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RESEARCH ARTICLE

Restoration of Native Plant Communities after Road Decommissioning in the Rocky Mountains: Effect of Seed-Mix Composition on Vegetative Establishment

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Abstract

Road decommissioning is increasingly recognized as a critical first step in the restoration of terrestrial and aquatic habitats. However, crucial gaps in knowledge exist about the efficacy and ecological effects of road-removal practices. One particularly important issue is the effectiveness of post-road-removal revegetation practices. This study evaluated (1) the short-term effects of road decommissioning on plant community composition, (2) the effects of seed-mix origin, species diversity, and seeding density on vegetative establishment, and (3) the impact of overstory canopy cover and coarse woody debris (CWD) on revegetation success on recently decommissioned roads. One year after decommissioning, total vegetative cover (on the former roadbed) declined by 60%, with non-native plants showing the greatest magnitude of response (circa 90% decline). Although the use of non-native seed is often justified by the need for rapid vegetative establishment on disturbed sites, we did not find significant differences in percent cover of total vegetation between plots seeded with native versus non-native species. As expected, cover of native species was significantly higher in plots seeded with natives compared to those seeded with non-natives (12.3 vs. 7.8%, respectively). Furthermore, in plots seeded with native species, 43% of total vegetative cover was due to cover of seeded species; in comparison, non-native seeded species accounted for only 18% of total vegetative cover in non-native plots. Cover of seeded species was not significantly impacted by overstory canopy cover or CWD. These findings suggest that native seed mixes may outperform non-native seed mixes in terms of vegetative establishment after disturbance associated with road removal.

Key words: **native plant restoration, native seeding, revegetation, road obliteration, seed establishment.**

Introduction

Roads are a primary source of ecosystem degradation in wildland settings (Forman et al. 2003). Habitat fragmentation and alteration of hydrologic processes caused by road networks directly impact wildlife (Mladenoff et al. 1995), fish (McCaffery et al. 2007), and water quality (Luce et al. 2001), and increase invasion by non-native plants (Tyser & Worley 1992; Gelbard & Belnap 2003). To mitigate these adverse effects, the United States Forest Service (2008*a*) is decommissioning a substantial number of roads. In 2008 alone, the Forest Service decommissioned 2,200 km of its network of circa 600,000 km of National Forest System roads. Removal of roads can reduce erosion and sedimentation, contributing to the restoration of disturbed ecosystems (Madej 2001; Switalski et al. 2004). However, there is little information available about the efficacy of conventional road-removal practices for achieving ecological restoration goals in general or restoration of native plant communities in particular (Grace 2000; Eschenbach et al. 2007). Areas of concern include the composition of seed mixes used after road decommissioning and lack of adequate light for germination.

A main objective of revegetation treatments is to quickly establish vegetation in order to deter erosion (Robichaud et al. 2000). Non-native species are often favored, in part because of the availability of relatively inexpensive seed of species that are thought to have rapid rates of growth and establishment (Maynard & Hill 1992), but also because of the lack of widespread availability and current high cost of native seeds. For instance, 65% of national forests in the Rocky Mountains and Pacific Northwest currently include non-native species in their seed mixes (Grant et al. unpublished data). However, even though commonly seeded non-native species may have rapid establishment rates, they may not provide enough cover in the short term to significantly reduce soil erosion (Robichaud et al. 2000). Furthermore, seeding with non-native species increases the potential for their invasion into the surrounding landscape, resulting in a cascade of ecosystem effects (Robichaud et al. 2006), including

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disruption of nutrient cycling (Milchunas & Lauenroth 1995), fire regimes (Mack & D'Antonio 1998), and other ecosystem functions (Roundy 2005). In comparison, revegetating with native species may reduce the risk of invasion by non-native plants while providing essential habitat and forage for native wildlife (Bugg et al. 1997). The U.S. Forest Service is increasingly interested in supporting efforts to replace non-native seed mixes with native ones (Landis et al. 2005) and recently adopted a national native plants policy that requires "selection of genetically appropriate plant materials (based) on site characteristics and ecological settings" (USDA Forest Service 2008*b*, p. 8). Given changing attitudes about the importance of using native plant material, it is imperative to determine which species will effectively establish on recently decommissioned roads in order to improve restoration practices (Cotts & Redente 1991; Elseroad et al. 2003).

Another issue with current revegetation practices is that conventional seed mixes have low species and functional richness. Most mixes include relatively few species and often only one life-form (graminoid; although nitrogen-fixing forbs are sometimes included) (Petersen et al. 2004). Seeding with a larger pool of species and life-forms may result in more species-rich communities and may facilitate the establishment of characteristic native plant communities (Tilman 1997). Furthermore, there is some evidence that plant communities with high species and life-form diversity may be more resilient to disturbance (Walker et al. 1999) in general, and invasion by non-native species in particular, due to the negative relationship between species diversity and resource availability (Tilman 1997, 1999). Given that abundance of invasive non-native species may increase with changes in disturbance regimes (Huston 2004), planting diverse species mixes may be especially important after disturbance associated with road removal.

Currently, managers pile dense coarse woody debris (CWD) on the former roadbed to deter erosion (USDA Forest Service 2005). Although low levels of CWD may create microsites suitable for establishment and survival of understory plants (Nelson & Halpern 2005), high quantities can obstruct light and inhibit seedling germination and establishment (Elseroad et al. 2003). In addition, seedling germination and establishment may be adversely affected by low-light conditions often found along decommissioned forest roads due to high overstory canopy cover from the surrounding forest. To counteract low establishment rates, managers often increase seeding density. However, it is unclear whether increasing seeding density actually improves revegetation success, although it does increase the cost of revegetation efforts. Information about appropriate seeding densities and effects of CWD and overstory canopy cover on vegetation establishment will help managers improve revegetation treatments on recently decommissioned roads.

We compared the efficacy of native and non-native seed mixes for revegetating recently decommissioned roads in forests of the Rocky Mountain region. Specifically, we addressed the following questions: (1) What are the short-term effects of road decommissioning on plant community composition? (2) How does seed-mix origin (native vs. non-native), diversity (3 vs. 6 species), and density (16.8 vs. 33.6 kg/ha) affect vegetation establishment? and (3) Does vegetation establishment vary with the amount of CWD or overstory canopy cover?

Methods

Study Area

This study was conducted on 13 road segments: 10 on the Kootenai National Forest's (KNF) Three Rivers Ranger District in northwestern Montana and three on the Clearwater National Forest's (CNF) Powell Ranger District in northeastern Idaho (Fig. 1; Table 1). During 2007, the KNF conducted road-removal projects in four watersheds on the Three Rivers Ranger District, all of which contain mesic, Pacific-maritime-influenced forests in *Thuja plicata*/*Clintonia uniflora* (western red cedar/bead lily) or *Abies lasiocarpa*/*Clintonia uniflora* (subalpine fir/bead lily) habitat types at elevations of 975–1,482 m (Table 1). Annual precipitation in the KNF averages 962 mm; during our study (2007–2008), mean annual precipitation was 917 mm (USDA Natural Resources Conservation Service 2009). Within the four watersheds selected for road decommissioning, projects occurred on the 10 road segments with the highest erosion potential; all segments had culverts (conduits used to allow water to pass underneath roads) that were undersized.

During the 2007 field season, the CNF decommissioned roads in one watershed in the Powell Ranger District. To ensure that the native species selected for seeding would be compositionally appropriate for both forests, we subjectively selected three sites in the CNF that had overland flow of water onto the roadbed and similar plant communities to the study sites in the KNF: *T. plicata*/*C. uniflora* or *A. lasiocarpa*/*C. uniflora* habitat types. In the CNF, annual precipitation averages 1,180 mm; during our study period, annual average precipitation was 1,150 mm (USDA Natural Resources Conservation Service 2009).

Pre-Treatment Data Collection

From 28 June 2007 to 22 August 2007 at each of the 13 road segments scheduled for decommissioning, we established a 63×7–m belt transect along the roadbed adjacent to a culvert removal area (Fig. 2), at least 1.5 km from the start and at least 1.5 km from the end of the segment scheduled for decommissioning. Within this belt transect, we installed seven 7×9–m experimental plots (Fig. 2a). Percent cover of understory vegetation was measured by visual estimation (vertical projection) for all vascular plant species within one 1-m^2 subplot located at the center of each experimental plot (Fig. 2b). Overstory canopy cover was measured by spherical densiometer at the edge of each subplot in every plot (Fig. 2c) (Lemon 1956). Nomenclature follows Hitchcock and Cronquist (1973), except *Bromus marginatus*, which follows Cronquist et al. (1977). In addition to sampling vegetation in

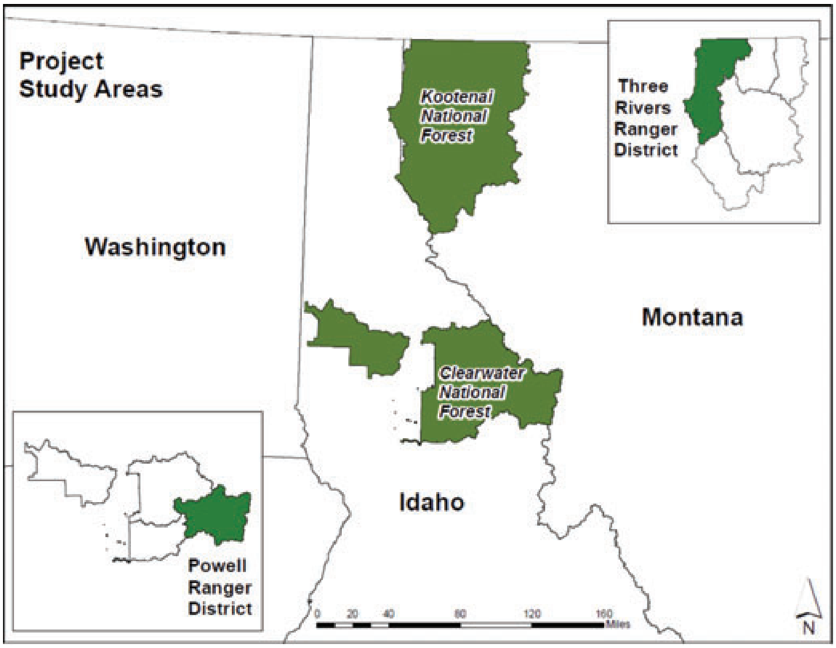

Figure 1. Map indicating the location of National Forests where the project was implemented. The upper and lower insets indicate the ranger districts where sites are located.

the roadbed, at each site we also visually estimated cover of vegetation within six 1-m^2 permanent subplots in the forest directly adjacent to the roadbed (hereafter "unroaded reference plots"): three located 10 m upslope and three 10 m downslope from the roadbed (Fig. 2d). Vegetation in these plots was characteristic of the forested matrix adjacent to the roadbed.

Road Decommissioning

From August to September 2007, after pre-treatment sampling, road segments were decommissioned. The KNF uses two methods to remove roads: full recontour and scarification. Selection of methods is based on the potential for soil erosion; typically, areas adjacent to culverts are recontoured while the rest of the roadbed is scarified. Recontouring entails moving soil that had been cast to the side during road construction back upslope onto the roadbed, reestablishing the original hillslope (Madej 2001), whereas scarification involves decompacting the top layers of soil (Luce 1997). In the CNF, every kilometer of roadbed removed is recontoured, due to highly erodible soils throughout this National Forest.

Experimental Seeding

Seeding of experimental plots was carried out in the autumn after road removal, to ensure cold stratification over the

Table 1. Characteristics of the 13 study sites, including national forest, aspect, slope, elevation, and habitat type.

		Site Characteristics			
National Forest	*Site*	*Aspect*	*Slope (degrees)*	*Elevation (m)*	*Habitat Type*
Clearwater	Rock Creek 1	SE	22	1,324	THPL/CLUN
Clearwater	Rock Creek 2	E	22	1,338	THPL/CLUN
Clearwater	Rock Creek 3	NE	19	1,353	THPL/CLUN
Kootenai	Beetle Creek 1	SSE	21	1,311	THPL/CLUN
Kootenai	Beetle Creek 2	NE	22	1,341	THPL/CLUN
Kootenai	Beetle Creek 3	N	12	1,379	THPL/CLUN
Kootenai	Gus Creek 1	NNW	9	975	THPL/CLUN
Kootenai	Gus Creek 2	NW	6	1,363	THPL/CLUN
Kootenai	Yodkin Creek 1	SE	52	1,282	THPL/CLUN
Kootenai	Yodkin Creek 2	W	24	1,478	ABLA/CLUN
Kootenai	Yodkin Creek 3	SW	28	1,482	ABLA/CLUN
Kootenai	Yodkin Creek 4	S	14	1,383	THPL/CLUN
Kootenai	Yodkin Creek 5	SW	21	1,382	THPL/CLUN

THPL, *Thuja plicata*; CLUN, *Clintonia uniflora*; ABLA, *Abies lasiocarpa*; CLUN, *Clintonia uniflora*; SE, Southeast; E, East; NE, Northeast; SSE, South-Southeast; N, North; NW, Northwest; NNW, North-Northwest; SW, Southwest.

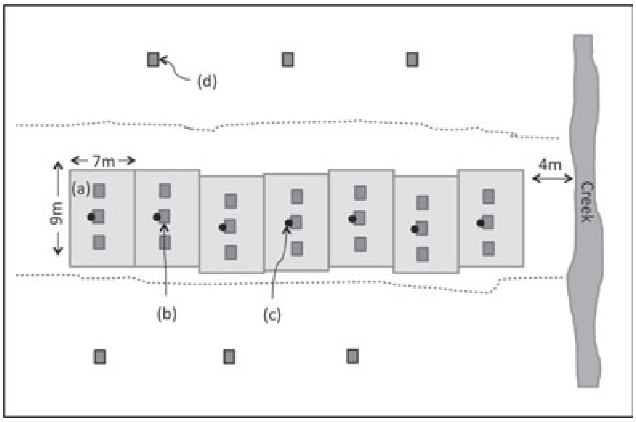

Figure 2. Sampling design at each of the 13 road segments. Within each 7×9–m experimental plot (a), we sampled vegetation cover within three 1-m^2 subplots (b) and overstory canopy cover at the edge of the middle subplot (c). We also sampled vegetative cover within 1-m^2 unroaded reference subplots adjacent to the roadbed (d). Drawing is not to scale.

winter and early exposure to spring precipitation to facilitate germination. The experimental seed mixes that we used varied by three factors, with two possible levels of each factor: seed origin (native vs. non-native), seed-mix diversity (three species [all graminoids] vs. six species [graminoids, forb, and shrubs]), and seeding density (low [16.8 kg/ha] vs. high [33.6 kg/ha]). Although there were four possible combinations of seed origin and seed-mix diversity, only three seed mixes were tested (Table 2). The non-native high-diversity combination was left out, due to concern about increasing the number of non-native species introduced during this investigation.

From 19 September 2007 to 10 October 2007, within each 63×9–m belt transect, we hand-seeded six of the seven 7×9–m experimental plots (Fig. 2a) with one of the six seed mixes and left one of the plots as an unseeded control. The assignment of seed treatment and control plots was random. During planting, seeds were incorporated into the soil using fine-tined garden rakes; graminoid species were incorporated to a depth of 1.26 cm and shrubs and forbs to a depth of 0.5 cm. Non-native seed mixes were only applied to the 10 sites located in the KNF.

Seed-Mix Composition

For our non-native treatment, we applied the seed mix commonly used for revegetation after road decommissioning in the KNF at the time of the study, using the same percentage of each species as in the mix used by the Forest (Table 2). For the native seed mix (Table 2), we selected species (1) that are present within *T. plicata*/*C. uniflora* and *A. lasiocarpa*/*C. uniflora* habitat types, (2) that are capable of establishing and surviving under harsh environmental conditions (as road decommissioning disturbs the developing soil organic layer on abandoned roadbeds, decreasing soil water-holding capacity), and (3) for which the seed was readily available. The three native perennial graminoid species that met these screening criteria and were selected for inclusion were *B. marginatus*, *Elymus glaucus,* and *Agrostis scabra*. *Bromus marginatus* and *E. glaucus* are frequently used in seed mixes on revegatation projects across the Northwest (Grant et al. unpublished data). *Agrostis scabra,* an early-seral bunchgrass, was included in our seed mix because it had the highest pre-treatment frequency of any native graminoid on both pre-treatment plots in the roadbed and unroaded reference plots (27.3% and 1.3% cover, respectively; Table 3). The seed for all graminoid species used in this project was collected in the KNF and stored at the US Forest Service's Coeur d'Alene Nursery (Idaho). The percentage of each species in the native seed mixes was determined based on species life-form and seed weight.

For the high-diversity seed-mix treatment, we selected six native species: the three graminoids included in the native low-diversity mix, as well as one forb (*Epilobium angustifolium)* and two shrubs (*Ceanothus velutinus* and *Arctostaphylos uva-ursi*) (Table 2). *Epilobium angustifolium* was selected because it is a common native forb with high seedling vigor that has the potential to rapidly colonize disturbed and coarse-textured soils (USDA Natural Resources Conservation Service 2008). *Ceanothus velutinus* is a native shrub that produces abundant seed and colonizes coarse-textured, disturbed soils

Table 2. Composition of the three experimental seed mixes, including percentage of each species within each mix.

Seed Mix	*Species Name*	*Percentage*	*Life-form*
Non-native, low diversity	*Dactylis glomerata* (Orchard grass)	20	Graminoid
	Festuca ovina (Sheep fescue)	20	Graminoid
	Lolium multiflorum (Italian ryegrass)	60	Graminoid
Native, low diversity	*Agrostis scabra* (Tickle-grass)	30	Graminoid
	Bromus marginatus (Mountain brome)	35	Graminoid
	Elymus glaucus (Blue wildrye)	35	Graminoid
Native, high diversity	*Agrostis scabra* (Tickle-grass)	21	Graminoid
	Arctostaphylos uva-ursi (Kinnikinnick)	20	Shrub
	Bromus marginatus (Mountain brome)	24	Graminoid
	Ceanothus velutinus (Buckbrush)	7	Shrub
	Elymus glaucus (Blue wildrye)	24	Graminoid
	Epilobium angustifolium (Fireweed)	4	Forb

For other details see *Methods* section.

Table 3. Mean frequency (percentage of experimental units within a site) and cover (± 1 SE) of native and non-native species that were seeded on experimental plots, pre-, and 1-year post-road decommissioning.

		Roadbed				Reference			
		Frequency, %		Cover, % (± SE)		Frequency, %		Cover, % (± SE)	
Species	Common Name	pre	post	pre	post	pre	post	pre	post
Native species									
Agrostis scabra	Tickle-grass	27.3	52.8	0.2 (0.1)	1.7	1.3	—	*t*	—
Arctostaphylos uva-ursi	Kinnikinnick	11.2	0.4	0.3 (−0.1)	*t*	2.6	1.3	0.4 (0.3)	0.2 (0.2)
Bromus marginatus	Mountain brome	3.5	53.5	0.1 (<0.1)	0.4 (<0.1)	—	—	—	—
Ceanothus velutinus	Buckbrush	—	—	—	—	1.3	—	*t*	—
Elymus glaucus	Blue wildrye	4.9	56.5	*t*	1.3 (−0.2)	—	1.3	—	*t*
Epilobium angustifolium	Fireweed	29.4	54.8	0.5 (−0.1)	1.4 (−0.2)	61.5	29.5	7.9 (1.9)	4.1 (1.1)
Non-native species									
Dactylis glomerata	Orchard grass	—	16	—	0.1 (<0.1)	2.6	—	*t*	—
Festuca ovina	Sheep fescue	—	17.9	—	0.3 (0.1)	—	—	—	—
Lolium multiflorum	Italian ryegrass	—	21.9	—	0.2 (0.1)	—	—	—	—

For mean cover <0.05, *t* indicates trace.

(Anderson 2001). It was selected because of its ability to fix soil nitrogen, enabling it to establish and survive in areas with low nutrient content (Anderson 2001). Once established, nitrogen-fixing plants can improve soil conditions, increase site productivity (Tillman 1985), and consequently deter soil erosion (Grace 2000). *Arctostaphylos uva-ursi* is a native low shrub that was included because of its capacity to grow on moisture-deficient sites with low nutrient levels (Klinka et al. 1989). In addition, this drought-tolerant plant can facilitate soil stabilization in disturbed areas (Crane 1991). *Ceanothus velutinus* and *A. uva-ursi* seed for this project was collected in the KNF and stored at the U.S. Forest Service's Coeur d'Alene Nursery. *Epilobium angustifolium* seed was collected at Glacier National Park (Montana) and provided by the Natural Resource Conservation Service's Bridger Plant Materials Center (Montana).

Tetrazolium tests for seed viability of all native seeded species were conducted at the Montana State Seed Testing Laboratory (Bozeman) (Association of Official Seed Analysts 2000). Viability was found to be above 90% for the three graminoids, circa 70% for *C. velutinus*, circa 50% for *A. uva-ursi*, and circa 30% for *E. angustifolium*.

Post-Treatment Data Collection

Data on 1-year responses to road removal and experimental seeding were collected from 25 June 2008 to 13 August 2008. Cover of vegetation and CWD were measured by visual estimation within three systematically located 1-m^2 subplots per experimental plot (Fig. 2b) and in the six 1-m^2 subplots in the unroaded reference areas that were measured pre-treatment (Fig. 2d). Overstory canopy cover was measured by spherical densiometer (Fig. 2c) (Lemon 1956).

Statistical Analysis

To evaluate the effect of road decommissioning (without subsequent seeding) on vegetation, we used paired *t* tests (Ott & Longnecker 2001) to test for differences in vegetation (total, native, and non-native) between post-treatment unseeded control plots and pre-treatment plots located in the proximity of the post-treatment controls, with separate tests for each vegetation response variable. It was not possible to install permanent subplots for sampling before and after treatment because of the substantial disturbance associated with road removal. However, the pre- and post-treatment data used for these analyses were collected from three 1-m^2 subplots in each sampling period (pre- and post-treatment), although the distance between subplots was greater in pre-treatment (2 m between subplots) than it was in post-treatment (1 m between subplots).

To evaluate the impact of seed-mix origin, diversity, and density on vegetative establishment, we used multifactor, univariate analysis of variance (ANOVA) models (Ott & Longnecker 2001). Two-way interactions (Origin × Density, Origin × Diversity, and Density × Diversity) were analyzed. Separate tests were conducted for cover of total vegetation, native vegetation, non-native vegetation, and seeded species. To evaluate the impacts of overstory canopy cover and CWD on vegetation, these factors were included as covariates in ANOVA models.

All statistical analyses were conducted using SPSS version 15 (SPSS 2006). Prior to analyses, variables were tested for normality using box plots, and residuals were evaluated with Q–Q plots. Levine's tests for homogeneity were conducted on all ANOVAs. An alpha level of 0.05 was used as the criterion for reporting statistical significance.

Results

A total of 108 species (92 native and 16 non-native) of vascular plants were found prior to treatment. The most common (>25% frequency) native species before treatment were *Alnus viridis* (10.8% cover), *Anaphalis margaritacea* (2.7% cover), and *Arnica latifolia* (1.0% cover). The most common

(>25% frequency) non-native species were *Hieracium aurantiacum* (2.5% cover), *Agropyron repens* (1.1% cover), and *Agrostis alba* (<1.0% cover). Of the native seeded species, *Epilobium angustifolium* and *Agrostis scabra* were present in more than a quarter of pre-treatment plots; *Arctostaphylos uva-ursi, Bromus marginatus* and *Elymus glaucus* were present, but less abundant; and *Ceanothus velutinus* was not present in roadbed sample plots but was found pre-treatment in unroaded reference plots (Table 3). None of the non-native seeded species were found in roadbed sample plots prior to road decommissioning (Table 3).

Short-Term Effects of Road Decommissioning on Plant Community Composition

After road removal, 111 vascular plant species were identified across all sites. The most common native species (>50% frequency) were the seeded species: *E. glaucus* (1.3% cover), *E. angustifolium* (1.4% cover), *B. marginatus* (<1% cover), and *A. scabra* (1.7% cover) (Table 3). The most common non-native species (>15% frequency) included *Spergularia rubra* (<1% cover) and two seeded species: *Lolium multiflorum* (<1% cover) and *Festuca ovina* (<1% cover) (Table 3). Disturbance associated with road removal had a significant impact on cover of total vegetation, native vegetation, and non-native vegetation, which all varied significantly from pre-treatment levels. In unseeded control plots, total cover and native cover declined by over 50% while non-native cover exhibited a 90% decrease, 1 year after decommissioning (Table 4).

Effect of Seed-Mix Origin, Diversity, and Density on Vegetation Establishment

Seed-mix origin did not affect cover of total or non-native vegetation, but did have a significant effect on cover of native vegetation ($df = 78$, $F = 0.915$, $p = 0.024$; Fig. 3a). Compared with plots seeded with non-native species, those seeded with natives had significantly higher native cover (12.3 vs. 7.8%, respectively; Fig. 3) and significantly lower non-native cover (1.4 vs. 3.0%, respectively; Fig. 3); however, cover of native vegetation did not vary between unseeded control plots and those seeded with natives.

Seed-mix origin also significantly affected cover of seeded species ($df = 78$, $F = 4.046$, $p = 0.001$): plots seeded with native species had higher total cover of seeded species than plots seeded with non-natives (Fig. 4). In plots seeded with native species, 43% of the total vegetative cover was due to cover of seeded species; in comparison, seeded species accounted for only 18% of total vegetative cover on non-native plots.

Neither seed-mix density nor diversity had a significant effect on any vegetation response variable (p values = 0.496–0.983 and 0.197–0.881, respectively; Fig. 3b & 3c).

Table 4. Pre- and post-treatment mean cover, and % change in cover, of total, native, and non-native vegetation. Degrees of freedom (df) and *p* values (p) are from paired t-tests.

			Cover (%)		
	df	*p*	*pre*	*post*	*% Change*
Total vegetation	12	0.001	34.1	13.6	−60
Native vegetation	12	0.013	26.0	12.8	−51
Non-native vegetation	12	0.014	8.0	0.8	−90

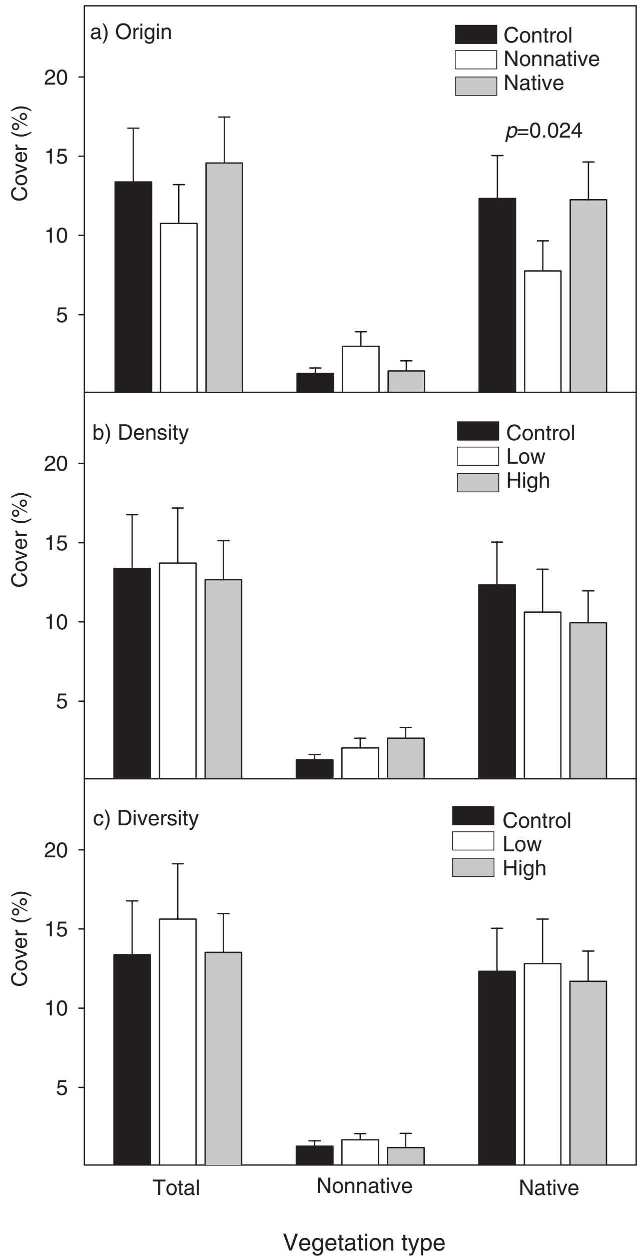

Figure 3. Mean percent cover (± 1 SE) of total, native, and non-native vegetation (seeded and nonseeded species) by main effects: (a) seed-mix origin (unseeded control, black shading; non-native seeded, white shading; native seeded, gray shading), (b) seed-mix density, and (c) seed-mix diversity (unseeded control, black shading; low-diversity mix, white shading; high-diversity mix, gray shading). *p* Values are only provided for significant differences.

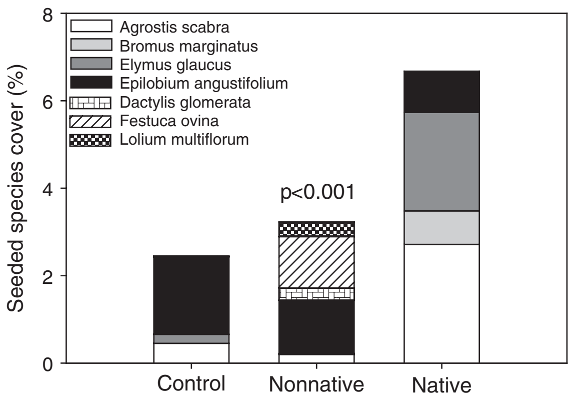

Figure 4. Cover of seeded species by treatment (unseeded control, non-native seed mix, and native seed mix). Bars with solid shading represent native species (see legend) included in seed mixes; hashed and stippled bars represent non-native species (see legend) included in seed mixes. *Arctostaphylos uva-ursi* and *Ceanothus velutinus* were not present at levels that could be visually displayed.

In addition, there were no significant two-way interactions between experimental factors (Origin × Density, Origin × Diversity, or Density × Diversity; $p = 0.173–0.989$).

Relationship between Vegetation Cover and CWD or Overstory Canopy Cover

CWD and overstory canopy cover did not have a significant impact on cover of total ($p = 0.469$ and 0.056, respectively), native ($p = 0.945$ and 0.058, respectively), non-native ($p = 0.971$ and 0.151, respectively), or seeded vegetation ($p = 0.526$ and 0.352, respectively).

Discussion

Short-Term Effects of Road Decommissioning on Plant Community Composition

As expected, disturbance associated with road decommissioning significantly reduced the cover of vegetation on former roadbeds 1 year after treatment. However, a substantial amount (circa 40%) of vegetation survived the disturbance. Although both native and non-native plants declined after treatment, non-native vegetation showed the greatest magnitude of response (circa 90% decline). Numerous studies have shown that both roads and disturbance can increase the presence of non-natives (Hobbs & Huenneke 1992; Gelbard & Belnap 2003). Thus, we had expected that the former road network combined with the short-term disturbance associated with road removal would result in high rates of invasion by non-native plants. In contrast, non-native plants were present at less than 1% cover 1 year after treatment. Given the low levels of non-natives immediately after road decommissioning, this time period may be crucial for establishing native vegetation on highly disturbed former roadbeds, before non-natives have the opportunity to colonize.

Effects of Seeding and Seed-Mix Composition on Vegetation Establishment

Our results suggest that seeding may not be critical for increasing vegetative cover after road decommissioning, as there was no difference in the overall vegetation cover between unseeded and seeded plots. The similarity in total vegetative cover between seeded and control plots is due primarily to the fact that post-treatment cover of remnant vegetation was much higher than post-treatment cover of seeded plants (propagules from the soil seed bank or from surrounding mature plants could also contribute to the lack of differences in cover between seeded and unseeded plots; however, within our plots we saw little establishment of nonseeded species, other than *Epilobium angustifolium*). This relatively high level of intact vegetation on the roadbed after decommissioning suggests that soil in some areas of the roadbed may not have been effectively de-compacted. Residual compaction within these areas may have negatively impacted seedling establishment.

We did find, however, large differences in the performance of native versus non-native seed mixes. Native seeded species contributed almost half of the total cover of vegetation found in the plots where natives were seeded. In contrast, non-native seeded species provided less than one-fifth of total vegetative cover on the non-native seeded plots. Despite the fact that federal land managers often favor non-native seed mixes (Grant et al., unpublished data) due to their alleged rapid establishment rates, our results indicate that native seed mixes can result in faster vegetative establishment, and potentially stabilize soils and reduce erosion more than non-native mixes, 1 year after road decommissioning—the period of time when vegetation establishment is most critical for erosion hazard reduction (Robichaud 2005). However, the overall contribution of seeding to erosion control is debatable; in our sites, mean cover of total vegetation was only 13% regardless of seeding treatment—a value much lower than the 60–70% cover found to be necessary to prevent short-term erosion at numerous sites across the western United States (Robichaud et al. 2000). Other studies have also found relatively low cover of seeded species on former roadbeds: seeding with native species after roadbed scarification in northern Arizona (Elseroad et al. 2003) and in Wyoming's Teton National Park (Cotts & Redente 1991) resulted in 2.9% and 4.8–11.5% cover, respectively. Findings from these studies, along with our own, indicate that seeding alone may not eliminate erosion hazards 1 year after road decommissioning.

Although soil stabilization and erosion control are primary objectives of road decommissioning, establishing native plant communities on decommissioned roadbeds is also an identified desirable goal (USDA Forest Service 2005). Seeding with rapid-establishing early-seral native graminoid species may facilitate this process. In our sites, native seed treatments had both higher cover of native plants and lower cover of non-natives. Thus, their use may accomplish revegetation goals, without causing adverse effects commonly associated with non-native plants (Mack & D'Antonio 1998; DiTomaso 2000).

Although the native seed mixes resulted in greater vegetative establishment than did the non-native mix, we found large differences in performance among the six native species that were seeded. All three of the graminoid species had relatively high frequency and cover. However, *A. scabra*—an early-seral graminoid that is not typically included in revegetation seed mixes—had higher rates of establishment and cover than did either *E. glaucus* or *B. marginatus*, graminoids that are common in revegetation seed mixes. Although *A. scabra* was present before treatment and in post-treatment control plots, its abundance increased more than five-fold in plots where it was seeded. This suggests that including *A. scabra* seed in revegetation seed mixes could increase vegetation cover in our area. In contrast, *E. angustifolium* was present at similar abundance in plots in which it was *not* seeded as in plots where it was, indicating that it was able to effectively colonize on its own after road removal. The lack of difference in *E. angustifolium* cover between seeded and unseeded plots was likely due to multiple factors. First, it was present in greater than 25% of the roadbed plots prior to road removal, and many of the residual plants were able to persist, particularly on scarified plots. Second, although this species has a short-lived seed bank, it is an effective colonizer due to wind-dispersed seeds; its high abundance in the surrounding area likely enabled it to rapidly colonize our sites (Dale 1989). Another factor contributing to the similarity in *E. angustifolium* abundance between seeded and unseeded plots is that the viability of our *E. angustifolium* seed was relatively low (circa 30%). Thus, it is possible that some of the observed lack of response was due to poor seed quality. For native plants that are both appropriate for revegetation and commonly occur in a project area, such as *E. angustifolium* in our sites, managers should consider potential for natural dispersal to the decommissioned site when evaluating the need for species inclusion in seed mixes.

This study was not effective in testing the effect of seed-mix diversity on vegetative establishment. The two shrubs that were included in the high-diversity native seed mix, *C. velutinus* and *A. uva-ursi,* did not establish in any plots, 1 year after treatment. The lack of establishment of these shrub species, coupled with the results discussed above for *Epilobium*, reduced the effective diversity of the high-diversity treatment to that of the low-diversity treatment. Determining the causes of low rates of establishment of these shrubs is difficult, as there are multiple variables that may have impacted germination and survival. For *C. velutinus*, seed viability (71%) was likely not an issue. In addition, sowing seeds in the fall should have resulted in exposure to the moist cold conditions necessary for seed stratification, which is required for *Ceanothus* germination (Anderson 2001). Seed predation by rodents may have occurred, but we have no evidence that this was a factor. In contrast, there are several plausible explanations for low establishment rates for *A. uva-ursi*, including its low seed viability (47%). In addition, *A. uva-ursi* is commonly endomycorrhizal (Crane 1991); this association may have been lacking in the disturbed soils after road decommissioning. Both *C. velutinus* and *A. uva-ursi* can germinate from the soil seed bank years after dispersal (Anderson 2001; Crane 1991). Thus, it is possible that the seed included in the experimental mix may result in greater establishment of these species in the future. However, both these species have slow rates of growth and, even if they do establish, they will likely remain at low abundance for many years.

Effect of Seeding Density on Vegetation Establishment

One year after road decommissioning, we found no difference in vegetative cover between high- and low-density treatments. Although policies are in place to increase the use of native plants (USDA Forest Service 2008*b*), budget limitations continue to restrict their use (Robichaud et al. 2006). Our data suggest that it may be possible to reduce the cost of using native seed by decreasing seeding density. The high-density treatment that we tested utilized the standard seeding density employed by the National Forests where our sites are located (USDA Forest Service 2005); our low-density treatment used half the seed of the high-density application. Despite this large difference in seed application rate, seeding density did not affect any vegetation response variable. Thus, cutting the current seeding density in half could halve the cost of seed for revegetation projects. Determining appropriate seeding densities should be a high priority in future studies.

Relationship between Understory Vegetation Cover and CWD or Overstory Canopy Cover

Competition for light is a key driver of plant community assembly (Tilman 1985), and light limitation on former roadbeds has been found to inhibit vegetation establishment after road decommissioning (Elseroad et al. 2003). Although slash left after management treatments can substantially reduce light availability, the level of CWD on our sites after treatment was low enough that it likely did not inhibit light or growing space for seedling establishment, possibly explaining the lack of observed relationship between slash and vegetation response. In contrast, overstory canopy cover did vary substantially among our sites (circa 5–90%); however, it also did not significantly affect post-treatment vegetative cover on our sites.

Conclusions

Given the large amount of money being spent on revegetation, and the high cost of developing and using native plant materials, it is critical to determine the efficacy of native species for revegetating decommissioned roads and other highly disturbed sites. Our results indicate that native seed mixes can be more effective than non-native ones in facilitating the establishment of vegetation 1 year after road decommissioning. These results, although short term, are significant given that the first year after disturbance is a critical time period for erosion control—a driving factor in many revegetation projects. The extent to which treatments vary with respect to longer-term vegetation responses remains to be seen with future years of monitoring on these and other sites. Longer-term observations are also needed to assess the extent to which seeded species may compete with or facilitate the establishment of colonizing native and non-native species, the dispersal of seeded species into the surrounding landscape, and the resistance and resilience of treatments to climatic stressors. In this study, we assessed the efficacy of only a limited number of native plants within only two plant associations: *Thuja plicata*/*Clintonia uniflora* and *Abies lasiocarpa*/*Clintonia uniflora*. There is a need for information on the performance of a wider variety of species in these, as well as other forest, grassland, and riparian habitat types. In addition, given that seeds are often planted into areas that have been sprayed with herbicides to control non-native, invasive species (Rice et al. 1997), it would be valuable to know how commonly used, native revegetation species tolerate herbicides. The United States has invested $180 million over the last 3 years (FY08–FY10) for road removal and restoration on federal lands (United States Congress 2008, 2009*a*,*b*). With new national native plant policies in place on federal land (USDA Forest Service 2008*b*), now is the time to identify which native species can best contribute to the restoration of roaded landscapes.

Implications for Practice

- Using native seed mixes, rather than conventional non-native ones, may result in increased cover of total and native vegetation and decreased cover of non-natives.
- Vegetative establishment from seed mixes may not substantially reduce erosion, 1 year after disturbance.
- Although it may not eliminate short-term erosion hazards, seeding with native species may facilitate the establishment of native plant communities before non-native, invasive plants become established.
- Current seeding amounts may be higher than necessary; cutting current seeding densities in half could substantially reduce revegetation costs without reducing vegetative cover.
- Managers should experiment with seeding a wider variety of native species in order to identify those that maximize the development of site-adapted native plant communities and best compete with non-native, invasive species.

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LITERATURE CITED

Anderson, M. D. 2001. Ceanothus velutinus. In: Fire Effects Information System, [Online]. United States Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. (http://www.fs.fed.us/database/feis/) accessed 18 March 2009.

Association of Official Seed Analysts. 2000. Tetrazolium testing handbook (Contribution no. 29). J. Peters, editor. Association of Official Seed Analysts, New Mexico.

Bugg, R. L., C. S. Brown, and J. H. Anderson. 1997. Restoring native perennial grasses to rural roadsides in the Sacramento Valley of California: establishment and evaluation. Restoration Ecology **5:**214–228.

Cotts, N. R., and E. F. Redente. 1991. Restoration methods for abandoned roads at lower elevations in Grant Teton National Park, Wyoming. Arid Soil Research and Rehabilitation **5:**235–249.

Crane, M. F. 1991. Arctostaphylos uva-ursi. In: Fire Effects Information System (online). United States Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (http://www.fs.fed.us/database/feis/) accessed 19 March 2009.

Cronquist, A., A. H. Holmgren, N. H. Holmgren, J. L. Reveal, and P. K. Holmgren. 1977. Monocots: intermountain flora: vascular plants of the intermountain west, U.S.A. Vol. 6. Columbia University Press, New York.

Dale, V. H. 1989. Wind dispersed seeds and plant recovery on Mount St. Helens debris avalanche. Canadian Journal of Botany **67:**1434–1441.

DiTomaso, J. M. 2000. Invasive weeds in rangelands: species, impacts, and management. Weed Science **48:**255–265.

Elseroad, A., P. Fulé, and W. Covington. 2003. Forest road revegetation: effects of seeding and soil amendments. Ecological Restoration **21:**180–185.

Eschenbach, E. A., R. Teasley, C. Diaz, and M. A. Madej. 2007. Decision support for road decommissioning and restoration by using genetic algorithms and dynamic programming. Pages 359–368 in R. B. Standiford, G. A. Giusti, Y. Valachovic, W. J. Zielinski, and M. J. Furniss, editors. Proceedings of the Redwood Region Forest Science Symposium: What Does the Future Hold? General Technical Report PSW-GTR-194, United States Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California.

Forman, R. T. T., D. Sperling, J. A. Bissonette, A. P. Clevenger, C. D. Cutshall, V. H. Dale, et al., editors. 2003. Road ecology: science and solutions. Island Press, Washington, D.C.

Gelbard, J., and J. Belnap. 2003. Roads as conduits for exotic plant invasions in a semi-arid landscape. Conservation Biology **17:**420–432.

Grace, J. M. 2000. Forest road sideslopes and soil conservation techniques. Journal of Soil and Water Conservation **55:**96–101.

Hitchcock, C. L., and A. Cronquist. 1973. Flora of the Pacific Northwest. University of Washington Press, Seattle.

Hobbs, R. J., and L. F. Huenneke. 1992. Disturbance, diversity, and invasion: implications for conservation. Conservation Biology **6:**324–337.

Huston, M. A. 2004. Management strategies for plant invasions: manipulation productivity, disturbance, and competition. Diversity and Disturbances **10:**167–178.

Klinka, K., V. J. Krajina, A. Ceska, and A. M. Scagel. 1989. Indicator plants of Coastal British Columbia. University of British Columbia Press, Vancouver.

Landis, T. D., K. M. Wilkinson, D. E. Steinfeld, S. A. Riley, and G. N. Fekaris. 2005. Roadside revegetation of forest highways: new applications for native plants. Native Plant Journal **6:**297–304.

Lemon, P. 1956. A spherical densiometer for estimating forest overstory density. Forest Science **2:**314–320.

Luce, C. H. 1997. Effectiveness of road ripping in restoring infiltration capacity of forest roads. Restoration Ecology **5:**265–270.

Luce, C. H., B. E. Rieman, J. B. Dunham, J. L. Clayton, J. G. King, and T. A. Black. 2001. Incorporating aquatic ecology into decisions on prioritization of road decommissioning. Water Resources Impact **3:**8–13.

Mack, M. C., and C. M. D'Antonio. 1998. Impacts of biological invasions on disturbance regimes. Trends in Ecology and Evolution **13:**195–198.

Madej, M. A. 2001. Erosion and sediment delivery following removal of forest roads. Earth Surface Processes and Landforms **26:**175–190.

Maynard, A. A., and D. E. Hill. 1992. Vegetative stabilization of logging roads and skid trails. Northern Journal of Applied Forestry **9:**153–157.

McCaffery, M., T. A. Switalski, and L. Eby. 2007. Effects of road decommissioning on stream habitat characteristics in the South Fork Flathead River, Montana. Transactions of the American Fisheries Society **136:**553–561.

Milchunas, D. G., and W. K. Lauenroth. 1995. Inertia in plant community structure state changes after cessation of nutrient-enrichment stress. Ecological Applications **5:**452–458.

Mladenoff, D. J., T. A. Sickley, R. G. Haight, and A. P. Wydeven. 1995. A regional landscape analysis and prediction of favorable gray wolf habitat in the northern great-lakes region. Conservation Biology **9:**279–294.

Nelson, C. R., and C. H. Halpern. 2005. Edge-related responses of understory plants to aggregated retention harvest in the Pacific Northwest. Ecological Applications **15:**196–209.

Ott, R. L., and M. Longnecker. 2001. An introduction to statistical methods and data analysis. Duxbury Press, Pacific Grove, California.

Petersen, S. L., B. A. Roundy, and R. M. Bryant. 2004. Revegetation methods for high-elevation roadsides at Bryce Canyon National Park, Utah. Restoration Ecology **12:**248–257.

Rice, P. M., J. C. Toney, D. J. Bedunah, and C. E. Carlson. 1997. Plant community diversity and growth form responses to herbicide applications for control of *Centaurea maculosa*. Journal of Applied Ecology **34:**1397–1412.

Robichaud, P. R. 2005. Measurement of post-fire hillslope erosion to evaluate and model rehabilitation treatment effectiveness and recovery. International Journal of Wildland Fire **14:**475–485.

Robichaud, P. R., J. L. Beyers, and D. G. Neary. 2000. Evaluating the effectiveness of postfire rehabilitation treatments. General Technical Report GTR-63. United States Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado.

Robichaud, P. R., T. R. Lillybridge, and J. W. Wagenbrenner. 2006. Effects of postfire seeding and fertilizing on hillslope erosion in north-central Washington, USA. CATENA **67:**56–67.

Roundy, B. A. 2005. Plant succession and approaches to community restoration. Pages 43–48 in N. L. Shaw, M. Pellant, and S. B. Monsen, editors. Sage-grouse habitat Restoration Symposium Proceedings. RMRS-P-38. United States Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado.

SPSS. 2006. SPSS 15.0 for Windows. SPSS Inc., Chicago, Illinois.

Switalski, T. A., J. A. Bissonette, T. H. DeLuca, C. H. Luce, and M. A. Madej. 2004. Benefits and impacts of road removal. Frontiers in Ecology and the Environment **2:**21–28.

Tilman, D. 1985. The resource-ratio hypothesis of plant succession. The American Naturalist **125:**827–852.

Tilman, D. 1997. Community invasibility, recruitment limitation, and grassland biodiversity. Ecology **78:**81–92.

Tilman, D. 1999. The ecological consequences of changes in biodiversity: a search for general principles. Ecology **80:**1455–1474.

Tyser, R. W., and C. A. Worley. 1992. Alien flora in grasslands adjacent to road and trail corridors in Glacier National Park, Montana. Conservation Biology **6:**253–262.

United States Congress. 2008. Legacy Roads and Trails Remediation Initiative. Consolidated Appropriations Act (H.R. 2764). Pages 286–287.

United States Congress. 2009*a*. Omnibus Appropriations Act for Fiscal Year 2009. Interior Chapter: Forest Service Capital Improvement and Maintenance Section (16 U.S.C.). Pages 532–538.

United States Congress. 2009*b*. Department of the Interior, Environment, and Related Agencies Appropriations Act, 2010. Title III, Department of Agriculture, Forest Service, Capital Improvement and Maintenance (H.R. 2996-37). Page 37.

USDA Forest Service. 2005. Road decommissioning program: monitoring report 2005 excerpt. United States Department of Agriculture Forest Service, Clearwater National Forest, Orofino, Idaho.

USDA Forest Service. 2008*a*. Roads accomplishment report and bridge accomplishment report. Unpublished Report, United States Department of Agriculture Forest Service, Washington Office, Washington, D.C.

USDA Forest Service. 2008*b*. Native Plant Material Policy Forest Service Manual 2070. United States Department of Agriculture Forest Service. RIN 0596–AC44.

USDA Natural Resources Conservation Service. 2008. National Plant Data Center (available from http://plants.usda.gov) accessed 7 February 2008.

USDA Natural Resources Conservation Service. 2009. National Water and Climate Center (available from http://www.wcc.nrcs.usda.gov/snow/) accessed 15 February 2009.

Walker, B., A. Kinzig, and J. Langridge. 1999. Plant attribute diversity, resilience, and ecosystem function: the nature and significance of dominant and minor species. Ecosystems **2:**95–113.