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Department of Horticultural Science
University of Minnesota, St.Paul, MN

Cool Desert Cryptogamic Crust 10-40 Years Post-restoration: Limitations, Successes, and Challenges

Karin Kettenring

Introduction

Successful restoration of cool desert ecosystems relies upon reestablishing the cryptogamic crust community. Cryptogamic crusts are a soil community that consist of moss, lichens, fungi, green algae, and cyanobacteria (Marble and Harper 1989). The exact composition of the crust community varies with temperature, rainfall, soil chemistry, and vascular plant community structure (Rosentreter and Belnap 2001). Cryptogamic crusts confer numerous benefits to the cool desert ecosystem in the form of soil stabilization which limits water and wind erosion, increased water absorption due to the microtopography it creates, and increased soil fertility from nitrogen fixation (Johansen and St. Clair 1986).

This paper focuses on crust restoration in the southern cool desert region of Utah and northern Arizona. These cool deserts are characterized by warm summers and cool winters with mean average temperature of 4-13^{o}C (Rosentreter and Belnap 2001). Contrary to hot deserts found further south, these cool deserts experience frequent soil freezing in the winter (Rosentreter and Belnap 2001). The average yearly precipitation is 130-510 mm and most precipitation occurs in winter as snow (Rosentreter and Belnap 2001). The plant communities of the southern cool deserts vary from north to south along a temperature and precipitation gradient. The warmer and wetter, more southern part of this cool desert region – the Colorado Plateau including northern Arizona - is shrubland and woodland dominated by species in the genera *Coleogyne*, *Atriplex*, *Artemisia*, *Pinus*, and *Juniperus* (Rosentreter and Belnap 2001). Further north, the Great Basin region, which includes Utah, is sagebrush steppe and salt desert shrub communities and is dominated by perennial grasses, *Atriplex* sp., or *Artemisia* sp., depending on the area (Rosentreter and Belnap 2001).

Historically, cool desert ecosystems of North America experienced relatively little disturbance from grazing animals and thus, evolved so that the soil crust community is not resistant to such disturbances (Evans and Belnap 1999). Since the migration of Europeans to the American southwest, introduced domestic livestock, specifically sheep and cattle, have negatively impacted cool desert ecosystems through the loss of cryptogamic crust species and their associated role in ecosystem functioning (Memmott et al. 1998). The extent of ecosystem recovery following disturbance has only recently been brought to the forefront of cryptogamic crust restoration ecology research. In this paper, I will review a series of 5 studies conducted in the cool desert region of Utah and northern Arizona (see Figure 1) that assess the long-term recovery of cryptogamic crust following the cessation of grazing. These long-term restorations provide a unique opportunity to consider the success of current restoration efforts as well as provide insight into the direction future research should take. Thus, in this paper I will address the status of existing research, techniques used to assess restoration effectiveness, success 10-40 years post-restoration, the limitations of current restorations, techniques that can assist recovery, and what challenges lie ahead for cryptogamic crust restoration.

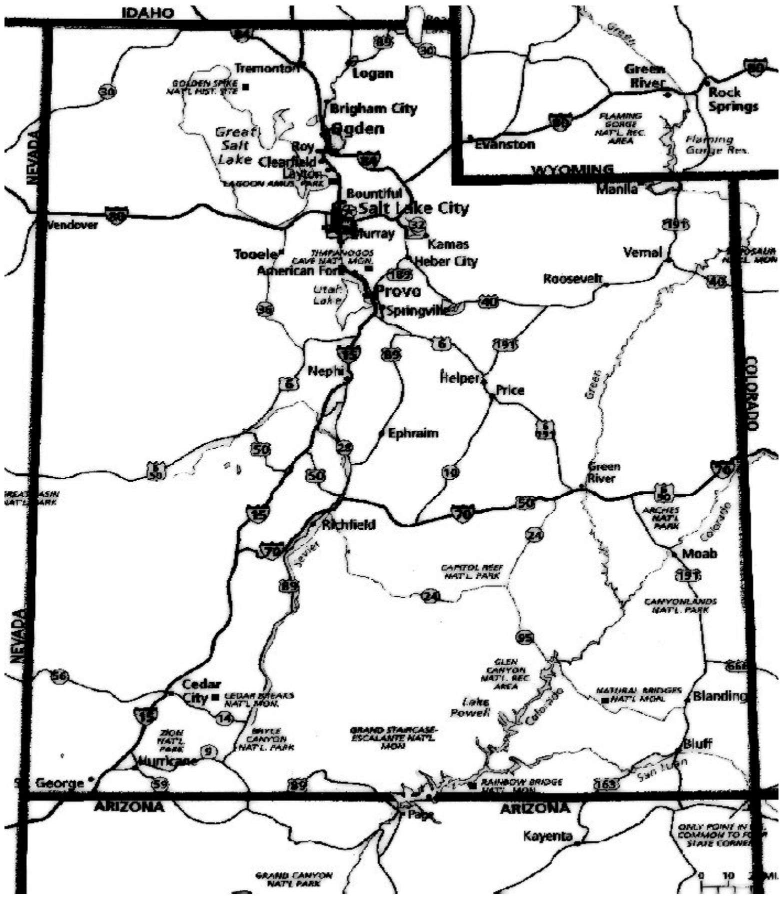

Figure 1. Utah with place names referenced throughout article.

Restoration research - status and findings

Initially, managers interested in attempting restoration of a disturbed ecosystem may focus on the success of ecosystem recovery through natural processes (e.g. natural recolonization of the vascular plants). This strategy is a logical first-step given that successful restoration through natural processes is the cheapest alternative. In the case of cool desert cryptogamic crust restoration, studies looking at long-term recovery have focused on restoration through natural recolonization after the cessation of disturbance. The following studies represent the major work that exists on long-term restoration following the cessation of grazing of cool desert cryptogamic crust. The restoration technique for all 5 studies was the same- grazing exclosures were established to delineate a park boundary and/or to prevent damage by cattle and sheep. However, none of these "restorations" were conducted with the intent of studying crust recovery but conveniently, however, these exclosures provide an opportunity to examine cryptogam restoration.

The first 4 of 5 studies I will describe are hallmark papers because they represent the first research ever conducted on long-term recovery of cryptogamic crust from grazing disturbance (K.T. Harper, personal communication). Prior to this time, the mere existence of cryptogamic crusts was largely ignored and the negative effects of grazing on these communities was not considered (K.T. Harper, personal communication). The final study I will describe is groundbreaking, as well, because it is the first long-term study considering the effects of grazing disturbance on ecosystem functioning (i.e. nitrogen dynamics).

Community recovery from grazing disturbances

The first study, conducted by Anderson, Harper, and Holmgren (1982a), explores the extent of crust community recovery through natural processes following the cessation of grazing. This study was conducted at the Desert Experimental Range (DER) of the USDA-Forest Service near Milford in western Utah (see Figure 1). The vascular community at this site consists of low shrubs and mostly *Hilaria* and *Bouteloua* grasses. The restorations were initially undertaken when exclosures were established on formerly grazed areas 40 years prior to the recovery monitoring. Exclosures were put in place by government employees interested in investigating grazing impacts on vascular plants (K.T. Harper, personal communication). Comparisons were made with adjacent disturbed areas that had continuous sheep grazing.

In this study, Anderson et al. (1982a) monitored combined lichen and moss coverage and density; average number of cryptogamic species per transect; and total number of species per treatment. They found that restored areas compared with continually grazed areas had higher combined lichen and moss cover (4.1 vs. 0.9%) and higher combined density of lichen and moss species (2.8 vs. 0.7 per $0.25m^2$). The average number of species per transect between restored versus currently grazed sites was higher for rock lichens (3.8 vs. 3.2), attached soil lichens (3.3 vs. 1.5), and moss (1.2 vs. 0.2); the same for filamentous algae (1.9); and lower for diatoms (6.7 vs. 7.2) and loose soil lichens (0.8 vs. 0.9). The total number of species per treatment was higher for attached soil lichens and moss for restored sites compared with currently grazed areas (6 vs. 5 and 3 vs. 1, respectively); the same for filamentous algae, rock lichens, and loose soil lichen between restored and currently grazed areas (5, 5, and 2, respectively); and lower for diatoms in restored sites than currently grazed sites (15 vs. 29). These results illustrate the significant recovery that occurred for combined lichen and moss coverage and density; number of rock lichen, attached soil lichen, and moss species per transect; and total number of attached soil lichen and moss species per treatment. Moreover, it is clear that some species are not affected by grazing (e.g. filamentous algae) or do better with grazing (e.g. diatoms).

A study published by Anderson, Harper, and Rushforth (1982b), investigates the recovery of cryptogamic crust communities on previously grazed sites. This study was conducted at eleven sites in salt desert shrub communities in the cool desert of central and eastern Utah (see Figure 1). To restore the sites, exclosures were established by federal and state employees on land previously grazed by domestic livestock (S. Rushforth, personal communication). This study is different from the previous one because it compares various sites that were currently grazed, 14-18 years post-restoration, or 37-38 years post-restoration; this set-up provides an opportunity to explore the "recovery curve" more in depth over the 40 year period. Because there is no specific information on pre-restoration conditions, comparisons with currently grazed sites were necessary (S. Rushforth, personal communication). At these sites, moss, lichen, algal, and total cryptogamic cover was assessed; the contribution of algae, lichen, and moss to the total percent cover of cryptogamic crust was assessed; and density of lichen and moss species was determined although information on particular species is not available (Anderson et al. 1982b).

In their study, Anderson et al. (1982b) found that the total cover of moss, lichen, and algae combined was significantly higher in restored sites (29% for sites 14-18 years post-restoration and 33% for sites 37-38 years post-restoration) compared with 12% for currently grazed sites. More specifically, algal cover was approximately 14.5% in sites 14-18 years post-restoration and about 17% in sites 37-38 years post-restoration compared with 8.5% in currently grazed sites. Also, percent moss cover was about 90% higher (6.00 vs. 0.61) and lichen cover was about 60% higher (9.42 vs. 3.18) in restored sites than currently grazed sites (specific information for the individual time-since-restoration treatments was not available). The algal cover as a percent of total cryptogamic cover declined from 82 to 57 to 56% at sites 0, 14-18, and 37-38 years post-restoration, respectively, indicating that algae contribute much more to the total crust cover initially, with decreasing importance over time. Moss cover as a percent of total cryptogamic cover increased from 7 to 21 to 24% at sites 0, 14-18, and 37-38 years post-restoration, respectively, indicating that the moss contribute more to the total crust cover as time-since-restoration increases. Lichen cover as a percent of total cryptogamic cover was lowest at currently grazed sites (11%), highest at sites 14-18 years post-restoration (21%) and intermediate at sites 37-38 years post-restoration (18%). Density of lichen species increased from 2.2 to 2.9 to 5.0 species/m^2 at sites currently grazed, 14-18 years post-restoration, and 37-38 years post-restoration, respectively. Moss species density was lower at currently grazed sites (0.9 species/m^2) compared with restored sites (1.6 species/m^2 at sites 14-18 years post-restoration and 1.5 species/m^2 at sites 37-38 years post-restoration). Overall, some of these findings (i.e. cryptogamic crust cover and moss species density) indicate that the cryptogamic crust recovery is large over the initial 14-18 years but does not change remarkably between 14-18 and 37-38 years post-restoration. This trend indicates, as the authors concluded, that the sites have reached equilibrium [at least with respect to some factors] after 15 years. However, the algal cover and lichen species density data indicate that much recovery is still occurring for these factors between 14-18 and 37-38 years post-restoration. Further monitoring is necessary to determine if significant recovery occurs after that time.

A study published by Brotherson, Rushforth, and Johansen (1983) describes the recovery of the cryptogamic crust community 40 years after the cessation of grazing. This study was conducted in Navajo National Monument which is 35 km northwest of the village of Kayenta in northeastern Arizona (see Figure 1). The restored site is in Betatakin Canyon, a site characteristic of a pinyon-juniper cool desert ecosystem. At this site, sheep grazing began in the late 1890's and sheep and cattle grazing has occurred since the early 1960's. The site was restored in 1943 when park managers installed a fence to delineate the park boundary (J. Brotherson, personal communication). To determine the recovery of the crust community, the mean cover of algae, moss, and lichen was compared between the restored area and an area immediately adjacent to the park boundary with continuous grazing. Also, while the last 2 studies did not consider coverage of particular species, this study compared the mean percent cover of the major moss and lichen species between the restored and grazed areas. This information is significant because it allows researchers to differentiate among which species are more quick to recolonize versus those that are slow; the latter will be species that may ultimately require reintroduction by managers.

In their study, Brotherson et al. (1983) found that the algal, moss, and lichen cover was significantly higher in the 40 years post-restoration site vs. the continuously grazed site – 32.7 vs. 13.8%, 6.8 vs. 1.1%, and 14.1 vs. 2.5%, respectively. In particular, all but one species of moss and lichen had a higher mean percent cover in the restored versus grazed site. Two moss species *Tortula ruralis* and *Bryum* sp. had a cover of 0.6 vs. 0.1% and 6.3 vs. 1.0% in the restored vs. grazed site, respectively. Five lichen species had a higher mean percent cover in the restored vs. grazed site - *Collema* sp. (8.1 vs. 1.4%),*Collema tenax* (0.8 vs. 0.4%), *Lichen* sp. 1 (1.3 vs. 0.2%), *Lichen* sp. 2 (0.2 vs. 0.1%), and *Lichen* sp. 4 (1.0 vs. 0.1%). Also, there were numerous species present in the restored site but absent in the grazed site: 2 moss species, *Grimmia ovalis* and *Grimmia* sp. and 12 lichen species, *Acrospora chloroplana*, *Candelariella* sp., *Fulgensia* sp., *Lecanora crenulata*, *Lecanora* sp., *Lecidea decipiens*, *Lecidea* sp., *Lecidea tessellata*, *Lepraria* sp.,*Toninia caeruleonigricans*, *Usnea* sp., and *Xanthoria elegans*. One species, *Lichen* sp. 3, had the same mean percent cover in both sites (0.1%). Further study would elucidate whether recovery occurred earlier than 40 years post-restoration as well as if significant recovery will continue beyond this 40 year period.

A study by Johansen and St. Clair (1986) compares the recovery of the soil crust community 7 and 20 years following the cessation of grazing. This study was conducted at Camp Floyd State Park, which is located 0.4 km southwest of Fairfield in Utah County in central Utah (see Figure 1). This small, 16.2 hectare park, was created in 1962. Prior to park creation, this land was subjected to heavy winter grazing (October to May) by cattle and sheep. Initially this restoration was undertaken by park managers who established the grazing exclosures when Camp Floyd State Park was established in 1962 (L. St. Clair, personal communication). While the previous study makes comparisons between sites 0 and 40 years post-restoration, this study investigated differences between an area 20 years post-restoration with an area immediately adjacent to the park's boundaries that was restored when livestock were removed 7 years prior to the start of the study. Comparison at these more intermediate years (7 and 20 versus 0 and 40) provides valuable information on the rate of recovery when it is possible that the crust community is rapidly changes. If crust communities are relatively similar between these 2 sites, then managers can expect that the majority of recovery occurred within 7 years. If great differences exist between the recovery in the two areas, then managers should expect that natural recovery is slower, and may continue even after 20 years; in that case, management action may be necessary to speed up recovery.

To monitor ecosystem recovery, the average percent frequencies of living algal species; average densities of diatom species; and average percent cover of lichens, mosses, algae, and cryptogamic crust were observed. Average percent frequencies of living algal species in areas 20 years post-restoration versus those 7 years post-restoration were significantly higher for one species, *Nostoc commune* (10.0 vs. 0.5), as well as for visible algal cover (36.5 vs. 16.6) and Shannon-Wiener diversity index (2.75 vs. 2.35). There was a significantly higher density of diatom species in the areas 7 versus 20 years post-restoration for 5 species – *Cyclotella kutzingiana* (39 vs. 11 cells/cm), *Epithemia turgida* (10 vs. 1 cells/cm), *Hantzschia amphioxys* (1250 vs. 562 cells/cm), *Navicula paramutica* (82 vs. 19 cells/cm), *Pinnularia borealis* (404 vs. 105 cells/cm). On the other hand, diversity of diatoms was significantly higher in the older restored sites than the younger sites (Shannon-Wiener diversity index of 2.14 vs. 1.94). Average percent cover was higher in areas 20 versus 7 years post-restoration for lichen (6.4 vs. 5.1%), moss (13.4 vs. 2.3%), algae (22.6 vs. 21.0%), and total cryptogamic crust cover (38.3 vs. 28.1%). Particular species with significantly higher coverages in areas 20 versus 7 years post-restoration were 1 lichen species, *Collema tenax* (6.2 vs. 3.0%) and 2 moss species, *Bryum pallescens* (7.3 vs. 2.0%) and *Tortula ruralis* (6.1 vs. 0.3%). Two lichen species had a significantly higher average percent cover in areas 7 versus 20 years post-restoration - *Caloplaca tominii* (0.4 vs. 0.1%) and *Dermatocarpon lachneum* (1.8 vs. 0.1%). These results illustrate that diatoms are actually more frequent in disturbed areas while significant recovery occurs for lichen, moss, algae, and total crust between 7 and 20 years post-restoration. Further study will be necessary to determine if recovery continues past 20 years.

These first 4 studies provide valuable information on recovery of crust communities after various time periods and the last 2 provide information on the recovery of particular species. However, due to the lack of reference sites in the area (L. St. Clair, K.T. Harper, S. Rushforth, and J. Brotherson, personal communication), comparisons cannot be made with a target system; thus, it's difficult to establish restoration goals with respect to factors like species composition or cover. This lack of reference sites is a result of intense grazing activity throughout the cool desert region over a long period of time. However, this next study is unique in that a reference site was available so comparisons can be made to assess the extent of ecosystem recovery. Furthermore, as I mentioned earlier, this study is distinctive in that it is the first long-term study to consider grazing impacts on ecosystem functioning, more specifically nitrogen cycling.

Ecosystem recovery after grazing cessation

A study by Evans and Belnap (1999), determined the long-term effects of grazing on the nitrogen dynamics in a cool desert grassland ecosystem. Their study was conducted in the Needles District of Canyonlands National Park, just southwest of Moab, Utah (see Figure 1), where nitrogen dynamics and crust coverage were compared between 2 sites. The restored site is located in Chesler Park – an area that was formerly disturbed by grazing animals but was restored in 1962 when the park was created and a fence line was established to mark the park boundaries. This fence line had the effect of also excluding domestic livestock and allowing for crust restoration. The target ecosystem, an undisturbed site, is located in nearby Virginia Park - an area naturally protected from grazing by a narrow rock wall around its border.

Monitoring of the restoration 32 years post-disturbance consisted of an assessment of moss, cyanobacterial, and lichen coverage; measures of nitrogenase activity and soil nitrogen content; a measure of percent microbiotic crust $\delta^{15}N$ (an indicator of changes in soil nitrogen cycling due to disturbance); and measures of net nitrification, net mineralization, and NH_4^+ mineralized (Evans and Belnap 1999). In their study, they found that when comparing the restored with the reference site, lichen coverage (1 vs. 17%), nitrogenase activity (22 vs. 48 nmol $C_2H_2/m^2/h$), NH_4^+ mineralized (2.0 vs. 11.0 ug NH_4^+-N/g/d), and soil nitrogen content (0.20 vs. 0.31mg N/g soil) were significantly lower. The latter 3 results indicate that disturbance to the soil crust has resulted in long-term alteration of nitrogen fixation by the free-living and symbiotic bacteria (Evans and Belnap 1999). The decreased lichen coverage (where nitrogen fixation would occur) in the disturbed site supports this hypothesis (Evans and Belnap 1999). Microbiotic crust $\delta^{15}N$ was significantly lower (1.2 vs. 3.8 and 4.6%) and soil $\delta^{15}N$ was significantly lower (3.8 vs. 4.8 and 5.8) in the undisturbed areas, indicating that significant nitrogen loss occurred in the disturbed areas due to the loss of nitrogen fixing bacteria in lichen (Note: higher $\delta^{15}N$ means higher disturbance). No significant difference was observed in moss coverage, cyanobacteria coverage, net mineralization, or net nitrification between the disturbed and undisturbed site. Overall, these results indicate that although certain components of the cryptogamic crust have recovered (e.g. moss and cyanobacteria coverage), several aspects are still showing the effects of grazing disturbance. Of particular concern are the results concerning lowered nitrogen availability. Because nitrogen is the second most limiting resource (next to water) for primary production in these ecosystems, these alterations could have significant impacts on such factors as vascular plant recovery (Belnap and Eldridge 2001) and thus could ultimately limit the overall success of cool desert ecosystem restorations.

Detailed comparisons among these restorations is not entirely appropriate given that many differences in recovery patterns among the sites are a result of site specificities like local climate or soil chemistry (Belnap and Eldridge 2001). Intensity and duration of disturbance will also affect ecosystem recovery (Belnap and Eldridge 2001) but detailed information on these factors is not available, thus further limiting comparisons among sites (L. St. Clair, K.T. Harper, S. Rushforth, and J. Brotherson, personal communication). In addition, each study used differing response variables and had different times-since-restoration. However, data combined from these 5 long-term studies provides valuable information on rates of recovery for the different crust components, what particular species are slow to recover, and what ecosystem functions are not occurring many years post-restoration. The "take-home message" from these restorations is that even after 20-40 years, significant aspects of the crust community are barely existent or functioning. Efforts to speed up recovery will be necessary.

Limitations and future directions

There are various limitations to the research being conducted in cool desert cryptogamic crust restoration. First, there is a lack of long-term restorations on cool desert cryptogamic crust restoration – they are the 5 studies I mention in this paper. Future studies could focus on more-frequent monitoring to more precisely determine the recovery rates of particular species. Also, particular groups of species have been neglected in current studies (e.g. cyanobacteria) and future studies could provide insight into which species are slow to recover and thus, may require more assistance. Also, future studies could evaluate the recovery of other ecosystem functions (e.g. water absorption) as the crust is restored because managers are sometimes more concerned with restoring ecosystem functions rather than particular species. These points lead to the fact that in general, more long-term studies are crucial in order to be able to predict the outcomes of restorations. Short term studies are not appropriate for developing predictions because they can often lead to miscalculations. For instance, Belnap and Eldridge (2001) concluded that short term studies lead to inaccurate estimates of recovery of crust species compared with longer-term studies. In a study conducted on the Colorado Plateau, scalped plots were evaluated 2-5 and 10-14 yeas post-disturbance (Jayne Belnap, unpublished data, in Belnap and Eldridge 2001). At 2-5 years post-disturbance, cyanobacterial coverage was estimated to recover in 45-110, yet it actually recovered in 14-34 years. Similarly, moss recovery was evaluated after 2-5 years and in turn, was predicted to recover after 400 years; after the second sampling, time-to-recovery was estimated to be 42 years. Accurate predictions of recovery time are essential for restoration managers. If recovery is predicted to be longer than is acceptable due to financial limitations or restoration goals, than action to speed up recovery will be necessary.

There are ecological reasons, as well, for why slow recovery can be problematic. For instance, some slow recolonizing crust species may be preempted by more rapidly colonizing soil crust organisms; ultimately slow recolonizers could be prevented from establishing (Belnap and Eldridge 2001). In addition, slow recovery of disturbed crusts results in a significant period of time when microsite conditions can favor greatly different plant communities from what would have been there historically (Evans and Belnap 1999). The presence of these different plant communities could ultimately prevent the establishment of the crust communities as well as a more "normal" plant community (Evans and Belnap 2001). Also, establishment of the native vegetation will be much delayed if the foundation of the ecosystem - the soil crust - has not established. Most native plants will not be able to establish until the crust and their accompanying ecosystem services (i.e. nutrient uptake, soil stabilization) have been reestablished (Belnap and Eldridge 2001). Finally, complete recovery may be impossible without intervention due a to a lack of source material or altered soil chemical, physical, or microclimate conditions (Belnap and Eldridge 2001).

All crust restoration research in this region has focused, thus far, on restoration through natural processes after the cessation of disturbance. However, as has been illustrated, the crust is not totally recovered even after 40 years. Also, predictions extrapolated from the existing literature of shorter term studies predict that full recovery can take on the order of 50-500 years (Belnap and Eldridge 2001). Future restorations may require managerial intervention through the use of such techniques as inoculation even from the start of the restoration to ensure successful and speedy recovery.

There has been some research on potential techniques that can be used in cool desert crust restoration to speed up the recovery process. A study by Belnap (1993), investigated the use of a soil crust inoculant on soil crust recovery on disturbed versus undisturbed plots in Arches and Canyonlands National Park near Moab, UT. The author monitored a number of sites 2-5 years after the inoculant was added. In general, she found that inoculant use sped up the recovery of green and blue-green algae (indicated by chlorophyll a readings), lichen cover, lichen species richness, and moss cover on disturbed, inoculated sites compared to disturbed, uninoculated sites. However, even with inoculant use moss and lichen recovery was very slow. It is unclear how this technique affects the long-term recovery of cryptogamic crust but the results of this study are promising as a potential technique that can improve the success of crust restorations.

A study by Buttars et al. (1998) investigated the use of pelletized cyanobacterial inoculant for potential restoration of cryptogamic crust. They found that the use of pelletized *Microcoleus vaginatus* in a laboratory experiment significantly increased the biovolume, the filament density, and nitrogen fixation in sterilized, inoculated soils compared with sterilized, uninoculated soils. Since this experiment lasted only 3 months and was conducted under laboratory settings, longer-term field studies will be necessary to truly assess the benefits of pelletized cyanobacterial inoculants for cryptogamic crust restoration. Similar studies to the 2 just mentioned have been conducted in other ecosystems but there is clearly still a paucity of information concerning the use of inoculants to hasten recovery of restored cryptogamic crusts. Further research and, in particular, long term studies of the effects of such practices on ecosystem recovery is essential.

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