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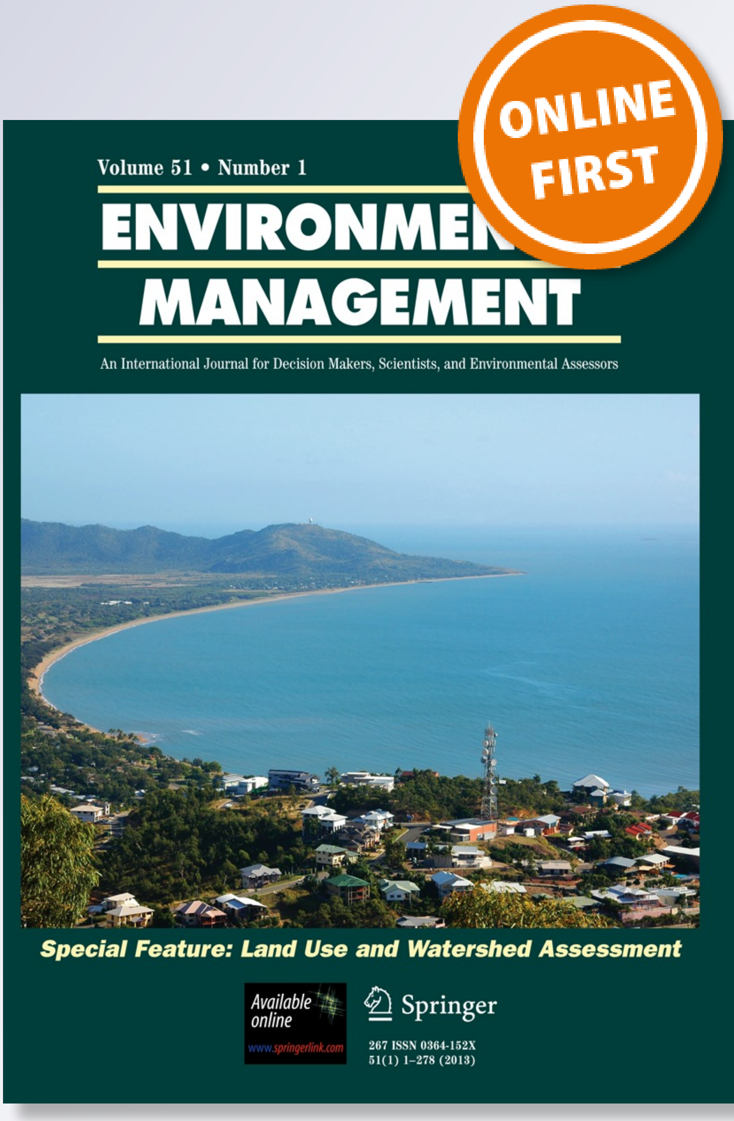
Environmental Management

ISSN 0364-152X

Environmental Management
DOI 10.1007/s00267-012-0015-3



Springer

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Long-Term Effectiveness of Restoration Treatments on Closed Wilderness Campsites

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Received: 7 March 2012 / Accepted: 26 December 2012

Abstract This study assessed long-term recovery of vegetation on six wilderness campsites in subalpine forests in Oregon that were closed to use and that received common restoration treatments (scarification, soil amendments, mulch, transplanting, and seeding). Vegetation cover was assessed every year for the first 7 years following treatment, as well as 10 and 15 years after treatment. This made it possible to compare long-term treatment effectiveness to short-term efficacy. Plots that were closed and not scarified had virtually no vegetation cover even after 15 years without use. If long-used campsites in these subalpine forests are simply closed and allowed to recover on their own, restoration of undisturbed conditions will require hundreds if not thousands of years. Study results show, however, that simple treatments can accelerate recovery rates substantially. Scarification and transplanting were highly effective treatments, with seeding and soil amendment with organic matter and compost also contributing to success, but to a lesser degree. The use of a mulch mat, in contrast, had no effect, either positive or negative. Assessments of success conducted within the first few years of treatment overestimate treatment efficacy, particularly the effectiveness of soil amendments and seeding.

Keywords Compost · Recreation impacts · Scarification · Seeding · Soil amendments · Transplanting

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Introduction

While the objective of ecological restoration is usually long-term improvement in site conditions, the success of restorative treatments is typically assessed over short periods, usually no more than a few years. This raises questions about how effective treatments are in the long term and the degree to which short-term assessments predict long-term effectiveness. One situation where ecological restoration for the long term is practiced is in mountainous wilderness areas in the United States where the impacts of camping can be severe. On campsites, understory vegetation disappears, as do organic soil horizons. Exposed mineral soil becomes highly compacted, reducing rainfall infiltration and increasing erosion (Cole and Fichtler 1983). Recovery of closed campsites can be a slow process, particularly in subalpine ecosystems where growing seasons are short and climatic conditions are harsh (Stohlgren and Parsons 1986). Many abundant subalpine plants establish infrequently and grow slowly (Eriksson and Fröberg 1996). Moreover, the soil on many closed sites no longer provides a good growing medium for plants. Many decades without vegetation and with minimal organic input have left soils physically, biologically, and chemically impoverished (Zabinski and others 2002).

High-elevation restoration studies, mostly conducted in alpine ecosystems, show that transplanting (Conlin and Ebersole 2001) and seeding (Smyth 1997) can accelerate recovery processes. Moreover, success can be increased using such treatments as mulches (Petersen and others 2004) and organic amendments (Chambers 1997). However, the few studies conducted on wilderness campsites suggest limited success. On degraded subalpine campsites in Yosemite National Park, California, only 19 % of transplants survived after 3 years and those that survived

Published online: 16 January 2013

had not spread (Moritsch and Muir 1993). Given the lack of information on long-term effectiveness of restoration treatments on subalpine campsites, experiments were implemented on closed campsites. Treatment effects were closely followed for 15 years, making it possible to also assess how conclusions about treatment efficacy vary with the length of study. The treatments assessed were (1) scarification, (2) transplanting and seeding with local, native species, (3) ameliorating microclimatic conditions with a mulch mat, and (4) amending soils with organic matter, compost, and soil inoculum.

Study Sites

The study was conducted on six campsites in the Eagle Cap Wilderness, Wallowa Mountains, a subrange of the northern Rocky Mountains in northeastern Oregon. All sites were located at an elevation of 2,215–2,300 m, adjacent to subalpine lakes, about 12–15 km from the closest road. They were located in forests with an overstory of *Abies lasiocarpa*, *Pinus contorta*, and *Pinus albicaulis*. Ground cover vegetation on adjacent, little-disturbed sites is discontinuous (typically about 50 % cover). Ericaceous dwarf shrubs, *Vaccinium scoparium* and *Phyllodoce empetriformis*, and the caespitose graminoids, *Juncus parryi* and *Carex rossii*, are the most abundant species (Johnson 2004).

This plant community type occurs throughout much of the western United States at high elevations, particularly in locations that are popular destinations for wilderness recreation. The impacts of wilderness camping are probably more common in this community type than any other in the United States, making information about effective restoration techniques particularly useful in this type. Soils are shallow, sandy, and acidic (pH between 4.2 and 4.8) and are derived from granitic substrates (Cryochrepts and Cryorthents). Although snow typically covers the ground until late June/early July, snowmelt is typically followed by hot, dry summers. The frequency of summer thunderstorms varies from year to year. When they are infrequent, soils can be highly droughty for several months (most of the growing season).

These campsites probably exhibited high levels of impact (lack of vegetation, minimal soil organic horizons, and compacted mineral soil) for at least 50 years prior to closure. Prior to restoration, these campsites were typically about 200 m^2 in size, with about 100 m^2 completely devoid of vegetation. Soil organic horizons had eroded away over substantial portions of these sites and mineral soils were so compacted that infiltration rates were reduced by almost 50 percent (Cole and Fichtler 1983). Organic C (carbon), total N (nitrogen), NH_4 (ammonium), potentially mineralizable N, microbial biomass C, and several indicators of the carbon utilization capabilities of the microbial community were also substantially reduced on these campsites (Zabinski and others 2002; Cole and Spildie 2007).

Methods

Treatments

Campsite restoration began in 1995 with the closure of these sites to camping, using closure signs and rope. The closures were highly effective, although one site was camped on between years 10 and 15. A three-factor experiment, using a split-plot design was employed (Fig. 1). Twelve treatment plots (1.5 by 1.5 m) were established on each campsite. The soil was scarified on these plots. Scarification involved the use of shovels, picks, pitchforks, hoes, and hand kneading to break up compaction and clods to a depth of about 15 cm. A control plot (not treated in any way) was established within the closed area on a part of the campsite that was not scarified. This plot was used in an analysis, separate from the factorial experiment, of the effect of scarification in the absence of mulch, soil amendments, and planting.

Of the three factors in the split-plot experiment, the mulch treatment was the factor used to establish whole plot units because it was most feasible to apply mulch blankets over large areas. Six contiguous plots on each site were covered with biodegradable mulch made of straw interwoven with cotton string and jute (Bionet®). The other six contiguous plots were not mulched (Fig. 1). Within each of

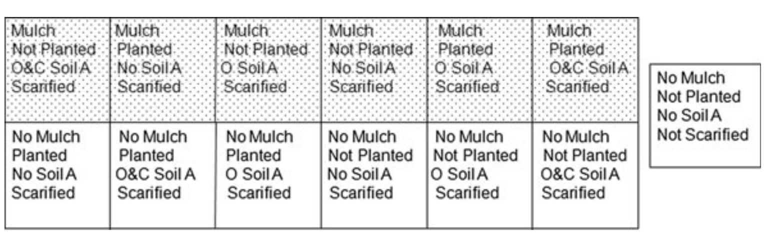

Fig. 1 Distribution of treatments for one campsite, illustrating the separate non-scarified control and random assignment of treatments (planted or not; and no soil amendment, organic amendment, or organic and compost amendment) within whole plots (mulched or not)

the two mulch whole plots, three levels of soil amendment and two levels of planting were assigned to split-plot units in a completely randomized design. Each combination of soil amendment and planting occurred in each whole plot. Each campsite had a unique ordering of treatments within the mulch whole plots and provided one of six replicates.

There were three levels of the soil amendment factor. Within each whole plot, two treatment plots (split-plot units) received no amendments. Another two plots were amended with organic matter and inoculated with native soil. The organic matter was a mix of peat moss (20 %) and well-decomposed, locally collected organic matter. The dry peat moss was mixed with water before application. A 2.5-cm layer of this organic material was mixed with mineral soil to a depth of 7.5 cm. Soil from the rooting zone of local transplants was the source for the inoculum. About 1.2 L of soil was mixed with about 20 L of water to make a slurry. Three liters of this slurry were sprinkled over each plot and raked into the soil. The final two plots were amended with compost in addition to the organic matter and inoculum treatment. We added 2.5 cm of commercially available compost (sewage sludge/log yard waste compost with a C:N ratio by weight of approximately 20:1; Ekocompost®, Missoula, Montana), lightly watered and raked into the top 10 cm of organic and mineral soil.

The two levels of planting were planted and unplanted. Within each whole plot, three plots were planted (seeded and transplanted) and three were not (Fig. 1). Seeding involved (1) collecting seed locally from several species with mature seed, (2) dividing available seed into equal quantities for each seeded plot, (3) pinch-broadcasting seed over the plot, and (4) raking seed into the upper 2.5 cm of soil. Seeded species and quantity of seed varied between campsites depending on locally available plants with mature seed. *Juncus parryi* and *Phleum alpinum* were seeded on three of the campsites. *Antennaria alpina*, *Antennaria lanata*, and *Sibbaldia procumbens* were seeded on two campsites. *Aster alpigenus*, *Danthonia intermedia*, *Penstemon globosus*, and *Sitanion hystrix* were each seeded on one campsite. One of the campsites was not seeded due to a lack of mature seed in the vicinity.

Transplanting involved (1) digging up enough transplants in the vicinity to plant equal numbers of each species in each plot, (2) digging a hole and placing transplants in the hole, along with Vita-start (vitamin B-1) to reduce transplant shock, and (3) giving each transplant 0.6 L of water. Plots not planted were given an equivalent amount of water. Most transplant plugs were between 5 and 25 cm in diameter, and most plots received five to six plugs. Most plugs contained only 1 species, but some contained more. Transplanted species varied between campsites. *Vaccinium scoparium* and *J. parryi* were intentionally transplanted on five of the six campsites. *P. empetriformis, C. rossii*, *Luzula hitchcockii*, and *S. procumbens* were intentionally transplanted on two campsites. Species that were intentionally transplanted on only one campsite were *Abies lasiocarpa, Achillea millefolium*, *A. alpina*, *A. lanata, A. alpigenus*, *Calamagrosts canadensis*, *D. intermedia*, *Hypericum formosum, Oryzopsis exigua, P. contorta, Polemonium pulcherrimum*, and *Spiraea betulifolia.* Thirteen other species were unintentionally included in plugs. Nomenclature follows Hitchcock and Cronquist (1973). The mean cover of transplants, immediately after planting, was 12 %. Shrub and graminoid species each accounted for 39 % of transplant cover, while forbs accounted for 19 % and tree species for 3 %.

Seeding and transplanting occurred only in the central 1 m^2 of each plot. Measurements were also confined to this central area, leaving a 0.5-m buffer between the measured portions of each treated plot. This reduced the potential for edge effects resulting from soil treatments. In 1996, when it appeared that soils were extremely dry, plots were watered several times. All plots were given an equal amount of water (about 2 L per plot). No supplemental watering occurred in later years.

Measurements

We measured transplant survival and growth and seedling density at least once every year for 7 years. Results based on those measurements are reported in Cole and Spildie (2006) and Cole (2007). The results reported here are based on measures of plant cover, ocularly estimated for the entire 1 m^2 plot, to the closest percent if the cover was 10 % or less and in 10 % increments thereafter. Total vascular plant cover was assessed, as was the cover of each vascular plant species. Because transplants were carefully mapped, we were able to distinguish the cover of transplants from that of plants that had established from seed. Seedlings included plants that grew from the seeding treatment, seed that dispersed onto the site, and possibly seed produced by transplants. Cover estimates were taken in September of every year between 1996 and 2002, as well as in September of 2005 and August of 2010, 10 and 15 years after seeding and transplanting.

Total vegetation cover and the cover of individual species were estimated in undisturbed plots near each restored campsite. The means from these six plots are used as targets for successful restoration.

Data Analysis and Presentation

Initially, a repeated-measures analysis of variance, appropriate for split-plot designs (using an autoregressive covariance structure, PROC MIXED in SAS 9.2) was used. When square-root transformed, data complied with assumptions about normality. In many cases, treatment effects varied significantly with time since treatment (i.e., interactions with time were significant). In these cases, treatment effects for each year of the experiment are described, but the significance of effects is only assessed at the end of the experiment, in 2010.

To assess the hypothesis that scarification has positive effects, the control (the plot that was not scarified) was compared to the one plot on each campsite that was scarified but not mulched, amended, or planted. For the three factors included in the split-plot design (mulch, soil amendment, and planting), main effects of each factor and interactions among factors were assessed. Interactions among the treatments were never statistically significant. Therefore, only the main effects of treatments are reported. For the soil amendments, Tukey–Kramer tests, adjusted for multiple comparisons, were used to assess differences between the amendment treatments.

Results

In the 15 years that followed campsite closure and restoration, mean total vascular plant cover increased from 0 to 15 % (Table 1). On planted plots, the immediate increase in cover resulting from planting exceeded the increase in cover that occurred over the subsequent 15 years. Regardless of treatment, cover peaked 4 years after treatment, declined for the next few years, and then increased to a new maximum in 2010. This response pattern was most pronounced on plots amended with organics and compost and for seeded species on planted plots, suggesting that the effect of soil amendments, particularly on seeded species (mostly graminoids), was most dramatic in the first few years following treatment. The largest change that occurred after the first year following treatment was a fivefold increase in the cover of species that volunteered (Table 1). Over the 15 years that followed the initial planting treatment, mean cover of transplanted species increased about 30 %, while seeded species cover was unchanged.

Treatment Effectiveness

Restoration treatments varied in effectiveness; some individual plots had no vegetation cover in 2010, while others had as much as 65 % cover. Mere closure was not successful. Mean cover on plots that were not scarified was only 0.5 % 15 years after closure. Scarification alone was significantly more effective (ANOVA, $F = 4.7$, $P = 0.02$); however, 15 years after treatment, mean cover on scarified-only plots was only 4 %.

The planting (both transplanting and seeding) treatment had the most pronounced effect on plant cover. Magnitude of effect varied significantly with year since planting (i.e., the interaction between planting and year was significant in a repeated-measures ANOVA). Consequently, the significance of treatment effects was assessed at the end of the study, in 2010 (Table 2). Fifteen years after treatment, mean (SE) plant cover was 21 (2) % on planted plots. This is more than twice the 9 (1) % mean cover found on plots that were not planted (Fig. 2a).

Soil amendments were also effective, but to a lesser degree (Table 2). Magnitude of effect varied with year since treatment, being least pronounced in the first 2 years

Table 1 Variation in the cover, mean (SE), of all plants, transplants, and seedlings over the 15-year study

	1996	1999	2002	2005	2010
Total plant cover	8.0 (1.0)	14.7 (1.6)	9.4 (1.0)	12.4 (1.3)	15.3 (1.4)
Transplants[a]	11.3 (0.8)	13.2 (1.1)	9.4 (0.9)	12.1 (1.3)	15.0 (2.4)
Trees	0.4 (0.1)	0.7 (0.2)	0.9 (0.3)	2.1 (0.8)	3.6 (1.5)
Shrubs	3.9 (0.6)	2.4 (0.4)	1.5 (0.3)	2.4 (0.6)	3.8 (1.0)
Graminoids	4.6 (0.5)	7.4 (0.8)	5.3 (0.5)	5.7 (0.6)	5.8 (0.5)
Forbs	2.3 (0.5)	2.7 (0.6)	1.7 (0.4)	1.9 (0.5)	1.8 (0.5)
Seedlings	3.9 (0.6)	11.0 (1.6)	6.0 (0.8)	7.3 (1.0)	10.0 (1.1)
Seeded species[a]	5.6 (1.0)	10.8 (2.3)	4.7 (1.1)	6.3 (1.8)	5.6 (1.7)
Volunteer species[b]	1.6 (0.4)	6.5 (1.2)	4.0 (0.6)	4.7 (0.8)	7.6 (1.4)
Trees	0.4 (0.0)	0.3 (0.0)	0.3 (0.1)	0.4 (0.1)	0.7 (0.2)
Shrubs	0.0 (0.0)	0.2 (0.0)	0.1 (0.0	0.2 (0.1)	0.1 (0.0)
Graminoids	2.3 (0.4)	8.8 (1.4)	4.2 (0.6)	5.0 (0.7)	7.5 (0.8)
Forbs	1.1 (0.2)	1.6 (0.4)	1.4 (0.4)	1.9 (0.6)	1.6 (0.5)

[a] For transplants and seeded plants, cover is the mean of those plots that were planted

[b] Volunteer seedling cover is derived from all plant cover on unplanted plots, as well as the cover of seedlings, of species that were not seeded on planted plots

Table 2 Analysis of variance results for the effects of planting, soil amendments, and mulching on plant cover in 2010

Effect	df	*F*	*P*
Mulch (M)	1	0.1	0.38
Error (whole plots)	5		
Planting (P)	1	60.7	<0.01
Soil amendment (S)	2	4.0	0.03
P*S	2	2.8	0.08
P*M	1	0.4	0.54
S*M	2	0.1	0.88
P*S*M	2	2.8	0.08
Error (within whole plots)	50		

following closure (Fig. 2b). Fifteen years after treatment, mean (SE) plant cover was 20 (3) % on plots that received organics and compost amendments, 15 (2) % on organics-only plots, and 11 (2) % on plots that received no amendments. The two amendment treatments differed significantly from the unamended treatment (adjusted Tukey–Kramer multiple comparison, $t = 4.6$, $P < 0.001$ and $t = 3.0$, $P < 0.04$), but not from each other (adjusted Tukey–Kramer multiple comparison, $t = 1.4$, $P = 0.09$). Mulching with a biodegradable mat did not have a significant effect on plant cover—over the entire length of the study or for 2010 (Table 2). After 15 years, mean (SE) cover both on mulched plots and on plots that were not mulched was 15 (2) %.

Transplants

The response of transplants differed from that of plants that established from seed, both in temporal patterns and in the effectiveness of restoration treatments. The mean transplant cover 10 years after planting (12 %) was similar to mean transplant cover immediately after planting (11 %); after 15 years, transplant cover had increased slightly to 15 % (Table 1). The magnitude of response varied among growth forms. The cover of small trees increased most; they had both high survival and growth rates. Forb cover was relatively unchanged over the 15 years. Graminoid cover increased greatly in the first years after planting, but was lower 15 years after planting than 4 years after planting. Shrubs had poor survivorship. Shrub transplant cover declined steadily for the first 7 years after planting; thereafter, it has increased steadily. *Vaccinium scoparium,* the shrub that comprises about 55 % of the groundcover on undisturbed sites, had a mean cover of 1.3 % immediately after planting. Seven years later, mean *V. scoparium* cover was just 0.6 %; thereafter, it increased and was 1.4 % after 15 years.

Fifteen years after planting, transplant cover did not vary significantly with any of the restoration treatments other than planting. Mean (SE) transplant cover on mulched plots was 16 (4) % compared to 14 (2) % on plots that were not mulched (ANOVA, $F = 0.2$, $P = 0.32$). Cover on plots amended with organics and compost was 20 (6) %, compared to 14 (3) % on plots amended with organics-only and 13 (3) % on unamended plots (ANOVA, $F = 1.0$, $P = 0.20$).

Seedlings

The cover of plants that established from seed was more temporally variable than the cover of transplants. Total seedling cover increased greatly for the first 4 years after treatment and then declined substantially before increasing again sometime between 7 and 10 years after treatment (Table 1). Graminoid seedling cover increased substantially over the 15 years of closure, while forb, tree, and shrub seedling cover did not increase much. Graminoids, many of which were seeded, were the only growth form to peak and then decline over the first 7 years of closure.

A repeated-measures ANOVA found that seedling cover varied significantly with seeding treatment and with soil amendments; however, effects varied between years. That is, treatment interacted with year. Consequently, treatment effects are reported for 2010 only (Table 3). The mulch had no effect in any year. The effect of seeding was positive, but apparently short-lived. After 15 years, mean seedling

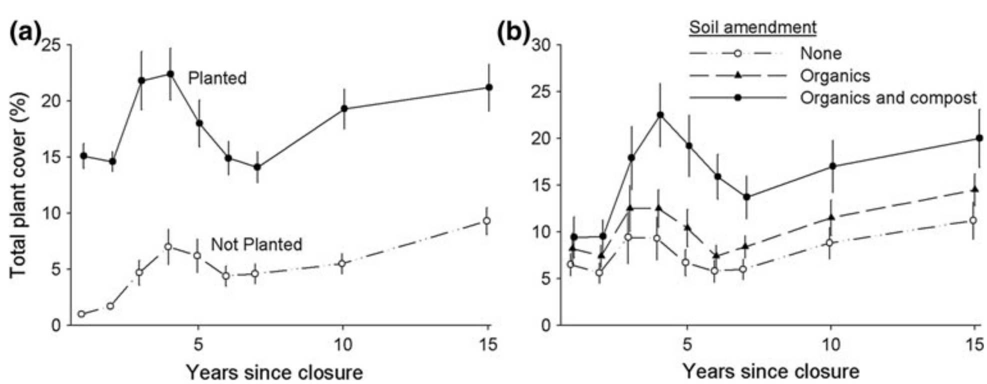

Fig. 2 Effects of **a** planting (transplanting and seeding) and **b** soil amendments on total vegetation cover (mean and SE)

Table 3 Analysis of variance results for the effects of seeding, soil amendments, and mulching on seedling cover in 2010

Effect	df	F	P
Mulch (M)	1	0.4	0.25
Error (whole plots)	5		
Seeding (SD)	1	0.3	0.28
Soil amendment (S)	2	6.7	<0.01
SD*S	2	0.7	0.49
SD*M	1	0.7	0.39
S*M	2	0.3	0.77
SD*S*M	2	1.3	0.32
Error (within whole plots)	50		

cover was higher on seeded plots than on plots that were not seeded (Fig. 3a), but this difference was not statistically significant (Table 3).

Effects of the soil amendments persisted 15 years after treatment (Table 3 and Fig. 3b). Fifteen years after restoration, plots that were amended with organics and compost or just with organics had significantly more seedling cover than plots without organic amendments (adjusted Tukey–Kramer multiple comparison, $t = 4.4$, $P < 0.001$ and $t = 2.2$, $P = 0.05$). Plots amended with organics and compost did not have significantly more seedling cover than organics-only plots (adjusted Tukey–Kramer multiple comparison, $t = -1.1$, $P = 0.15$).

Species Richness

The mean number of perennial vascular species per plot decreased with time since treatment, despite the steady increase in cover of seedlings that volunteered. Fifteen years after restoration, mean species richness at the 1 m^2 plot scale was four species, with a maximum of 11. In 2010, species richness on planted plots was significantly higher than on plots that were not planted (ANOVA, $F = 79.1$, $P < 0.01$) (Fig. 4). Species richness did not vary significantly with either mulching (ANOVA, $I = 1.8$, $P = 0.10$) or soil amendments (ANOVA, $F = 0.9$, $P = 0.22$). Species richness at the scale of the campsite also declined from a mean of 14 species the first season after treatment and a maximum of 15 species 3 years after treatment to nine species 15 years after treatment. None of the species that established were non-natives.

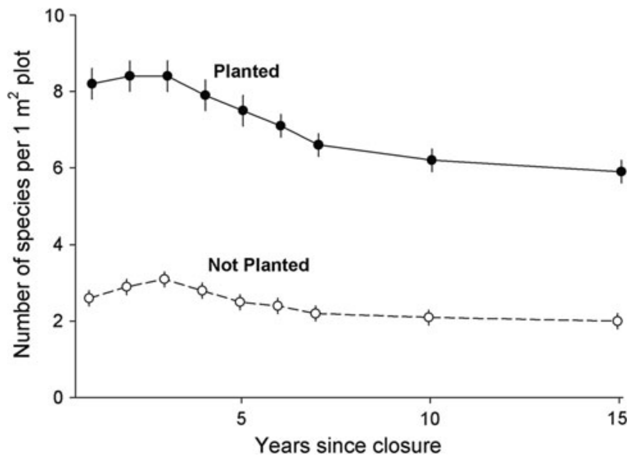

Fig. 4 Effect of planting on species richness

Conditions Relative to Reference Conditions

Progress toward the restoration of pre-disturbance vegetative conditions varied with treatment. On plots that received the most effective treatment (scarification, organic and compost amendments, and planting), mean vegetation cover 15 years after campsite closure was 28 %, slightly more than one-half of the 50 % cover typical of reference conditions. In contrast, on sites that were scarified but neither amended nor planted, mean cover was just 4 % after 15 years. In terms of vegetation composition, recovery rates were even slower. On the closed campsites, graminoids make up 69 % of the groundcover and shrubs account for 14 % of the cover. On undisturbed sites, graminoids make up 26 % of the groundcover and shrubs account for 57 % of the cover. Treatments were not highly effective in restoring native composition. Even on plots that were planted and amended with organics and compost, the vegetation was 19 % shrubs, 54 % graminoids, and 27 % forbs.

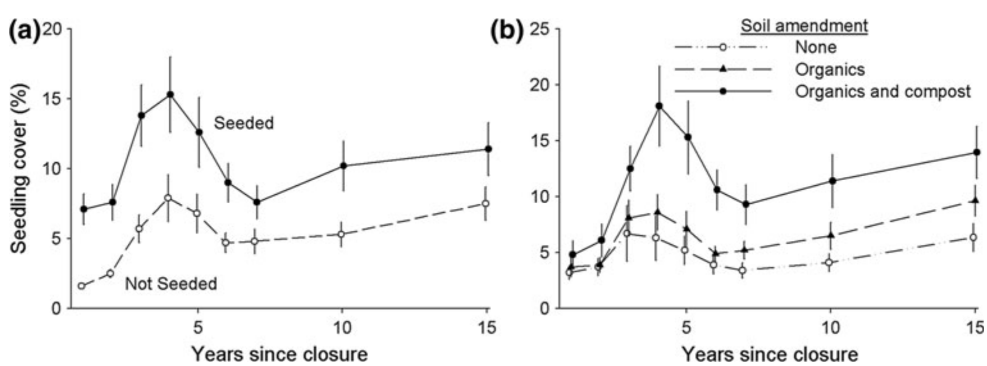

Fig. 3 Effect of **a** seeding and **b** soil amendments on seedling cover (mean and SE)

The species that established and grew most without assistance was *C. rossii*. Even on plots that were not planted, mean cover of *C. rossii,* after fifteen years of closure, was 6.8 %, much higher than the 1.5 % cover typical of undisturbed sites. No other species had cover on unplanted plots that approximated that species' cover on reference sites. Cover of most of the transplanted species, after 15 years, was at least as high on campsites as on reference sites. The important exceptions were the dominant shrub species, *V. scoparium* and *P. empetriformis*.

For two of the seeded forbs (*A. alpina* and *P. globosus*), cover was as high on campsites as on reference sites within 1 year after seeding and increased steadily over 15 years. For two other seeded forbs (*A. alpigenus* and *S. procumbens*), cover was as high on campsites as on reference sites 1 year after seeding, but declined steadily over 15 years. Cover of *A. lanata* increased steadily and, after 15 years, was nearly as high as on reference sites. Two seeded grass species (*D. intermedia* and *P. alpinum*) grew abundantly for several years after seeding, but were virtually absent after 15 years. Of the other seeded graminoids, *J. parryi* established abundantly and persisted, while *S. hystrix* never established with any abundance.

Discussion

Vegetative recovery following long-term camping disturbance in these environments is extremely slow. Plots that were closed and not scarified had virtually no vegetation cover even after 15 years without camping use. If long-used campsites in these subalpine forests are simply closed and allowed to recover on their own, restoration of undisturbed conditions will require hundreds if not thousands of years. Several other studies of closed campsites in subalpine forest have also reported slow rates of recovery, but trends were only observed for several years (Stohlgren and Parsons 1986; Moritsch and Muir 1993).

Longer studies have been conducted in the alpine zone. Willard and Marr (1971) estimated it might take centuries to "rebuild a natural and persistent ecosystem" in Colorado alpine tundra disturbed by trampling, and after 42 years plant cover was still declining, despite lack of recent trampling (Willard and others 2007). Scherrer and Pickering (2006) report that 15 years after closure, seeding and mulching, a trail through alpine herbfield in Australia had only partially recovered. Recovery times measured in the centuries, for subalpine dwarf shrub-dominated forests, are all the more problematic, given that only a few nights of camping can eliminate much of the groundcover (Cole and Monz 2003).

The results of this long-term study show, however, that simple treatments can accelerate recovery rates substantially. Scarification and transplanting were highly effective treatments, with seeding and soil amendment with organic matter and compost also contributing to success, but to a lesser degree. The use of a mulch mat, in contrast, had no effect, either positive or negative. Other studies have documented the effectiveness of transplanting as a means of increasing recovery rates in alpine (Brown and Johnston 1979; Bayfield 1980; Buckner and Marr 1990; Urbanska 1994; Conlin and Ebersole 2001; Bay and Ebersole 2006) and subarctic (Deshaies and others 2009) environments. Compared to the experience on subalpine campsites in Yosemite National Park, California, where only 19 % of transplants survived after 3 years (Moritsch and Muir 1993), our success with transplanting was high; about two-thirds of transplants survived (Cole and Spildie 2006) and, by 10–15 years after planting, transplant cover exceeded the cover of the original transplants. As has been found elsewhere (Brown and Johnston 1979; May and others 1982; Conlin and Ebersole 2001), transplant survivorship was particularly high for graminoids (Cole and Spildie 2006). Success with seeding of native species has also been reported in alpine environments (Urbanska and Schütz 1986; Chambers and others 1990; Smyth 1997), particularly among grass species.

Transplanting and seeding successes were increased by amending soils with organics and compost, but benefits were modest and dissipated somewhat over the years. We could not conclude with confidence that the addition of compost increased success, beyond the beneficial effects of adding organic matter and soil inoculum. This may reflect the low power of our research design; we only had six replicates. Cover was always substantially higher on plots that received compost, in comparison to organics-only plots. Plants were visibly more robust on composted plots and soil characteristics more closely approximated those on undisturbed sites (Cole and Spildie 2007).

The success of these treatments likely reflects the effectiveness of amendments in raising levels of organic C and potentially mineralizable N, and increasing microbial activity in soils where these characteristics had been adversely affected by decades of disturbance and lack of organic inputs (Zabinski and others 2002; Cole and Spildie 2007). Similar amendments have had positive effects on soil attributes and accelerated recovery elsewhere. For example, Legg and others (1980) found that incorporation of wood chips into soil on campsites prolonged the positive effects of scarification on soil bulk density. In Rocky Mountain National Park, Colorado, revegetation success has been increased with the addition of soil inoculum (Rowe and others 2009) and organic fertilizers (Paschke and others 2000).

The ineffectiveness of the mulch blankets was surprising, given that they are frequently employed and recommended (Urbanska and Chambers 2002) and have been found to be effective in research conducted in other high-elevation ecosystems (Petersen and others 2004). However, on alpine ski runs in the Alps, mulch blankets had no effect on transplant survival (Fattorini 2001). They may be more effective in inhibiting erosion and keeping visitors from trampling closed sites than in accelerating recovery rates.

Finally, by periodically assessing treatment effectiveness over 15 years, this study shows that assessments of success conducted within the first few years of treatment would have overestimated treatment efficacy, particularly the effectiveness of compost soil amendments and seeding. As reported by Cole and Spildie (2006), organic amendments did not increase transplant survival; however, the organics and compost treatment did have a positive effect on the size of transplants 2–6 years after treatment. This positive effect was not long lasting, however. Fifteen years after treatment, transplant cover did not differ significantly among the soil amendment treatments. Similarly, for each of the first 10 years following treatment, seeded plots had significantly more seedling cover than plots that were not seeded (Cole and Spildie 2007). Fifteen years after treatment, the cover of plants that established from seed did not differ between plots that were and were not seeded. Notably, the cover of volunteer seedlings exceeded the cover of seeded species between 10 and 15 years after seeding. Other studies have shown that conclusions about long-term treatment effectiveness can differ from short-term evaluations (e.g., Paschke and others 2005).

Management Implications

Given how slow recovery rates are in these subalpine forests, it is important to avoid campsite impacts in the first place, except where they are considered acceptable. Once impacts occur, restoration will be a lengthy and costly process. Where campsite closure and restoration are desirable, it will be necessary to accelerate recovery rates, so restoration requires decades rather than centuries. For this purpose, the following treatments can be recommended: (1) effective curtailment of use, (2) soil scarification, (3) organic soil amendments, (4) transplanting, and perhaps (5) seeding. Ropes and signs, instructing people to stay off the site and explaining why, were effective during our study. It is important to insure that there are other places to camp close by and that people know where they can go to camp.

Acknowledgments Funding and in-kind support for this project were provided by the Forest Service, US Department of Agriculture. I appreciate the field assistance of many Forest Service personnel, particularly Dave Spildie, and the statistical advice of Dave Turner.

Conflict of interest The author has declared no conflict of interest.

Ethical Standards Experiments comply with the current laws of the country in which they were performed.

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