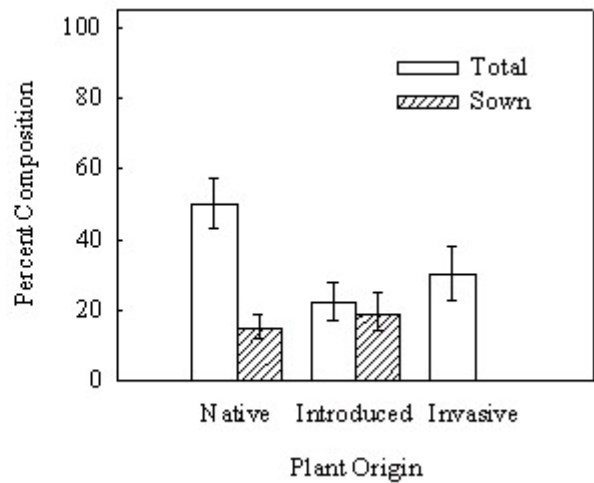
Figure 4. Mean plant composition (± 1 SE) (N = 21) by plant origins (see text for definitions) on Bureau of Land Management rehabilitation projects at least 5 years after seeds were sown within the Great Basin, U.S.A.

Although natives were typically sown in lower proportions than introduced species, we found that native plant composition in established rehabilitation stands (> 5 years post-rehabilitation) was significantly ($P < 0.05$) higher than introduced composition on BLM projects in the Great Basin (Figure 4). This result was unexpected since it is commonly believed that introduced species establish and compete with invasive plants better than native species (Pyke 1996, Roundy & Biedenbender 1995). Nearly two-thirds of the native plant composition came from unsown species whereas nearly all of the introduced composition was comprised of species sown in the rehabilitation project. For those species that were sown, the composition did not differ significantly between the native and introduced species in the established rehabilitation community. The most constantly established native species among rehabilitation projects were two grasses, *Pseudoroegnaria spicata* (bluebunch wheatgrass) and *Achnatherum hymenoides* (indian ricegrass), and a shrub, Wyoming big sagebrush, whereas the most constant introduced species were grasses, desert wheatgrass, *Agropyron cristatum* (crested wheatgrass), and *Thinopyron intermedium* (intermediate wheatgrass). Invasive plant composition did not differ significantly from either native or introduced composition. Unfortunately, unseeded control sites within rehabilitation project areas were not designated in BLM projects, therefore we were unable to determine if the goal of reducing the spread of invasive species on individual projects or within the region was met through rehabilitation.

4. CONCLUSIONS

Dobson, Bradshaw & Baker (1997) encouraged restoration ecologists to identify natural ecosystem processes in disturbed, rehabilitated, and intact ecosystems and to use that knowledge to enhance remediation of disturbed lands. Invasive species are degrading ecosystems and causing widespread environmental and economic consequences (Pimentel 2002). Within the Great Basin of the U.S.A., invasive annual grasses are fueling wildfires that convert diverse native ecosystems into near monocultures impacting other land uses and values (agricultural, recreational, and species survival). We are using a combination of process-based and retrospective studies to enhance the land manager's ability to rehabilitate and restore diverse communities of sagebrush grasslands.

Through studies of decomposition and nutrient cycling mechanisms, we hope to enhance native plant restoration while reducing the impacts of exotic annual grasses such as cheatgrass. We are continuing studies on bacterial and fungal diversity associated with decomposing litter and with soils. We are initiating studies to examine nematode and protozoa activity in soils at invaded and native sites. Simultaneously, we are examining the impacts of various forms of carbon applications on the microbial community, on nutrient cycles, and on the competitive relationships between native species and exotic annual grasses.

Our finding that native and introduced plant composition on wildfire rehabilitation projects did not differ even though introduced species contribute greater composition of the sown seed mixture was unexpected. We believe this provides hope that native species composition will increase in rehabilitation projects as additional amounts and species of seed become available. However, our finding that unsown native species contributed the greatest amount of composition in rehabilitation projects leads to recommend the need for two types of controls in future projects. The first would control for both the addition of seeds in a rehabilitation seed mixture and for the seeding equipment itself (no seed applied and

no equipment disturbing the soil). The second would control only for the seed sown on the land, therefore seed drills and soil disturbance equipment would be used on the land, but no seed would be applied. These controls would allow determinations of the need for rehabilitation and of the impact of seeding equipment on natural recovery.

We believe that coordinated studies of ecosystem processes and restoration-rehabilitation technologies will aid us in determining degradation states that currently exist on rangelands (Milton *et al.* 1994). Early degradation states may only require management changes for ecosystem recovery to occur, whereas later states will require vegetation manipulations or in the severest cases, modifications of the physical environment. Until we gain a more complete understanding of the relationships among ecosystem processes and rehabilitation/restoration successes our ability to prescribe treatments will remain limited. Through the combination of management level rehabilitation-restoration projects and ecosystem-based research studies, we hope to elucidate factors to aid our understanding of ecosystem change and our ability to repair of degraded lands.

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