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Is Climate Change Mitigation the Best Use of Desert Shrublands?

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ABSTRACT

In a world where the metrics of the carbon economy have become a major issue, it may come as a surprise that intact cold desert shrublands can sequester significant amounts of carbon, both as biomass and in the form of SOC (soil organic carbon). Xerophytic shrubs invest heavily in belowground biomass, placing fixed carbon in an environment where it turns over only very slowly. In order for humans to gain this important ecosystem service, desert shrublands must be kept intact and prevented from frequent burning. The biggest threat to shrubland integrity is the invasion of exotic annual grasses that increase fire frequency to the point that most shrubs can no longer persist. Not only do annual grasslands sequester very little carbon, they also increase the turnover rate of existing SOC. From the point of view of carbon sequestration, restoring the many millions of hectares of annual grass dysclimax in the Interior West to functioning shrubland ecosystems should have high priority. The elimination of perennial understory vegetation and cryptobiotic crusts is a nearly inevitable consequence of livestock grazing in deserts. This opens these systems to annual grass invasion, subsequent burning, and loss of a major carbon sink, a heavy price to pay for the minimal economic gains derived from direct use of these intrinsically unproductive lands for livestock production. On a more immediate scale, the conversion of stable desert shrublands to annual grasslands that burn frequently has also created major issues with windblown dust. Good evidence exists to show that deposition of this dust on mountain snowpack can have the effect of reducing water yield by causing premature melting. Water is clearly the most limiting resource for agriculture in our region, and protecting mountain watersheds from dust deposition should become another important priority. As climate disruption in all its forms becomes a major threat to production agriculture, it is imperative that serious steps be taken to minimize this threat, including restoration of degraded shrubland ecosystems, and prevention of degradation of shrublands that are still intact. Here the argument is made that the best use of cold desert shrublands is mitigation of both short term and long term climate disruption.

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INTRODUCTION

Deserts and semideserts occupy approximately 22 percent of the earth's land surface (Janzen 2004), yet because of their low productivity, they are generally assumed to be relatively minor players in the global carbon cycle. Schemes to mitigate global climate change have rarely included the idea that improving carbon sequestration in deserts could make a significant contribution at a global scale. Many ideas for increasing carbon sequestration, such as tree plantations in marginally suitable environments, involve tradeoffs with other resource values such as water use and quality (Jackson and others 2005). In contrast, improving carbon sequestration in deserts by restoring degraded shrublands to a more functional state would address a broad suite of resource values, including improved air and water quality, wildland fire abatement, enhanced wildlife habitat, biodiversity conservation, and aesthetic and recreational values.

The question addressed here is whether such restoration on a broad scale in the interior West could also make a significant contribution to climate change mitigation. The premise is that restoration of degraded cold desert shrublands could result in sequestration of significant amounts of carbon, and could also reduce the negative climatic effects of excessive windblown dust. The consumptive uses of these ecosystems, which could potentially interfere with management for carbon sequestration, could be said to be relatively unimportant economically, at least in the Interior West. If the carbon credit market that is currently taking shape internationally becomes fully functional, well-managed cold deserts may be able to provide more revenue as carbon sinks than as grazing lands. In addition, management for carbon sequestration can also be viewed as management for maximum return in terms of many other ecosystem services and amenity resources.

Carbon Storage In Deserts

Examination of carbon (C) storage patterns in major biomes on a global scale reveals that deserts (including semideserts) are responsible for the storage of a substantial proportion of the terrestrial C pool (table 1). Stored carbon may be present as standing biomass or as soil organic carbon (SOC), with SOC generally considered to be the more stable and persistent form. It dominates the terrestrial carbon pool at about 80 percent of total stored C (Janzen 2004). The relative contribution of C as standing biomass versus SOC in deserts is even more strongly biased, with over 95 percent of the stored C as SOC. Standing biomass C in deserts is estimated to account for only 1.7 percent of global total, whereas desert SOC is estimated to account for 9.5 percent. Overall, deserts account for about 8 percent of terrestrial C stocks (Janzen 2004). This indicates that deserts are generally about a third as effective as the average biome at storing C on a per area basis. Given the intrinsically unproductive nature of deserts, these figures at first seem surprising. It is hard to see how systems that support such low standing biomass can generate so much SOC. But the same factor that generally limits biomass production in deserts, namely lack of water during much of the year, particularly when temperatures are warm, also limits the rate of microbial respiration in soil, leading to accumulation and persistence of SOC (Jobbagy and Jackson 2000).

The vertical distribution of C in deserts also helps explain how they can be effective carbon sinks (figure 1). When compared with other temperate region biomes, standing biomass, particularly in cold deserts, is dominated by the belowground portion, with root: shoot ratios averaging between four and five (Jackson and others 1996; figure 2). The maximum rooting depth is deeper for cold deserts than for any other biome examined (Canadell and others 1996), and less than 55 percent of root biomass is found in the upper 30 cm of soil (Jackson and others 1996).

This contrasts with perennial grasslands, which have similar standing biomass and relatively high root: shoot ratios, but with >80 percent of the root biomass in the surface 30 cm. This pattern of deep and extensive rooting in cold deserts is probably related to the need to capture winter precipitation stored at depth during the ensuing growing season, which is usually quite dry. The pattern is not seen in warm deserts, where summer monsoonal moisture patterns dominate and root: shoot ratios average less than one (Jackson and others 1996). In deserts, and in shrublands in general, SOC and standing belowground biomass follow similar distribution patterns, that is, with more SOC in deeper soil layers relative to the surface layer than is found in either grassland or forest vegetation (Jobbagy and Jackson 2000). The estimated proportion of total SOC found from 1-3 m in depth is higher for deserts (0.86) than for any other temperate ecosystem.

Table 1. Estimated terrestrial global carbon stocks by biome (Janzen 2004) and estimated mean carbon stock per unit area for each biome.

Biome	Area (10^9 ha)	Global Carbon Stocks (Pg)			Carbon stock/area
		Plants	Soil	Total	
Temperate Forests	1.04	59	100	159	152.9
Boreal Forests	1.37	88	471	559	111.6
Temperate Grasslands/Shrublands	1.25	9	295	304	89.3
Deserts and Semideserts[1]	3.04	8	191	199	58.2
Tundra	0.95	6	121	127	17.9
Croplands	1.60	3	128	131	81.9
Tropical Forests	1.76	212	216	428	243.2
Tropical Savannahs/Grasslands	2.25	66	264	330	108.1
Wetlands	0.35	15	225	240	68.6
Total (not including ice cover)	13.61	466	2011	2477	182
% of total in deserts/semideserts	22.3%	1.7%	9.5%	8.0%	

[1]Area and carbon stock per area estimates in Janzen (2004) for the desert/semidesert biome have been adjusted by removal of areas of ice cover.

In general, SOC has a deeper distribution in soil than roots, and this is especially true in ecosystems with lower precipitation. The most likely explanation for this is that SOC turnover at depth is very slow. Dominance of more slowly degrading forms of carbon, lower nutrient concentrations, and more resistant root tissues at depth contribute to SOC persistence (Jobbagy and Jackson 2000).

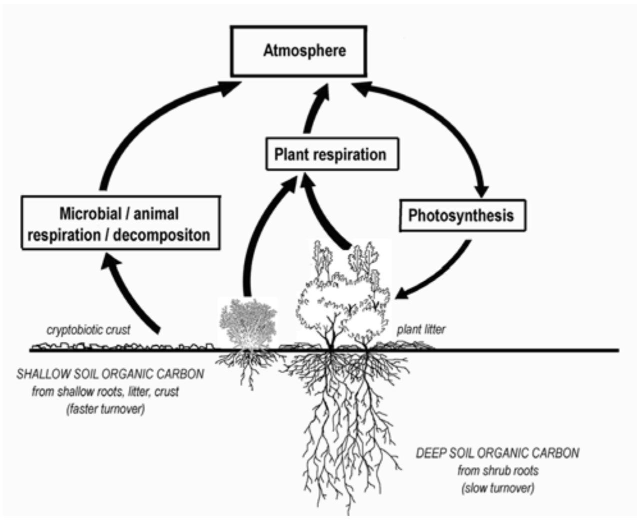

Figure 1. The carbon cycle in a cold desert ecosystem, showing fluxes to the atmosphere (plant respiration and animal/microbial respiration /decomposition), uptake from the atmosphere by plants (primarily shrubs and grasses; photosynthesis), standing plant biomass, and shallow and deep soil organic carbon (SOC). If C uptake exceeds C flux to the atmosphere, C sequestration to a net carbon sink takes place, whereas if flux to the atmosphere exceeds uptake, the system functions as a net carbon source. Deep SOC (soil organic carbon), the most stable form of stored C, dominates C storage in deserts and semideserts.

The ability of cold desert soils to retain SOC could be reduced by the effects of ongoing climate change. Aanderud and others (2010) showed in an 11-year rain manipulation study that near-surface (0-30 cm) SOC stocks in a sagebrush steppe (*Artemisia tridentata*) community were significantly reduced when precipitation was shifted from a winter pattern to a spring-summer pattern. They credited this loss to increased microbial activity in wet surface soil at warm temperatures. Shifts from winter to spring-summer rainfall patterns are predicted for many parts of the Interior West as climate continues to warm (Zhang and others 2007). Rainfall timing impacts on deep SOC would be expected to be lower, however, because deep SOC is more buffered from seasonal temperature changes. This would tend to mitigate the effects of increased warm-season precipitation on soil C storage.

Carbon cycling on US rangelands has been the subject of several recent studies and reviews (e.g., Bird and others 2002, Hunt and others 2004, Schuman and others 2002, Svejcar and others 2008, Follett and Reed 2010, Brown and others 2010). Synthesis of information on carbon storage on rangelands is complicated by the fact that many different vegetation types occurring under many different climatic regimes fall under the rubric of rangelands. Hunt and others (2004), working in Wyoming, found that mixed grass prairie vegetation was carbon-neutral, whereas sagebrush steppe vegetation was acting as a carbon sink. Schuman and others (2002) focused on the potential to increase carbon sequestration in rangelands through improved management, particularly grazing management. Their emphasis was primarily on grassslands. Svejcar and others (2008) report the results of a very interesting 6-year study on net ecosystem C exchange at eight rangeland sites across a range of habitats. They found that both sagebrush steppe sites and three of four perennial grassland sites generally acted as C sinks during the course of the study, whereas the two warm desert sites acted as C sources. Whether a site acted as a source or a sink varied across years and was closely tied to precipitation patterns. Drought years limited productivity and tended to make even the most productive sites temporary C sources.

Because cold deserts store much of their carbon belowground, and because the carbon is stored in deeper soil layers, these deserts are likely to store more carbon per unit area than warm deserts with monsoonal moisture regimes. In addition, the desert shrublands of the interior West might be more appropriately classified as semideserts, as they generally have much higher standing biomass than the true deserts, for example, the Sahara Desert of North Africa, which is virtually plantless over large areas except in drainageways (wadis). This combination of high belowground allocation and relatively high biomass production appears to make cold deserts exceptionally good candidates for management for carbon sequestration.

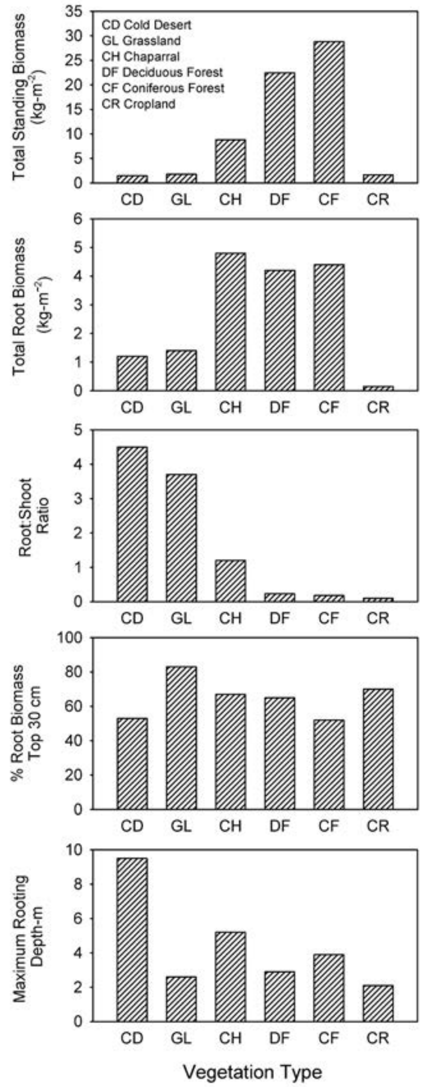

Figure 2. Quantity and distribution of biomass carbon in cold desert biomes contrasted with other temperate zone biomes (grassland, chaparral, deciduous forest, coniferous forest, and cropland): A) total standing biomass, B) total root biomass, C) root:shoot ratio, D) % root biomass in the top 30 cm, and E) maximum rooting depth (adapted from from Jackson and others 1996).

Shrubland Degradation and Carbon Storage

Historically, intact desert ecosystems were most likely in a steady state relationship with regard to carbon budgets, acting in the long term neither as sources nor sinks. But two sets of factors have been operating to disturb this steady state, and these factors generally operate in opposing directions. First, woody 'encroachment' of former desert and and other temperate grasslands is often thought to have shifted the carbon balance in these ecosystems to make them net carbon sinks. Whether conversion from perennial grassland to woody vegetation results in a net increase in C sequestration is the subject of considerable debate, however. Jackson and others (2002) found that whether woody encroachment of perennial grasslands resulted in an increase or decrease in SOC depended on precipitation. There was substantial loss of SOC with woody encroachment in more mesic environments, a loss sufficient to more than counterbalance the increase in standing biomass C resulting from the conversion to dominance by woody species. At the dry end of the spectrum, on the other hand, conversion from perennial desert grassland to shrubland resulted in increases in both standing biomass C and SOC. Most land managers regard woody encroachment as a form of degradation, but its causes are complex and in many cases not completely understood. Climate change may itself be driving woody encroachment in some ecosystems, for example, in the northern Chihuahaun Desert, where creosote bush (*Larrea tridentata*) and tarbush (*Flourensia cernua*) are actively invading desert grasslands (Van Auken 2000). Changes in historic fire regimes, poor grazing management, and other factors may contribute to woody encroachment in other semiarid ecosystems, for example, the invasion of juniper (*Juniperus* spp.) species into sagebrush steppe in the Interior West.

The second process that has had a major impact on carbon storage in the deserts of western North America is the displacement of desert shrubs by invasive annual grasses through increased frequency of fire following destruction of the perennial herbaceous understory through improper grazing management. This phenomenon has not received the attention of carbon brokers that has been given to woody encroachment, but it potentially has more impact on carbon budgets, as it is very likely in the process of converting large portions of the Great Basin and surrounding areas into carbon sources. This possibility was apparently first noted by Bradley and coworkers (Bradley and Mustard 2005, Bradley and others 2006). Using sophisticated remote sensing technologies, these authors conservatively estimated that the area of former salt desert and shrub steppe vegetation in the Great Basin alone that has been converted through repeated burning to cheatgrass monocultures as of 2006 was on the order of 20,000 km^2. In addition, cheatgrass is not the only invasive annual grass that is having major impacts in western

North America. Medusahead wildrye (*Taeniatherum caput-medusae*) and North Africa grass (*Ventenata dubia*) are major invaders in the Interior Northwest, while red brome (*Bromus rubens*) has become a driver of frequent large-scale fires in the Mojave Desert. Many of these fires are occurring in fire-intolerant shrub communities, for example, blackbrush (*Coleogyne ramosissima*) shrublands, that had very low pre-invasion probabilities of burning (Brooks and others 2004).

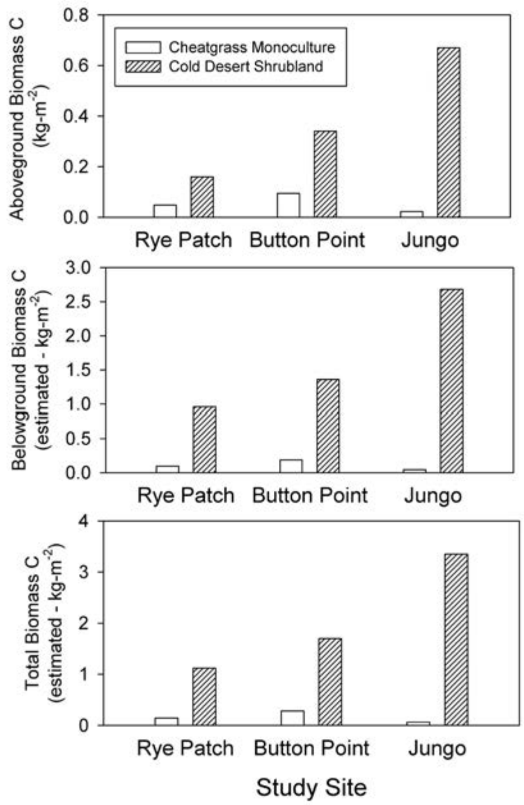

Figure 3. Standing biomass carbon in intact cold desert shrubland communities versus adjacent areas that have been converted to cheatgrass (*Bromus tectorum*) monocultures at Rye Patch NV (salt desert shrubland), Button Point NV (sagebrush steppe), and Jungo NV (sagebrush steppe). Aboveground biomass data from Bradley and others (2006); belowground and total biomass estimated from independent root:shoot ratio data.

Bradley and others (2006) also carried out an on-the-ground assessment of carbon stocks in cold desert shrublands versus cheatgrass monocultures. They measured above-ground carbon stocks and SOC in the near-surface soil horizon in burned and unburned salt desert shrubland (one site) and Wyoming big sagebrush steppe (two sites). They demonstrated a three- to thirty-fold decrease in standing aboveground carbon stocks as a consequence of type conversion to cheatgrass (figures 3 & 4).

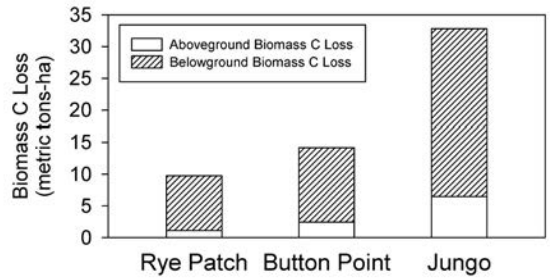

Figure 4. Estimated loss of biomass carbon resulting from conversion from cold desert shrubland to cheatgrass (*Bromus tectorum*) monoculture at three Nevada sites (adapted from Bradley and others 2006; see text for details).

While the study of Bradley and others (2006) did not include any assessment or estimate of root biomass C, root:shoot ratio information for the dominant species obtained from other studies can provide at least a rough estimate of root biomass C in these communities. The estimate of two used here for the root:shoot ratio for cheatgrass is undoubtedly high; in greenhouse and field studies, root:shoot ratios greater than one for this species are rarely encountered, but a conservative estimate was chosen for purposes of avoiding exaggeration of differences (Meyer unpublished data). The estimate of six for the root:shoot ratio of *Atriplex* shrubs is based on estimates by Brewster (1968), while the estimate of four for the root:shoot ratio of *Artemisia* is similar to the estimates for cold desert shrublands in Jackson and others (1996). By revising the carbon stock data of Bradley and others (2006) to include these rough estimates, it can be demonstrated that the loss of belowground biomass carbon has the potential to contribute greatly to the effect of burning on carbon storage in these shrublands (figure 2). Using these estimates, the biomass carbon stocks in the salt desert shrubland were reduced eight-fold through burning and conversion to annual grasslands, while those of sagebrush steppe were reduced from at least six-fold to over fifty-fold.

It is true that belowground carbon from shrub roots is still present for some undetermined length of time post-conversion, after the large pulse of CO_2 emission from the combustion of the above-ground shrub biomass. But ultimately this carbon will be released to the atmosphere, and without actively growing shrubs to replenish this belowground stock, the effect will be conversion of this formerly carbon-efficient system into a long-term source of atmospheric C. Estimates of biomass C loss from the study of Bradley and others (2006) ranged from 1.1 to 6.5 metric tons per hectare for aboveground biomass C, 8.6 to 26.4 metric tons per hectare for belowground biomass C, and 9.8 to 32.8 metric tons per hectare for total biomass C.

Bradley and others (2006) combined their estimates of the areal extent of conversion to cheatgrass monoculture in the Great Basin with their estimates of reduction in above-ground biomass C stocks as a consequence of this conversion to calculate total biomass C released to the atmosphere (table 2). They estimated that about 8 teragrams of C have been released to the atmosphere through shrubland conversion to annual grassland in the Great Basin as of 2006, and the potential for continuing type conversion and carbon release is immense. Adding estimated long-term belowground biomass carbon stock reduction resulted in an estimate of 29 to 60 teragrams of C that will ultimately be released to the atmosphere as a consequence of type conversion from shrubland to annual grassland that has already occurred in the Great Basin.

Invasive annual grass monocultures are not only very poor at carbon sequestration in terms of standing biomass relative to shrublands, but also tend to concentrate their SOC near the surface and to facilitate very rapid turnover of both soil C and N (Norton and others 2004). This is perhaps one reason why it has been difficult to demonstrate direct losses of SOC following annual grass invasion or conversion to annual grass dysclimax (Gill and Burke 1999, Ogle and others 2004, Bradley and others 2006). Most of these studies have examined only the near-surface soil, where SOC under annual grasslands is concentrated. The technology for the study of deep SOC remains cumbersome, so that information on this fraction of the carbon pool is not readily obtained.

Shrubland Degradation and Windblown Dust

Another consequence of anthropogenic disturbance on a landscape scale in arid and semiarid regions is a large increase in the load of windblown dust. To examine the magnitude of this effect, Neff and others (2008) analyzed rates of sediment accumulation in mountain lakes in southwestern Colorado over the last 5000 years. They showed clearly that the rate of sediment accumulation peaked very sharply in the second half of the nineteenth century, a time frame that corresponds with a massive increase in the scope and intensity of livestock grazing in the arid and semiarid regions to the west. These workers further demonstrated using mineralogical analysis that these sediments were not of local origin, but instead represented deposits of windblown dust from the valleys to the west of the watershed.

Livestock grazing and other human activities that disturb the surface soils of deserts generate dust by removal of herbaceous plant cover and, often more importantly, through destruction of the cryptobiotic soil crust that stabilizes the surface in many desert regions (Neff and others 2005). These effects are further exacerbated by annual grass invasion and associated frequent fire. Annual grass cover provides some protection against wind erosion relative to bare ground, but it prevents cryptobiotic crust recovery, resulting in increased dust generation, especially when these areas burn. The Milford Flat fire of 2007 was the largest wildfire in the history of Utah (Miller and others 2011). An enduring legacy of this fire has

Table 2. Estimated biomass carbon loss as a consequence of conversion from cold desert shrubland to cheatgrass (*Bromus tectorum*) monocultures in the Great Basin as of 2006 (adapted from Bradley and others 2006).

	Salt Desert Shrubland	Sagebrush Steppe	Total
Aboveground biomass C loss (tons/ km^2)	110	250-650	360-670
Estimated total biomass C loss (tons/km^2)	1000	1500-3200	2500-4200
Estimated area burned (km^2)	2,000	18,000	20,000
Estimated aboveground biomass C loss (teragrams)	0.2	4.5-11.7	4.7-11.9
Estimated total biomass C loss (teragrams)	2	27-58	29-60

been massive dust storms that have swept windborne dust into the urban areas of northern Utah and onto mountain watersheds. In addition to direct impacts on air quality and human health, this windborne dust exacerbates the effects of climate change through its effect on snow melt rates.

Snow cover has the highest albedo (light reflecting ability) of any natural land surface, and this ability to reflect light also reduces heat loading and melting rate (Flanner and others 2009). When particulate matter, such as dust or carbonaceous pollutants, is deposited along with snow, it lowers the albedo of the remaining snow cover as the snow melts, because the dark particles are concentrated near the surface of the snow. While it is true that particulate matter in the air lowers insolation and heat load on snow at the surface, this 'dimming' effect is more than compensated by the reduction in snow albedo from these particles once they are deposited ('darkening effect'). This effect is especially pronounced in spring, when large areas are snow-covered and incident solar radiation is high. Flanner and others (2009) found that progressively earlier snow melt dates observed in Europe over the last few decades are almost as much due to this snow darkening effect of pollutants from fossil fuel combustion as to longterm increases in spring temperature caused by global warming. Moreover, the positive feedback from earlier snow melt caused by darkening created warmer spring temperatures independently of the effects of global warming, thus compounding the problem.

Though not as potent a darkening agent as carbonaceous pollutants, windborne dust can also significantly increase snow melt rates (Painter and others 2007). Spring dust storms in the desert region to the west of the mountain study area in southwestern Colorado resulted in several dust-on-snow deposition events per year, with more events in a drought year (2006, 8 events) than in an average moisture year (2005, 4 events). These dust-on-snow deposition events resulted in snow cover durations that were decreased by 18 to 35 days. Shortened snow cover duration has measurable ecological impacts at the local scale in alpine and subalpine areas (Steltzer and others 2009). More importantly, it also has the potential to significantly reduce water yields from mountain watersheds. Given that most of the agricultural and culinary water supplies in the Interior West are closely tied to mountain snowfall, and that the thickness and duration of the snow pack and its rate of melting have a strong impact on the ability to harvest this water supply, the fact that desert dust storms can shorten the duration of snow cover in mountainous areas downwind by a month or more should be of grave management concern (Painter and others 2007).

Managing Desert Shrublands for Climate Change Mitigation

Climate change mitigation through desert shrubland management has the goal of maintaining or restoring adapted native shrubland vegetation that produces maximum carbon storage in the long term by exploiting all available niches and thereby maximizing productivity. It is likely that the vegetation that evolved in response to the selective forces in a particular environment will be best able to exploit its resources. This vegetation includes the woody shrub overstory, the herbaceous understory, and also the cryptobiotic crust community that occupies the interspaces. All these components are essential for longterm stability, including surface stability, and sustained carbon storage capacity.

An intact shrubland community is much more likely to be resilient in the face of continued climate change and other disturbances than 'shrub plantations' analagous to the tree plantations currently being proposed and implemented for carbon sequestration. Emphasizing shrubs to the exclusion of other community components in a short-sighted effort to maximize carbon storage would probably result in vegetation that would require intensive management to be sustained. Annual grass weed invasion of the bare interspaces and consequent shrub loss through fire would be a constant threat. A more realistic goal, and one that is bound to be more effective in the long term, is to manage for intact shrubland communities that can rebound even from disturbances such as prolonged drought and fire without high risk of conversion to annual grass dysclimax. Both prevention of further degradation and restoration of degraded shrublands are part of this management scenario.

Cold desert shrublands in the Interior West currently exist in one of three states along a continuum of ecological condition. Some sites still have relatively high-condition shrubland, with native understory and cryptobiotic crust still intact. Many more sites, perhaps most of the area still occupied by shrubs, are in some intermediate condition, with native perennial understory and/or cryptobiotic crust damaged or absent and with annual weed invasion in the understory. These sites are often at high risk of conversion to the third state, which is loss of the shrub overstory through fire and post-burn dominance by annual grass weeds. Shrublands in these different states present different challenges and opportunities for management for carbon sequestration and windblown dust abatement.

Obviously, the most important consideration for high-condition shrublands is prevention of degradation. This means keeping the cryptobiotic crust and the herbaceous understory in the best possible condition. This minimizes the probability of massive annual grass expansion after fire and also maintains surface stability to minimize dust generation. Direct protection from invasion, for example, by controlling nearby weed infestations that could be propagule sources, is another way to maintain ecosystem integrity, as is providing priority protection in the event of wildfire. Even though occasional wildfire was a natural occurrence before settlement, especially in sagebrush steppe, protection from burning under current conditions is a top priority because of the threat of annual grass invasion.

Shrublands in intermediate condition often present more problems than opportunities in terms of improvement for climate change mitigation. Protecting from further disturbance may result in little improvement in these shrublands. Loss of the seed bank of native understory species limits recruitment, and the cryptobiotic crust often cannot recover because of the heavy litter resulting from annual grass invasion. In addition, a common occurrence, especially in sagebrush steppe, is shrub stand thickening or shrub canopy closure in response to loss of understory vegetation. The site at Jungo (Bradley and others, 2006) seems to represent such a scenario. Sagebrush standing biomass was very high, and the understory was completely dominated by cheatgrass. Such a site could be described as 'walking dead' in terms of the risk of conversion to annual grassland, as eventually a shrub-destroying fire is nearly inevitable. Natural shrub recovery after fire is often nil for dominant shrub species like sagebrush and shadscale (*Atriplex confertifolia*), which cannot resprout after fire and rarely establish from seed in areas of high annual grass competition. Active management of shrublands with an understory dominated by cheatgrass will necessitate the development of effective tools to eliminate cheatgrass, reduce shrub cover if necessary, establish understory species, and encourage cryptobiotic crust recovery, all with a minimum of surface disturbance. At present such tools are largely unavailable.

Shrublands that have been converted to annual grass dysclimax communities have usually been given up for lost because of the futility of seeding into dense annual grass stands. But these annual grass dysclimax communities present the most hopeful scenario for increased carbon sequestration. If restoration of these communities is successful, substantial gains in carbon storage can be achieved. There should therefore be a strong emphasis on research aimed at increasing restoration success in areas that no longer support perennial vegetation. Many of the same tools needed for improving degraded shrublands will be needed for restoration of areas that no longer support shrubs, namely innovative methods for annual grass weed control, and new approaches to improving seeding success in environments with low and variable precipitation. At present most seedings in these environments fail, which may seem discouraging. But this points the way toward the development of new approaches that, while they may be more expensive up front, could result in greatly improved seeding success and therefore a much better cost: benefit ratio for shrubland restoration in the long run. It is our challenge as researchers to develop these new approaches. With climate change mitigation as the goal, rather than management of these shrublands for consumptive uses such as livestock grazing, the most creative scientists among us will be inspired to 'think outside the box' and devise the methodology needed to make Interior Western shrublands a significant carbon sink. Even better, along with our partners in management, we will at the same time have the opportunity to enhance the many other ecosystem services and amenity resources provided by these landscapes.

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