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PREFACE

The 17th biennial High Altitude Revegetation Conference was held at The Hilton Ft. Collins, in Ft. Collins, Colorado on March 7-9, 2006. The Conference was organized by the High Altitude Revegetation (HAR) Committee in conjunction with the Colorado State University Department of Soil and Crop Sciences. The Conference was attended by 183 people from a broad spectrum of universities, government agencies and private companies. It is always encouraging to have participants from such a wide range of interests in and application needs for reclamation information and technology.

Organizing a three-day conference and workshop is a difficult task made relatively easy by the sharing of responsibilities among the members of the HAR Committee.

In addition to the invited papers and poster papers presented on March 8 and 9, a Special Erosion Control Intensive Session was held on March 7. This special session was attended by more than 60 people.

We would also like to acknowledge and thank all of the people who took time to prepare invited papers and poster papers. These Proceedings are their product, and the HAR Committee members express our gratitude to them. The Proceedings include 20 papers and 2 abstracts grouped into eight conference sessions, 8 poster papers and 3 poster paper abstracts.

For current information on upcoming High Altitude Committee events, visit our website at www.highaltitudereveg.com.

Warren R. Keammerer
Editor

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COMPONENTS OF SOIL REGENERATION FOR REVEGETATION OF HARSH SUBSTRATES

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ABSTRACT

Natural soils provide many characteristics that are necessary for plant growth including physical support, moisture for plant transpiration, nutrient cycling, and microbial activity. Soils that are drastically disturbed (all topsoil and biological activity removed) often have deficiencies of one or more of these characteristics, so that plant growth is limited. In order to identify which of the soil characteristics must be remediated in order to effectively revegetate the site, guidelines or target thresholds are needed. Agricultural soils aren't good target examples because they are managed for biomass production, not nutrient cycling and conservation. Undisturbed wildlands soils may not provide the best guidelines either, because these may have accumulated high levels of nutrients and organic matter during centuries of development. Providing similar high levels on revegetation projects may not be cost-effective. The best revegetation guidelines are provided by sites that have been disturbed within the last few decades, but now support an acceptable plant community. Evaluating these impacted soils in a quick and effective manner, however, is a challenge because of the highly variable conditions encountered on these sites. While some traditional soil analysis tests can be applied to wildlands soils, other tests are not adequate and must be revised. Using these modified tests, a variety of barren, erosive sites in Northern California have been evaluated and treated. Vigorous perennial plant growth was established without additional irrigation on a range of geological substrates. In spite of the complexity of soil and plant interactions, the most critical treatments were relatively simple.

INTRODUCTION

Because of the wide variety of disturbance impacts and site conditions that occur on large-scale construction sites, the factors that may limit revegetation also vary widely from site to site. The appropriate treatments or amendments will depend on geology, landform, climate, plant type and the extent of disturbance. In an attempt to get a more complete understanding of the different conditions that could limit plant growth, the different components of soils that plants require for growth were listed. Each potentially limiting component was evaluated to determine if treatment or amendment were necessary. Although this study focused on roadway construction, the process is widely applicable to projects with harsh growing conditions and extensive disturbance impacts.

Disturbance impacts to the substrate can range from mild to severe. Impacts restricted to the soil surface, such as those from burns or grazing, may leave the soil intact, but the denuded surface may be chemically altered or compacted and susceptible to erosion. Disturbance from tillage or trenching does not actually remove soil from the site, but since the soil horizons have been mixed to some depth, the material left exposed at the graded surface may have different properties than when the soil horizons were in their natural order. In the case of "drastic disturbance" (Box 1978), all topsoil and biological material has been removed, as occurs with deep excavation or deep burial of the soil under overburden. In these cases, there is no actual soil left, and the process of soil formation (primary succession) must start all over again, beginning with raw mineral substrates. As disturbance impacts to the soil become more severe, plant growth often becomes more constrained, and the likelihood decreases that plants will vigorously recolonize or sustain growth on the site.

For many years, a revegetation approach for harsh sites was to use hardy, "super" plants that would grow rapidly even under tough site conditions. Where plants survived, this practice sometimes resulted in the spread of aggressive species that became weed management problems. Or, if these plants failed to do well, the site gradually became barren again, generating chronic sediment losses. Given the problems of controlling invasive species, as well as increasing regulatory emphasis on non-point source sediment control from poorly vegetated sites, we proposed an alternative approach to revegetation of these harsh sites. If the underlying soil or substrate could be sufficiently regenerated, then it could potentially be recolonized with local native plant species and, if the plant cover were dense enough, then surface erosion could also be reduced. These local native plant species also have other characteristics besides growth potential that would be beneficial in the local environment. These include disease resistance, adaptation for local climate conditions and seasonal growth cueing, compatibility with local pollinators, and resistance or tolerance to local herbivores and various other attributes inherent in plant ecotypes that retain local biodiversity.

Although using native plants to revegetate harsh sites is beneficial for the reasons listed, regenerating disturbed soils to support their long-term growth can be complicated because of the many different functions that soils provide for plants. Functions that are critical for sustained plant growth include such diverse characteristics as providing physical slope stability, infiltration of rainfall and retention of soil moisture, providing nutrient availability and supporting biological activity. To guide soil regeneration work, a sufficiently broad testing scheme was needed in order to identify which of several characteristics on a disturbed site could potentially be limiting to growth of these native plant species.

To evaluate the various soil functions that are required for sustained plant growth on an impacted site, a suite of soil tests were organized into the Soil Resource Evaluation process for use on harsh site revegetation projects for the California Department of Transportation. This process involves a series of steps that roughly parallels the activities of a typical construction project, from conceptual planning through earthmoving, application of bulk soil amendments, spreading of fertilizers or microbial inoculation, followed by surface stabilization and erosion control, and finally, installing or seeding plant materials. The idea of the system is to verify that the different aspects of the soil or substrate on a disturbed site are ready and adequate to grow the intended vegetative community. Each amendment or recommendation is intended to be based on an existing or modified soil test, along with some guidelines for thresholds of adequacy, if possible. Ideally, these tests are rapid, inexpensive and relevant to field conditions. Realistically, though, continued development work is required, especially as sites with more and more atypical or extreme substrate conditions are addressed. The following paragraphs outline the general sequence and the main objectives of each evaluation step. Each section describes the general significance of this aspect of substrate function for plant growth and then gives examples of how to correct deficiencies of that particular soil or substrate function if needed. Most of these discussions are too brief to provide detail or to cover all situations, so if a site appears to have a limiting condition, get qualified help to generate a treatment or amendment that is appropriate and effective for the specific site.

SOIL RESOURCE EVALUATION STEPS

Step 1. Reference Site Selection

The overall goal of the reference site selection step is to identify a realistic model of a potential plant community that has the plant types or erosion resistance that satisfies various stakeholders. Plant transects are often used to describe the vegetation cover on the site. Sediment loss information may also be used to verify that a reference site is appropriately erosion resistant. Detailed soil analyses are not usually included in this first step, so that the focus remains on the general attributes of the site rather than the mechanics of how it works. Both reference and impacted sites are evaluated in detail in Steps 2 through 8, by comparison to this empirically described revegetated reference site.

To provide the best model for regeneration of soil conditions on the impacted site, the reference site should be similar in geology, substrate texture, slope angle and aspect, and topographic position to the impacted site. Over the centuries, undisturbed soils often accumulate much greater nutrient or moisture availability than is necessary for initial revegetation growth (although these reserves can be critical for plant survival during climatic extremes in the long term). The types of reference sites that can most cost-effectively be recreated after disturbance are often those that have had a similar level of disturbance within the past few decades, but have since revegetated to an acceptable level of cover or diversity. These "disturbed-but-revegetated" reference sites typically have soil conditions that are more modest and are more easily regenerated compared to the larger reserves of an undisturbed, native soil. Make sure, though, that the vegetation is not just growing on non-sustainable additions of fertilizer or in an atypically good growing season. Unless a full "restoration" to original conditions is the objective, reconstructing soils to native soil conditions is not usually undertaken. The objective of Step 1, then, is to identify a real-world example of an acceptable site that stakeholders can observe and that verifies that acceptable vegetation can actually be grown under these local environmental conditions.

Step 2. Geotechnical Stability and Rooting Depth

The overall goal of geotechnical design of revegetation slopes is to prevent slope failures such as rotational slumps or slides. Compaction is a common method used to increase substrate strength by increasing particle-to-particle contact, reducing water infiltration, and thereby preventing slope liquefaction and movement (Goldsmith et al. 2001). Compaction also reduces water infiltration into the slope, however, which has the potential to reduce plant growth. Compaction can also negatively impact plant growth by reducing the volume of substrate into which the plant can grow roots, increasing the plant's susceptibility to drought. Therefore, geotechnical slope design has contrasting objectives of "getting water off" the slope for structural stability versus "getting water in" the slope for improved plant growth and reduced surface erosion.

An alternative to compaction of the entire fill volume is to create a zone of less compacted fill at the surface to provide rooting volume for plant growth, while still compacting the fill underneath to meet structural objectives. Care must be taken in this process not to create a "blanket" effect that is geotechnically unstable. Blanket treatments may occur if organics are applied to the surface and then incorporated or raked into a uniform depth. This treatment creates a porous surface layer with a smooth interface over the underlying, compacted material. The porous layer will inevitably saturate with water during storm events or snow melt-off, and, on an inclined slope, this surface layer can liquefy and slide down slope.

On a fill slope, tilling or backfilling with amendments on nearly horizontal benches or steps across the slope serves to "key" the amended, porous soil to the underlying material. When treating an existing fill slope, a backhoe with a long boom may be used to reach over the guardrail and scoop out and mix soil and coarse organic amendments so that the bucket carves out a volume with a base that is relatively flat (horizontal). The substrate and organics can be placed back in the excavated space, or perhaps placed just uphill so that they are blended as they fall back into the excavated step. The excavated areas can be constructed in continuous benches if the substrate is strong enough to hold the mixed volume when saturated, or the steps can be arranged in a diamond pattern across the slope with undisturbed spaces between the steps. Geotechnical engineers should review these slope design elements. Some level of compression or firming of the soil can be tolerated by plant roots; trackwalking on dry materials by crawler tractors, for example, reduces void space but does not stop root growth in the way that excessive compaction does. Gray (2002) addresses some of the issues of bridging between structural and revegetation objectives.

The hydrologic objective of these bench or step treatments is to create benches or steps of more porous, amended soil to quickly infiltrate the rainfall from a storm. Then, the water retained in the amended soil can slowly percolate vertically or seep laterally down the slope over several days time. If the water daylights, or seeps out to the slope surface at the exposed edge of a compacted bench, a woven erosion control blanket may be required to allow water to slowly drain down the slope without cutting a rill. When established, a dense layer of vegetation or plant litter will have the same function. While scattered steps require less amendment and soil disturbance than continuous benches, the less area that is treated to improve infiltration, the more surface flow will be generated from the remaining compacted, untreated slope sections and the less effective the slope will be to infiltrate water from an intense storm.

On a cut slope, bulldozers often start at the top of the slope and cut downward on a wide bench, pushing excess material over the old slope face to be hauled away. In this situation, as the working bench area is progressively being excavated down the hill, a step can be cut into the slope face of the final grade and then backfilled with loose material. This can be repeated at intervals as the working bench is cut downward. When the cut slope is completed, the slope would look like it has an even, sloping surface, but underneath there would steps filled with uncompacted and rootable material.

The required rooting depth to be constructed depends on several components including climatic location and plant type, as well as soil texture and rock content. In general, rooting depths less than 20 cm typically grow only annual grass and weeds in the Mediterranean (summer dry) climate of California. For perennial species, rooting depths of 50 to 150 cm or more are needed. Greater rooting depths are needed for large shrubs or trees. Since these greater rooting depths are not typically constructed on cut slopes, larger shrubs and trees may be restricted to areas with natural fractures in the subsoil material. Actual rooting depth of existing native plants can be estimated from the revegetated reference site or from local soil pits or road cuts. Earthwork specifications on the impacted site can be designed to approximate the rooting depths of the desired plant types.

Penetrometers can be used to probe substrates to see if they are soft enough to be rootable, but different soil moisture contents cause wide variation in soil hardness, making representative measurements difficult. Rocks can also change the required rooting depth. They decrease water availability of the rooting profile if they are impermeable (hard igneous rocks or river cobbles), but if they are porous (such as shales or sandstones), they can actually hold additional water for plant growth. Rocks may actually increase infiltration if the fine soil contains shrink-swell clays that pull away from the rock surface during drying. These cracks or other natural fractures mean that roots may be able to extend well below the tilled or treated soil volumes, increasing the water available to plants.

The purpose of Step 2, then, is to assure that adequate rooting depths are available for plant growth, and that the porous substrate is geotechnically stable. Interact with geotechnical engineers to generate designs with rootable soil volumes as well as geotechnical stability. Work proactively so that the site doesn't end up with hard, smooth graded slopes that are difficult to revegetate.

Step 3. Plant Available Moisture

The overall goal of this step is to assure that sufficient moisture is available for plant growth or survival through dry periods. The components that control moisture content in soils or substrates are infiltration and percolation, as well as retention of water for eventual use by plants. Whereas Step 2 addresses large-scale earthmoving and rooting depth, Step 3 deals with evaluation and treatment of whatever substrate material ends up at the surface of the site.

The rate of infiltration into the soil is largely determined by substrate texture and particle aggregation. Rainfall infiltration rates on native, undisturbed soils are typically very high because individual mineral particles are well aggregated, or clumped together, by a combination of mineral surface oxides, accumulated organic matter, living roots, or fungal hyphae. When they are aggregated, individual particles don't migrate as readily with water flow through the soil pores, where they can settle and plug the pores. The larger particles also have larger pores between the aggregates, which increases the infiltration capacity of the soil.

During the first year after site construction, mechanical disturbance from tillage can decompact the substrate, generating high infiltration rates. As rains settle the particles into a close packed configuration, infiltration typically decreases. If rainfall amounts exceed soil infiltration rates, overland flow occurs. Because surface flow diverts moisture from soaking into the soil, the site may become droughty when rainfall decreases. Infiltration rates on non-porous sites can be immediately increased by incorporating coarse wood shreds, such as from unscreened yard waste composts or shredded mulch from clearing and grubbing operations, as described in Step 4.

The depth of a substrate that is required to infiltrate a storm event can be estimated in a general way by dividing the storm intensity amount (in millimeters or inches of water) by the water holding capacity of the soil. As an example, if a soil holds 9 % of its weight in water when dry and 35 percent when wet, then 35 – 9 = 26 % water holding capacity between dry and wet soil conditions. If a storm drops 75 mm of rain, and each mm of soil depth holds 0.26 mm of water, then 75 / 0.26 = 288 mm (about 11 inches) of soil that will be moistened by the rainstorm. So, following this example, the soil must have rapid infiltration to this depth in order to imbibe a storm of this intensity without developing overland flow.

Many different variations of this calculation can be done. Soil moisture could be estimated at saturated water content (nearer to 50 percent) rather than field capacity, or a greater storm intensity could be used, or the effect could be estimated of several storms occurring in quick succession (meaning there is less capacity to imbibe additional moisture). The capacity to infiltrate moisture will be less if there is a potential restriction of percolation in some soil layers. The depth of wetted soil from a given storm will be deeper if the substrate contains rocks that do not adsorb water. In general, the message is that a typical shallow 100 to 150 mm (4 or 6 inch) rooting zone from surface scarification is not going to infiltrate the rainwater from many storm events, and overland flow will frequently scour the site. This constant removal of the fine organic duff and surface mineral soil will reduce revegetation success and increase sediment loss from the site.

In the field, a glistening sheen of rainwater on the ground surface is an empirical indication of impaired infiltration. Unfortunately, standing surface water is so commonly observed on disturbed sites that it is assumed to be a normal occurrence during rainfall. Any ponded water on the soil surface soon leads to surface erosion as the water accumulates and runs down-slope. When surface flow occurs, the fine, decomposed organic duff and dead plant litter and fine soil particles that provide nutrients for plant growth on a site are washed away. An accumulated layer of residual sands or gravels on the surface indicates loss of fine particles from previous erosion cycles. Pedestals or pillars forming under protective cap rocks and rills are other indicators of surface erosion. Visiting the site during rains and comparing observed surface runoff to rainfall amounts of that storm event is a great source of information about the site. Use an on-site rainfall gauge to get local rainfall intensities, if possible, since storms vary greatly with distance. Actual measurement of infiltration rates is best done with a drop-forming rainfall simulator. This is because the drop size and distribution mimic actual rainfall. Ponded ring infiltrometers are often used but these methods may pipe water down animal burrows and root channels, giving unrealistically high readings.

Percolation of water deeper within the soil partially empties the surface horizons so that they can infiltrate more rainwater during a subsequent storm. Over days or weeks, the water in soil pores gradually percolates downhill to swale areas or springs. This delayed conveyance of rain water moderates stream flow volumes and reduces flooding. Percolation through the subsoil requires large, interconnected pores, such as from plant roots growing deep in the soil. These may take several years to develop in disturbed or compacted soils. Some geological substrates may have natural percolation channels through fissures in fractured sedimentary rocks or in well-jointed and weathered (decomposed) granite. Impermeable substrates should be ripped or excavated if they are not naturally fractured.

Although retention of moisture within the soil buffers water flow into streams, the downside is that the water retained within the slope may cause soils to liquefy and fail. Placement of unconsolidated materials on a horizontal bench reduces this likelihood, as reviewed in Step 2. Vigorous vegetation, especially of woody shrubs and trees, not only creates deep pores for percolation, but it also provides root strength to hold the wet soils in place. For this reason, the amended soil depth needs to be determined not just by the depth needed to imbibe a rainfall event, but it should also be deep enough to support the plants needed to grow roots to hold the soil together and to grow roots deep enough to percolate moisture away from the surface. Percolation tests in augered holes can be used to indicate porosity and drainage of different soil layers. Subsurface limiting layers can cause localized liquefaction and slumping during rains. When specifications for adequate threshold infiltration rates are not available, then just use comparisons of infiltration rates between the reference and impacted sites.

If infiltration and percolation are adequate to get rainfall into the soil, the next condition needed to improve revegetation is to retain moisture for plant growth during later droughty periods. The presence of adequate water holding capacity for plant growth is evaluated by comparing moist and dry water contents in the soil. “Plant available water” is defined as being the difference in the amount of water held in a soil from when it drains after a rainfall event (called field capacity) compared to when the soil is so dry that plants can no longer extract moisture (wilting point). In general, the potential for water retention is predominantly determined by clay content because the water is held in the pores between the fine clay particles. Clayey soils retain a large proportion of water, but some of the water may be held so tightly that some plants cannot remove it. Drought-tolerant wildlands plants can draw water down to much drier levels than many horticultural and some agricultural species, so the definition of “available” moisture may change with different plants. Sandy soils hold water in larger pores where it is more easily available for extraction. As sandy soils dry, however, the amount remaining is very low, so these soils become droughty. The greatest amount of plant available moisture is usually in loamy textured substrates.

If the plant available water in the soil is less than about 10 percent, amendment with composts will improve moisture conditions for plant growth. If the soil has greater than this amount of plant available moisture, then both soil and compost will hold about the same amount and there is little gain in moisture availability from adding the amendment. Some inorganic clay products can also be used to increase water holding capacity. Their plant available water content may range from 10 to 18 %, so the soil volume that they replace must hold less moisture than these amounts if a net increase in moisture is to be gained from amendment. The clay amendments represent an additional cost, but they become a permanent component of the soil, as opposed to organic amendments, which decompose with time. But, the clays do not provide the biological stimulation obtained from compost amendments. Commercial laboratories offer soil tests for measuring field capacity and wilting point water contents. Either straight substrate materials or blends of substrates and amendment materials can be sent in for testing of water contents.

Evaluation of the substrate to determine if it has sufficient moisture for plant growth can be made in a general way by using a target value for plant water use and dividing it by the available moisture content in the soil. Note that this is not the total water used by the plant through the whole year, but is only for some defined droughty period, usually lasting for several weeks or months until the next significant rain. At a harsh site in California's Coast Range mountains, water use by a perennial grass (*Elymus elymoides*, squirreltail) through the extended summer drought was estimated to be 84 mm (Curtis and Claassen, 2005). Using a 10 percent plant available moisture content of the soil, for example, then 84 / 0.10 = 840 mm (33 inches) of soil that would be needed to hold moisture for plant growth during this stress period. The proportion of the soil volume occupied by rock fragments having no available water must also be accounted for, which makes the required soil volume deeper as rock content increases. Annual species were estimated to use less than 50 mm water during the summer season, while other literature values for shrubs and trees suggested up to 600 mm water used during the summer drought.

Step 3 then, involves evaluating the site for the ability of the exposed substrate to infiltrate rainfall at a rate that exceeds defined storm events, in order to avoid overland flow. Once the soil is wetted, the difference between saturated or field capacity content and wilting point determines the amount of water available for plants to use during drier seasons. If analysis of soil moisture is not possible, empirical evidence of adequate water availability may be used. Observe the revegetated reference site conditions for lack of evidence of surface erosion, for rooting depths, similar soil textures, rock content and soil aggregation. If coarse organic amendments are to be used, they must be integrated with objectives of Step 4. If water retention is to be improved by composts or clay amendments, the plant available moisture of the amended material must exceed that of the ambient soil if there is to be any improvement in plant growth. It may be more cost-effective to improve plant available moisture by simply increasing the rooting depth in the ambient material by shattering or ripping the subsurface material or by backfilling extra depth of rootable materials rather than to haul in and incorporate soil amendments.

Step 4. Soil Organic Matter

The overall goals of soil organic matter (SOM) amendment, if needed, are 1) to improve infiltration, 2) to support microbial activity through carbon decomposition, and 3) to provide long-term pools of plant available nitrogen (N) for plant growth. The need for organic matter amendments may be established in other site evaluation steps, but specification of a SOM amendment is addressed here in Step 4 because of the need to integrate and balance all the different effects of organic amendment, including amendment with coarse woody materials for infiltration, C additions to the substrate for microbial decomposition, and N release for plant growth.

The first function that SOM amendments can provide is to improve infiltration rates, if tests in Step 3 indicate that they are lower on the impacted site than on the revegetated reference site. Amendment with coarse woody material can regenerate infiltration immediately when it is incorporated at the time of construction. This coarse woody material can be generated from clearing and grubbing activities on a site, from forest thinning activity, or from coarse, unscreened yard waste compost. Shredded wood is expected to be more beneficial than wood chips because the longer fiber length of the shreds (75 to 125 mm; 3 to 5 inches) creates longer continuous pores for drainage. In a study of infiltration on eroding decomposed granite slopes, amendment volumes of approximately one part unscreened yard waste compost to three parts decomposed granite, incorporated to 50 cm, were needed to bring infiltration to the level of a "disturbed-but-revegetated" slope. Although amendment with composts as surface-applied mulches is beneficial because they add insulation and raindrop protection, they do little to increase infiltration into the soil unless they are tilled in.

The second function that SOM amendment provides is energy for microbial growth. When plant materials are added, they rapidly decompose so that only a small fraction of the original amendment remains in the soil after a few seasons. In the process of decomposition, however, perhaps half of the carbon is converted to microbial biomass, some components of which have a much greater resistance to decomposition, and therefore a greater stability in the soil. These microbial cell walls and exudates, along with some resistant components of plant tissue like lignin, are reformed to produce soil humus, the stabilized, dark, soil organic matter that provides many soil benefits. Soil humus has a high ionic charge, so it holds onto nutrients, it aggregates fine particles into larger particles that keep soils from setting up hard when dry, it holds moisture, and it stimulates microbial activity. Compost additions to our harsh sites consistently show increases in plant vigor, beyond that accounted for by increased N or moisture availability. Soil organic matter interactions on disturbed sites are reviewed in detail in Smith et al. (1987).

The third function that SOM provides for revegetation is a steady, long-term supply of N for plant growth. This results from the decomposition of N in stabilized organic matter and conversion of organic N into plant-available forms. The speed of this N-release process is determined by moisture and temperature and the chemical makeup of the organic substrate. If the carbon in the organic matter is easily decomposed and there is limited N available from the substrate, microbes will rapidly take up N from the substrate or environment and incorporate it in their biomass. This process is called N "immobilization" and can reduce the amount of N in the soil that is available for plant growth. In contrast, as carbon-rich decomposable materials become depleted and microbial populations decline, the net effect is that dying microorganisms release their biomass N in a process called N "mineralization." This provides additional plant available N, mostly as ammonium and nitrate. A large proportion of the stabilized SOM is in the form of bacterial or fungal cell wall residues, which decompose slowly and create an inherent, slow-release N source for plant uptake and growth.

Whether an organic matter substrate produces net N immobilization or net mineralization depends on the relative proportions of decomposable N and decomposable C components in the organic materials. In general, composts and woody materials that are very fibrous and uncured (not aged) tend to immobilize N and reduce plant available N pools. Well-cured composts (several months of aerobic curing after the thermophilic stage is finished) and materials with biosolids components tend to mineralize N, increasing the amount available for plant growth or for leaching losses.

These examples show the dilemma inherent in specifying organic matter amendments. Infiltration is best improved with very coarse, fibrous, decay resistant material, but microbial biomass growth requires finer particles with greater surface area and decomposable chemistry. Furthermore, the mineralization of N by organic matter decomposition is greatest in a well-aged compost, in which most of the coarse material has already been broken down and converted to the crumbly, amorphous carbon approaching soil humus. Blends of various materials may work, although the proportions need to be established empirically, since no specification or measurement system exists to balance these contrasting properties of an organic matter amendment. The availability of bulk organic materials is often governed by their proximity to the site (to avoid hauling costs), so an ideal material may not be economically feasible. Blends of different kinds of materials may be a reasonable approach to get around the problem. The N release of some uncured (immobilizing) organic materials (coarse composts or forest thinning products) could be increased, for example, by blending varying amounts of a soil amendment with a known N release rate. Soil amendment products made up of fungal and bacterial residues may function similarly to the slowly mineralizing microbial biomass that is found in natural soils, providing an N supply that matches the C decomposition patterns.

The amount and type of N that is appropriate for amendment to a field site depends on the ambient availability of N for plant growth on the site and the N requirements for revegetation. Between 200 and 400 kg N/ha is commonly taken up into various biologically active components of the complete revegetation community, including live plant roots and shoots, standing dead material, decaying litter, microbes and larger animals (Reeder and Sabey 1987). Whether acquired from residual soil N pools, fertilizer amendments or other sources, the regeneration of a revegetation community requires significant amounts of N to rebuild its component parts. Too little N available late in the recovery process may result in a thin stand or lack of an erosion resistant duff layer. Too much available N early in the recovery process when plant cover is not complete may facilitate weedy invasion and delay succession (McLendon and Redente 1992). These complex aspects of N cycling on disturbed lands are nicely reviewed by Reeder and Sabey (1987).

Areas near metropolitan areas or near roadways may get substantial additional N from atmospheric deposition. Some apparently pristine areas are impacted in this way. The Rocky Mountain National Park receives over 5 kg N/ha/yr from atmospheric N deposition compared to 0.2 historically (Morris 2002), The southern Sierra Nevada receives 2.8, Yellowstone NP receives 1.4, and the Mohave desert in California receives a little over 3 kg N/ha/yr. Regions with greater vehicular traffic can receive greater amounts, including and estimated 6.8 kg N/ha/yr for the urbanized, southern Lake Tahoe area and 25 - 45 kg N/ha/yr for the LA basin (Tarnay et al. 2005). Even non-urbanized areas can receive over 20 kg N/ha/yr deposited within the first few meters adjacent to the roadway (Rusmore 2004). Compared to the low nutrient contents in high elevation plant communities, even small amounts of additional N can generate significant biological effects, especially from weedy species.

Depending on the level of disturbance, some residual N could also remain in the substrates of the site. Narrow zones of disturbance such as paths, roadways or pipelines have undisturbed soil within reach of roots. This can reduce or eliminate the need for additional N amendment. Sedimentary or overburden substrates often contain measurable residual N that weathers from the rocks (Reeder and Sabey 1987).

In the igneous substrates of Lake Tahoe, however, there is little or no geological N. Here the N needed for plant growth is supplied by relatively large, stabilized soil organic matter pools with slow N mineralization rates. Sites that have been disturbed within the last 3 to 25 years but have since revegetated to greater than 40 % cover (typical for the region) were measured to have an average total N of over 1200 kg N/ha (Claassen and Hogan 2002). Of this large amount, however, the mineralizable N was estimated at 26 kg N/ha, or only about 2 % of the total N. In this way, large, stabilized soil organic matter pools resemble a low risk - low yield type of financial investment, with a large initial capital cost and a stable but small return on the N investment. This pattern of soils with large, stabilized soil organic matter contents and slow N mineralization rates is typical for disturbed sites with sustainable revegetation communities. The soil organic matter can accumulate over several centuries, or it can be loaded on at construction using composts or other organic amendments, if the organic matter is well stabilized.

So, in selecting organic amendments for a site, several different functions need to be integrated. These include balancing the effects of coarse, fibrous wood shreds for infiltration, the microbial activity that generates aggregates, and the decomposition dynamics that immobilize or mineralize N. Organic matter amendments can have multiple and sometimes contradictory effects on site conditions and plant growth. Practical guidelines are still being developed.

Step 5. Non-N Nutrient Availability

The overall goal of this step is to assure that one or more non-N nutrients or soil chemical characteristics do not limit plant growth on the impacted site. These non-N nutrients encompass all the macro and micronutrients except N, including P, K, S, Ca, Mg, micronutrients, EC (salt), CEC (cation exchange capacity), pH (acidity) and sometimes sodium levels. While the soil organic matter components in Step 4 were closely linked together in biological cycles, the non-N nutrients and soil chemical conditions are much more controlled by inorganic processes, particularly on drastically disturbed sites. Because they are less biologically controlled, conventional soil tests developed for chemical agricultural systems are useful for testing most of these conditions. Interpretations of the tests, however, need to be adjusted for wildlands plant growth rather than for production agriculture. In general, this means accepting lower levels of available nutrients and higher levels of salts.

Plant growth requires each of the non-N nutrients and soil chemical factors to be at adequate levels. When extra water or N is available, plants will often grow larger. When excess levels of non-N nutrients or chemical conditions occur, however, plant growth will generally not increase as much because plant growth is often still limited by water or N. Deficient levels in non-N conditions, in contrast, can definitely reduce plant size or increase spacing between plants. So, while water and N are viewed as controlling the amount of plant biomass generated on the site, the amendments of non-N nutrients are made with the objective of making them "sufficient", or not limiting. They are generally not viewed as drivers for plant growth on most of our sites.

Because native plants can adjust biomass production, plant spacing or root:shoot ratios, the thresholds for deficiency are not specific values. Plants can adjust to a wide range of nutrient concentrations, as long as there is some minimum level of availability. So, while commercial tests are used, acceptable levels are usually assigned to what would be equivalent to "low" or "moderate" levels in agricultural soils. If definitive threshold numbers are not known for particular soil, substrate or native plant combinations, then the nutrient levels that are assumed to be adequate can be established by comparison to levels from the "disturbed-but-revegetated" reference site. As soil chemistry becomes more atypical and sites more disturbed, it becomes more likely that a trained soil scientist should review the sampling plan, analysis methods and test data. This is because local soil texture and soils depth, or atypical soil chemistry can interact with the amended nutrients, either increasing or decreasing their availability.

A useful supplement or alternative to soil testing is to create field demonstration plots. In this case, complete the plan for the site using the best available or recommended treatment, but withhold the amendment from a small area of the site and add it to another similar sized area. With relatively little extra effort, three comparison treatments can be created, including an unamended control (the area left untreated), a 1X (the standard) treatment, and a 2X treatment (amendment rate doubled using the material not applied to the unamended control). Plant response to the treatment gradient can be observed or measured at various levels of intensity, depending on project resources and objectives. This method avoids the complexity of soil sampling, analysis and generation of suitable amendments, but it requires waiting for plant growth responses from the field plots. A third alternative is to grow trial plots in buckets in a greenhouse or lath house. Plant growth in pots, however, may not be representative of field conditions if the pots constrict root growth.

Because there are many different nutrients included in this group, each with different chemical interactions with the substrate, a soil specialist should be involved in reviewing these amendments. The revegetated reference site provides examples of empirically "adequate" levels in the face of limited information regarding the nutrient response of many poorly described native plant species.

Step 6. Soil Microbiology and Biology

The overall goal of this step is to evaluate disturbed sites for microbial activity. Microbes are absolutely critical for nutrient cycling and soil aggregation, but in most cases, if plant growth is regenerated, microbial colonization and activity will follow. Facilitation of microbial growth is more commonly needed than inoculation. The tests vary according to whether decomposing microbes or mycorrhizal fungi or N-fixing microbes are the focus.

A primary microbial function on disturbed sites is nutrient cycling. In most cases, impacted sites have reduced populations of microbial decomposers because organic matter inputs from plant growth are greatly reduced. Providing decomposable organic matter to the site is usually all that is needed to regenerate microbial decomposer populations. In most drastically disturbed cases, dispersion of microbial inocula is rapid enough that organic amendments or plant material grown on site will be rapidly colonized and microbial cycling will be regenerated. As seen in Step 4, the quality of the organic matter has strong effects on the course of organic matter decomposition and release of nutrients for plant growth.

An additional critical, but long-term, effect of microbial activity is generation of stabilized (humified) soil organic matter. As reviewed in Step 4, these carbon compounds occur when plant material is used to generate microbial biomass, some of which is decay-resistant cell wall structures or exudates. Through various processes, organics accumulate that are very stabilized, or slow to decompose. These humified organics serve to glue particles together in aggregates so that they do not settle, plug pores and reduce infiltration. Organic matter production and decomposition processes must continue for many years, because a relatively small fraction of the initial organic matter is actually converted to stabilized, humic materials. In some mined land sites, microbes cannot grow because of high metal concentrations in the substrate. This situation is often made visible by the accumulation of undecomposed organics from plant growth or woody debris that is covered with metal-rich crusts. Removal of the source of the metals or burial are common treatments for this situation.

A separate type of microbial activity is the formation of mycorrhizae, or symbiotic "fungus-root" relationships between plant roots and certain soil fungi. In concept, the mycorrhizal fungi improves plant nutrient uptake and the plant root feeds the fungus. Although sometimes mycorrhizal colonization is critical for plant growth, there are many cases in which the plant does not clearly benefit. Although many plants form mycorrhizae, whether this association is beneficial probably depends on specific pairings of plants and fungal species and on soil chemistry.

Most common mycorrhizal fungi are in three major groups. Endomycorrhizae or arbuscular mycorrhizae (AM) generally grow on herbaceous plants, most shrubs and a few tree species. Most herbacious and woody plants colonize with AM fungi, with some exceptions such as mustards and chenopods like Russian thistle. Oaks and conifers (except cedar), however, are associated with a different group of fungi called ectomycorrhizae (EC), although some overlaps and multiple colonizations occur. A third general type (ectendomycorrhizae) colonizes ericaceous species (*Manzanita, Vaccinium*). Orchids utilize still a different group. The inoculum must match the general plant type.

For the purposes of a brief soil evaluation, EC fungi can be observed by hand lens or naked eye as white, brown or yellow fungal growths on shallow conifer or oak roots, or by the short, stubby (coralloid) shape of the roots. Many "toadstool" mushrooms are the fruiting bodies of these mycorrhizal fungi. Because the spores from these fruiting bodies are widely dispersed through the air, inoculation occurs readily over large distances. Inoculation is done on nursery stock to speed the formation of the mycorrhizal relationship, or if special types of fungi are desired.

The AM fungi, however, cannot be seen by hand lens or eye, and must be evaluated with root samples that are cleared and stained in the lab and then counted under a microscope. The spores of these AM fungi are large enough (30 to 100 um) that they do not disperse as readily as EC spores. Therefore, inoculation may be beneficial for revegetation of sites that have never had plant growth, have been fumigated, or have had long fallow periods with no plant growth. The source of the inoculum may be critical for plant response, however. Site-collected native fungi from the Rocky Mountains showed greater benefits for native plant species than colonization by non-native fungi. Growth of weedy annuals was not facilitated (Rowe et al. 2006). Inoculation of shrubs in Spain with non-native AM fungi improved growth in the first year or two, but after 5 years the non-native inoculum decreased shrub growth compared to shrubs inoculated with native AM fungi (Requena et al. 2001). Because the relationship of plants to mycorrhizal fungi is complex, a symbiosis that is beneficial may also be expressed in ecological terms (increased survival, seed set, disease resistance) even though it may result in smaller plant size. Mycorrhizal inoculation may provide significant benefits to revegetation of drastic sites, but the results should be expected to vary by plant, site and inoculum source. Use of inoculum from the revegetated reference site from similar plant species is an effective approach to getting appropriate inoculum, as long as the soil chemistry is also similar.

For soils with reduced infiltration, growth of mycorrhizal fungi may be a primary process for improving soil structure. This may occur as fungal hyphal strands entwine soil particles to keep them from settling into pores as well as from exudate production. With AM fungi, this exudate is produced in large amounts and may be a major factor in soil regeneration and organic matter stabilization. So, mycorrhizal fungi may have important ecological effects other than through nutrient uptake.

Biological N fixation is often cited as a way to provide additional N on a revegetation site, but the actual amount of N contributed to the community is probably low (Reeder and Sabey 1987). Sweet clover and alfalfa have been reported to fix several hundred kg N/ha/yr, but agricultural species may not survive on arid wild land sites. Native legumes may be an effective alternative, but N fixation rates have not been well documented. As a result, the N required to regenerate a revegetation system is predominantly provided by fertilizer and organic amendments, with biological N fixation as a supplemental source.

Soil biological evaluation surely should also include burrowing insects, earthworms and rodents. The effect of burrows on infiltration is relatively unstudied, as are the effects of the burrowers on the plant growth. Although herbivory can be devastating for vegetation plantings, the effectiveness of habitat improvement (shelters, roosts) for predators to control rodents has received little attention. In short, while we can correct soil conditions to grow plants, we should recognize that there are many other influences within the revegetation community, some of which would probably be useful to control or enhance.

Step 7. Temporary Surface Stabilization and Surface Mulches

The overall goal of this step is to control water flow onto, through and off of the site without mobilizing sediment. A wide range of erosion control methods and products are commercially available. Many of these methods or products are needed because the infiltration of the site too low initially, as reviewed in Step 3. Tillage or ripping of adequate depth, with a surface that is left in a rough condition may eliminate much of the surface flow that generates sediment. Surface application of straw, compost or chipped wood mulches or hydro-mulch materials is important to reduce raindrop splash detachment and to slow water flow so that it has more time to infiltrate. On many sites, however, continued construction activity means that surfaces will be compacted and water will be shed during rains. In this case, sediment control may require containment methods in addition to prevention. This is often more expensive and less effective. From a soils perspective, infiltration (Step 3) is a major determinant of erosion potential because it determines the presence of surface water. Surface water, in turn, enhances splash detachment in the liquefied surface particles, followed by surface flow and scouring as standing water runs off the site. Minimizing compacted areas is the most critical preventative step.

An additional function of surface mulches that can be observed on many vegetated reference sites is that the accumulated plant litter and duff insulates the soil from freezing during the winter and insulates the soil from drying during the summer. Barren areas of fill along pipeline installations in the Tahoe area were observed to freeze to more than a foot deep, while the adjacent, undisturbed soil with even moderate organic matter accumulation had only surface frost. The unfrozen soil also allows continued root activity and infiltration. Mulch depths required for insulating temperatures and preventing moisture loss will vary with different materials but these are easy to check in the field by adding test areas with greater amounts. A simple check for frozen soil during the winter and dryness during the summer can be used to determine an adequate mulch application under local climatic conditions. The dangers of excessive mulch are that the soil may be cooler during the spring thaw, which may delay seedling growth, or that sporadic rains may wet up the mulch but fail to penetrate the mineral soil underneath. Guidelines for slope drainage patterns, duff or mulch layers, or infiltration rates can all be approximated from the revegetated reference site, if it has weathered several seasons with a range of storm intensities.

Step 8. Special Plant Materials

The overall goal of this step is to evaluate whether specific plant materials are appropriate for the environmental conditions at the site. The purpose is not to develop a whole plant list, but to evaluate whether the special conditions at the site require special types of plant materials. As the substrate chemistry or conditions get more extreme, the likelihood increases that specifically adapted plant materials may be needed. For example, plants adapted to serpentine substrates (high magnesium, low calcium) have different calcium regulation abilities and rooting behaviors compared to non-serpentine adapted plants (O'Dell and Claassen 2006 a,b,c). Some plant types may be more or less adapted to acid mined materials. High elevation sites may require different environmental cueing for fall dormancy or germination than accessions from lower elevations. For example, the commercially available sterile hybrid crosses available for revegetation, have different cold tolerances. Get a general description of the site conditions together and talk to a good field botanist or seed producer. Different plants have different tolerance ranges, and different agencies or stake holders have varying standards for use of locally collected plant materials. If using materials collected from the revegetated reference site, make sure that it wasn't planted with non-local materials some time in the past, if tracking plant materials sources is a priority issue. Many species are tough and locally adapted, but are not recognized generally as "revegetation" species or produced commercially. Scour the edges of the site or come back during different seasons to find a range of materials. Correcting soil conditions at the site opens up the possibility that many other native species are available that could revegetate the site well.

CONCLUSION

These eight steps are not intended to cover all the complexities of soil remediation on a revegetation project. They are, however, a fairly complete itemization of the categories of conditions that should be addressed to assure plant growth on the site and continued vitality of the vegetative community for decades after establishment. As more site analyses are completed, more concrete examples of adequate conditions or amendments will be available, so that sites can be addressed by practical example rather than by numerical detail. In the mean time, when in doubt, follow the examples from the plants and soils of the "disturbed-but-revegetated" reference site. The first step in this process is simply to recognize the various components of soil and plant relationships and to observe how disturbed substrates may or may not support plant growth.

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PRELIMINARY EFFECTS OF PATCH-BURN GRAZING ON A HIGH-DIVERSITY PRAIRIE RESTORATION

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ABSTRACT

Once most of the plant species become established in a new restoration planting, the project focus shifts from restoration to management. Management is critical because it will largely determine the composition and function of the new plant community. The Nature Conservancy has been conducting high-diversity grassland restoration along the Central Platte River in Nebraska since the early 1990's. Over 1,000 acres have been planted with seed mixes containing between 180 and 230 species. As those plantings become established, the Conservancy is testing and using a variety of management tools and strategies to increase and maintain plant diversity. One strategy is Patch Burn Grazing, which combines prescribed fire and cattle grazing. A portion of the site is burned each year, but cattle are given access to the entire site. Cattle concentrate their grazing in the recently burned portion of the site and ignore most of the rest of the site. This results in intense season-long grazing followed by a period of recovery and rest until the same portion of the site is burned again. Moreover, cattle graze primarily grasses within this system, ignoring forb species that are normally preferred. Early evaluations indicate the system will favor plant diversity and create heterogeneous habitat structure.

Helzer, C.H. and A.A. Steuter. Preliminary Effects of Patch-Burn Grazing on a High-Diversity Prairie Restoration. ECOLOGICAL RESTORATION 23(3):167-171. Copyright 2005. Reprinted by permission of The University of Wisconsin Press.

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INTRODUCTION

The Nature Conservancy has been restoring marginal cropland to high-diversity grassland/wetland restorations along the Central Platte River in Nebraska since 1994 (Steinauer and others 2003). Each planting consists of locally harvested seed from 150 and 225 plant species. Many of our older restorations have now reached the point where they require stewardship to maintain and improve the diversity of the established plant communities. Because the combination of fire and cattle grazing is an important part of our stewardship program for remnant prairies, we have begun investigating its potential role in managing our restored plant communities as well.

The Nature Conservancy initiated a patch-burn grazing system on some of its large, bison-grazed grasslands in the Great Plains in the late 1980s (Steuter and others 1990). While the method varies, the basic idea is to annually burn part of a grassland (on a scheduled derived from an estimated aboriginal fire-return interval) and then give grazers, such as bison, access to both the burned and unburned portions of the pasture. In general, bison spend the majority of their time grazing in the most recently burned portion, less time in the portions burned in prior years, and very little time in the remaining portion during the grazing season. Thus, burning results in intense grazing pressure during the first year after the fire, which opens up space between the dominant grasses for new growth of forbs, particularly short-lived annuals and biennials. Those "weedy" forbs become dominant during the next year or two and then slowly subside under competition from the recovering perennial grasses. The periodic intense disturbance is also likely to help other longer-lived plants establish new individuals through seedlings. While the method was first used on grasslands larger than 5,000 acres (2,000 ha) using bison, a number of people are now testing its potential to manage much smaller sites using cattle as the grazers (Fuhlendorf and Engle 2001). Initial results from these studies have been very promising, with cattle in small pastures following much the same patterns as bison in larger enclosures.

Grazing by cattle is viewed by many prairie enthusiasts as a negative force in high-quality grasslands, and particularly harmful to conservative forb species. This view is perpetuated by the poor quality of the generally small pastures that are scattered within the eastern range of the tallgrass prairie. However, poor-quality pastures are the result of long-term, continuous overgrazing, not cattle grazing in general. In fact, the use of large ruminants as a management tool may be the most flexible method available to prairie managers because of their selective nature.

In this study, we investigated the effects of the patch-burn grazing system in restored prairie along the Central Platte River, near Grand Island, Nebraska. Specifically, we wanted to look at the grazing selection by cattle of various forb species and compare the impacts to forbs in the burned and unburned portions of a prairie. In addition, we wanted to begin to evaluate the usefulness of the patch-burn grazing system for managing small prairies with cattle and fire.

METHODS

The study took place in the summer and fall of 2002 in a wet-mesic site along the Central Platte River with follow-up data collection in 2003. The site is located within the Central Mixed-Grass ecoregion but, because of the site's proximity to groundwater, the native vegetation is dominated by tallgrass prairie plants, such as big bluestem (*Andropogon gerardii*), Indiangrass (*Sorghastrum nutans*), Illinois bundleflower (*Desmanthus illinoisensis*), and Maximillian sunflower (*Helianthus maximillianii*).

The 2002 study site consisted of 185 acres (75 hectares), about a third of which consisted of restored prairies planted in 1995 and 1997. However, we only tracked plant-burning-grazing effects in the 1995 planting. Within the 185-acre site, we conducted two 25-acre (10-ha) prescribed burns in the spring of 2002 (Figure 1). Both burns covered restored and remnant portions of the site. The site was grazed with 15 cow/calf pairs (0.6 Animal Unit Months/acre)

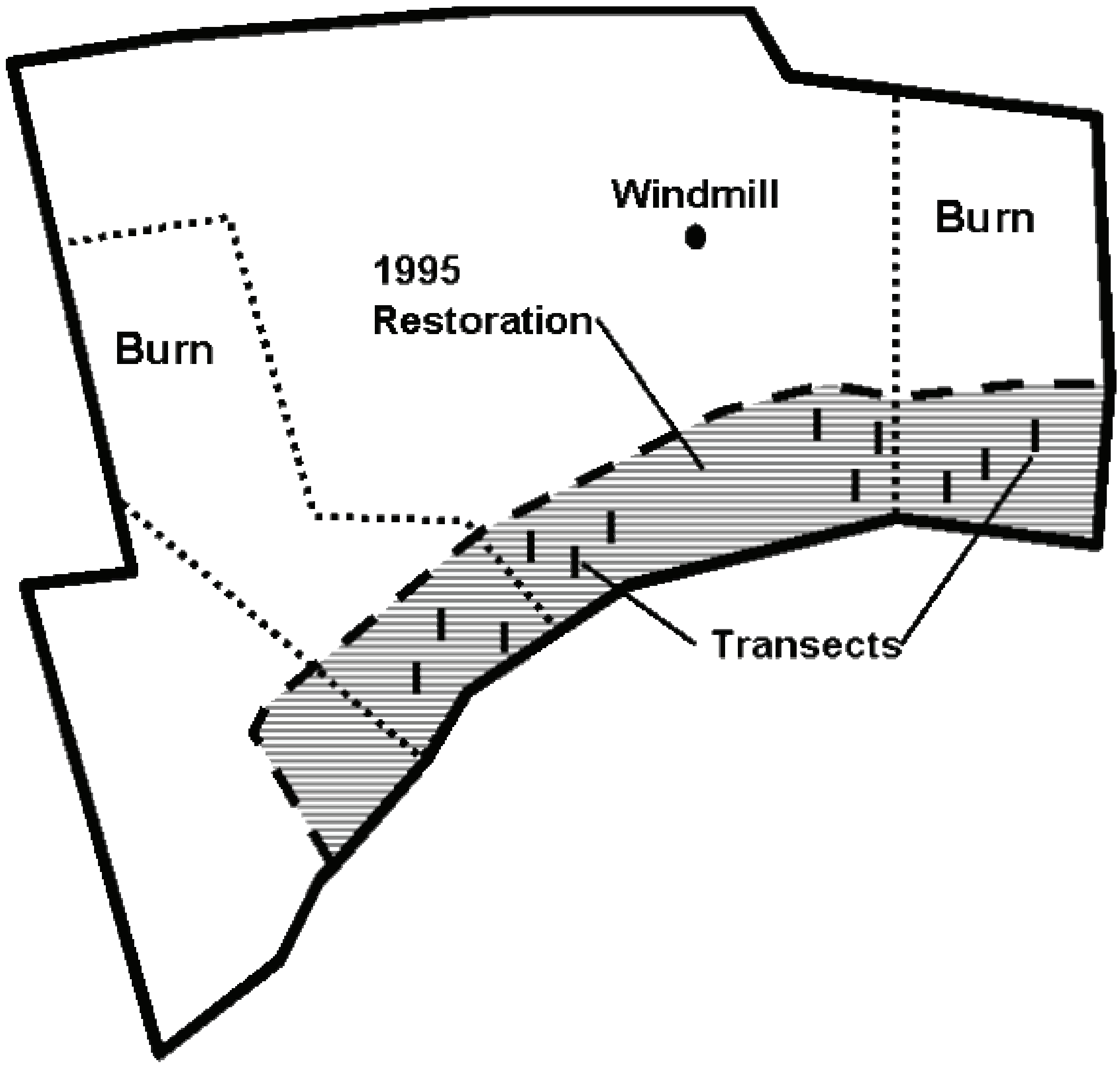

Figure 1. Map of the study area showing approximate locations of transects. Twelve transects were laid out – half in the burned sites and half in the unburned sites.

between May 10 and October 15. The cattle had unrestricted access to the entire site, including both the burned and unburned portions, for the entire season. We constructed several small exclosures on the site to allow us to look at ungrazed portions of both the burned and unburned treatments.

Within the 1995 restoration planting, we set up monitoring transects in both the burned and unburned areas to track the effects of grazing on nine prairie forbs. Those species were Illinois bundleflower, prairie clover (*Dalea purpurea* and *D. candida*), Canada milkvetch (*Astragalus canadensis*), rigid sunflower (*Helianthus laetiflorus*), Maximillian sunflower, tall boneset (*Eupatorium altoides*), heath aster (*Aster ericoides*), entire-leaved rosinweed (*Silphium integrifolium*), and rigid goldenrod (*Solidago rigida*). Five individuals of each species were marked along twelve 30 x 4m transects (six in burned areas and six in unburned; Figure 1). We marked each individual plant with a small piece of red wire around the base of the plant and recorded its location within the transect. We visited the plants about every ten days through grazing period. If a plant had been grazed, we counted the grazed tips and marked them with red fingernail polish to allow us to determine if the plant had been re-grazed between our visits.

RESULTS

The 2002 growing season turned out to be an unusually dry season that followed on the heels of two dry years. Specifically, we received about 44 percent of the average rainfall in June and 30 percent in July. Between May 26 and August 9, there were only two rainfall events of 0.4 inches (1 cm) or more. This had a severe impact on the vegetation at our site because of both the lack of rainfall and the resultant low groundwater level, which is normally within several feet of the surface. Most of the plants within the site went into dormancy during part of that summer.

The cattle followed our predictions early in the season by grazing mainly in the burned areas, but as the drought progressed, they began to spend more and more time grazing in unburned areas as well. However, even when much of the grass in the burned plots had gone dormant in July, the cattle still spent a high proportion of their time grazing there, to the point of eating the dried grass rather than the greener grass in the nearby unburned plots. Nevertheless, the unburned plots received more grazing than we would have expected in a year of normal precipitation, although it was much less than the burned plots.

The cattle grazed the unburned portions of the restorations more than the remnant prairie. They were likely attracted to the higher proportion of big bluestem and Indiangrass in the restorations. The remnants had strong components of switchgrass (*Panicum virgatum*) and Kentucky bluegrass (*Poa pratensis*), both of which are generally less preferred by cattle. In addition, there was a fairly continuous layer of previous years' standing dead grass in the remnant prairie, compared to a patchier layer in the restoration. Despite the level of grazing, enough ungrazed biomass was left in the unburned portions of the restoration after the end of the season to easily carry a fire next spring.

Of the nine species we tracked during the study, rigid goldenrod and tall boneset received no grazing at all in any of the treatments. Three other species--Maximillian sunflower, rigid sunflower, and rosinweed--were not grazed until very late in the season. Rigid sunflower was first grazed in mid-August, and Maximillian sunflower in early September. Both species had been affected by the drought and about two-thirds of the plants in each species had gone into early dormancy by late summer. About 90 percent of those that were still green were grazed. Other than one apparently anomalous event, no grazing was seen on rosinweed until early October, and then only a small number were grazed. About 36 percent of rosinweed plants in burned plots were grazed and 12 percent in unburned plots. The results on heath aster were difficult to interpret because unlike all the other species that we monitored, it was difficult to distinguish cattle grazing from that of other grazers and browsers. There was some defoliation on some plants nearly every time we looked, but while some of it was almost certainly from cattle, much of it may have been from other species.

The remaining three species--Illinois bundleflower, the two prairie clovers, and Canada milkvetch--were all grazed earlier and more frequently than the other six species. However, none of the forbs we tracked were grazed during the first 10-day period of the study. After that initial period we began to see some grazing of bundleflower, prairie clover, and milkvetch, primarily in the burned plots.

Illinois Bundleflower

Half of the bundleflower plants in the burned plots were grazed during the second period, but none in the unburned plots. The percentage of bundleflower plants grazed in the burned areas stayed fairly constant through the end of June and then began to drop off, presumably as re-growth slowed. During the five sampling periods before plants began to go dormant (late June), more than 20 percent of the plants in the burned plots had yet to be grazed, while others were recorded as being grazed up to four times. Thus, grazing pressure was patchy, even in the heavily used, burned plots. Grazing on bundleflower in the unburned areas increased through mid-July, corresponding roughly with the pattern we saw of cattle using those unburned areas more frequently.

Prairie Clovers

There was no apparent difference in cattle grazing preference between the two species prairie clover, so we combined them to ensure that we had adequate abundance in our plots. No grazing was seen on prairie clover until our second sampling period, by which time the grass in the burned areas had been cropped closely to the ground. Until a good proportion of the plants began going dormant in July, there were always much higher percentages of prairie clover plants grazed in burned areas than in unburned. As with bundleflower, grazing on prairie clover plants was patchy during the first five sampling periods, even in burned plots, with some plants receiving repeated grazing and others none at all. Additional data showed that plants in grazed plots generally had more flowers than those in exclosures, and plants in unburned/grazed plots tended to have more flowers than those in burned/grazed plots (Helzer and Steuter unpublished data).

Canada Milkvetch

Canada milkvetch plants were grazed less frequently than either prairie clover or bundleflower. However, we were only able to collect data from one of the two burn patches because of a lack of abundance of this species. Therefore, our data is much less robust for Canada milkvetch. Like prairie clover, milkvetch plants were grazed much more often in burned plots than unburned. As with bundleflower and prairie clover, some plants were grazed repeatedly during the first five sampling periods and others received no grazing at all.

In general, the cattle in our study seemed to greatly prefer grass to forbs of any kind and to prefer grazing burned areas to unburned (Figures 2 and 3). During the first week of grazing none of the forbs we tracked (or any others casually observed) were grazed at all. As the season wore on and grass became less available because re-growth was severely limited by a lack of moisture, grazing on forbs increased, first in burned and then in unburned areas. Some forbs, such as the three legumes (bundleflower, prairie clover, and milkvetch), were grazed fairly frequently throughout the season, particularly in burned plots, but others (the sunflowers and rosinweed) were ungrazed until late in the season. The latter situation raises the question of whether those three plant species were particularly attractive to cattle during the late summer season or they were grazed because more preferred forages were not available. Because the plants were less mature and leafy earlier in the season it seems likely that they were grazed later because preferred forages were unavailable at that time.

Figure 2. July 16, 2002. The unburned/grazed portion of the restoration. Note the tall stature of both the grasses and forbs, showing the very low rate of grazing taking place. (Photo by Chris Helzer/The Nature Conservancy).

Figure 3. July 16, 2002. The burned/grazed portion of the restoration. Note the short-cropped grass next to the tall, ungrazed forbs. (Photo by Chris Helzer/The Nature Conservancy).

2003 Data

In order to address the question of whether or not forb grazing was tied to the availability of grass forage, we collected additional data in 2003 on two remnant prairies close to the 2002 study site. Both prairies were grazed with the patch-burn method and 25 percent of each was burned in the spring of the year. The 195-acre (78-ha) Caveny tract was grazed at a low stocking rate (0.47 AUMs/ac) and the 115-acre (46-ha) Brown tract at a high stocking rate (1.47 AUMs/ac). We tracked purple prairie clover plants in both the burned and unburned portions using the same methods as the 2002 study. In 2003, May and June weather was cool and wet, but in July the drought returned in full force and by late July both pastures were extremely dry. Unfortunately, we were unable to collect data after the end of July, unlike 2002, when we tracked grazing patterns through October.

Grazing was patchy throughout the season in the burned portion of the Caveny tract (light stocking rate) and very light on the unburned portion. By late July, the burned area consisted of patches of full-size big bluestem in bloom intermixed with patches of very short grass. At the Brown tract (heavy stocking rate) the grass in the burned patch was nearly uniformly short early in the growing season and remained short during the entire study period, while the unburned patch was grazed only lightly.

There was a large difference between the two sites in the percentage of grazed purple prairie clover plants (Figure 4). On the Caveny tract, where the stocking rate was low, the cattle grazed

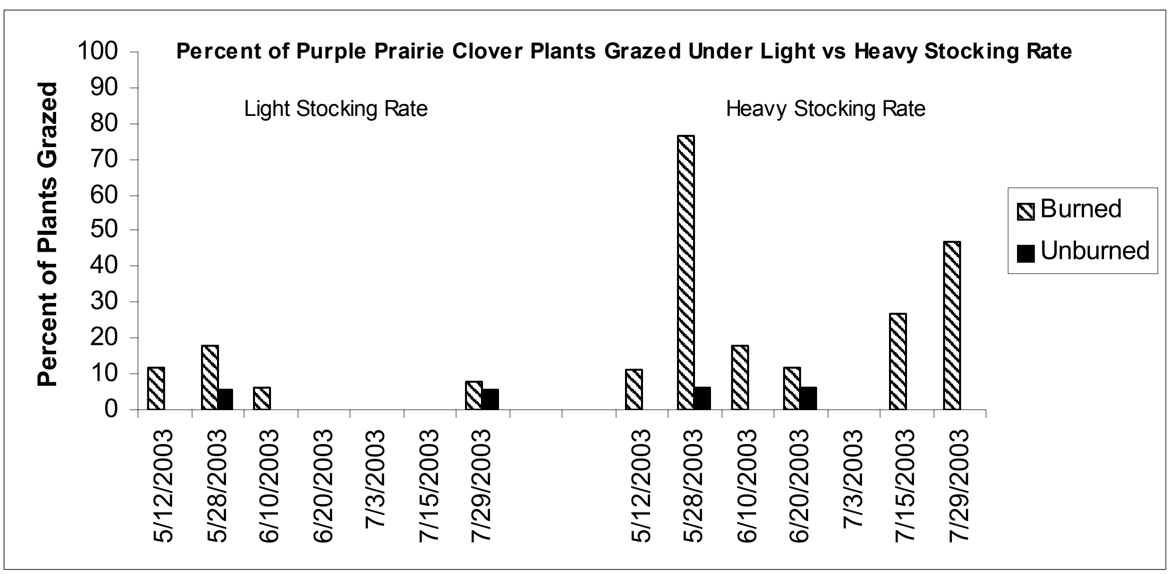

Figure 4. 2003 data collected from two remnant prairies, one grazed with a light stocking rate and one with a heavy stocking rate. The figure shows the percent of purple prairie clover plants grazed per sampling period within each prairie.

very few prairie clover plants in either the burned or unburned portion. However, on the Brown tract where the stocking rate was much higher, there was a high percentage of prairie clover plants grazed within the burned portion of the site. Although this data was preliminary and did not cover an entire season, it seems to support a correlation between grazing on forbs and the availability of grass, or at least a correlation between forb grazing and grazing intensity.

Next Steps

The severe drought during this short study makes it difficult to make generalizations from this data. We will continue to test the patch-burn grazing system on small high-diversity sites within improved experimental designs. Subsequent years will help us get a better grasp both on how cattle graze in high-diversity grasslands and the long-term responses of the plant community. We will continue to experiment with animal stocking rates to help clarify the relationship between available grass forage and the rate of forb grazing. We also hope to examine grazing behavior on restored sites compared to remnant prairies, and to look at forb selection in other kinds of grazing systems.

Maintaining high plant diversity in restored and remnant prairies will continue to be a tremendous challenge in the foreseeable future. Rather than being categorically bad for prairies, cattle may be one of our most flexible and valuable tools for managing that diversity. This study shows that under some conditions cattle select grasses rather than forbs, and that their selectivity can be managed by adjusting stocking rates. However, there is still much to learn and we need to continue experimentation with grazing on various kinds of sites. Unfortunately, we also need to work to remove the stigma that has been attached to grazing by many prairie conservationists who have seen only the negative effects of overgrazing on plant communities. Not until that stigma is removed can we truly move forward and realize the full potential of cattle grazing for prairie conservation.

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HAYMAN FIRE HOLISTIC PLANNING, IMPLEMENTATION AND RESULTS

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ABSTRACT

This paper takes a close look at holistic planning and implementation in the recovery of 137,000 acres burned by the Hayman Fire in 2002. After a pre-fire assessment of the conditions that lead to the Hayman Fire are discussed, the post-fire approach to restoration is discussed. Organizational structure, budget, priority setting and accomplishments are presented. Finally, some lessons learned are discussed.

INTRODUCTION

The following information addresses pre-fire, fire assessment and post-fire conditions related to the planning, implementation and ongoing restoration of the Hayman Fire on the Pike National Forest. This paper focuses on the methods used to address restoration needs resulting from the Hayman Fire and the lessons learned from implementing these methods. It is intended as an overview to a process that was dynamic and responsive to changes in budget, resources and environmental needs.

PRE-FIRE ASSESSMENT

The Upper South Platte watershed delivers to metropolitan Denver 80% of its drinking water. It contains some significant wildland/urban interface, especially along the U.S. Highway 285 corridor. Cumulative human effects have altered the structure, composition and landscape pattern of vegetation of this watershed.

Area Fire History

The ponderosa pine forests that dominate the Colorado Front Range montane zone and specifically the Upper South Platte watershed have adapted to frequent fire, and this pine montane forest is also part of the largest urban interface area in Colorado.

Previous Fires

Leading up to the Hayman Fire, severe wildfires were becoming more common in the Upper South Platte watershed:

Buffalo Creek, 1996 - nearly 12,000 acres,
Turkey Creek, 1998 - 350 acres
Hi Meadow, 2000 - nearly 11,000 acres,
Snaking Fire, 2002 - 2,312 acres,
Schoonover Fire, 2002 - 3,860 acres
Hayman Fire, 2002 - 137,526 acres.

Fire Weather

The entire state of Colorado and the Front Range in particular, were in severe drought conditions. Records show the progressive decline in available moisture for the area over the previous three-year period. Because of precipitation conditions, wildland fire potential was projected to be above average. This prediction was borne out by large fires burning at high intensities, the extreme difficulties experienced with suppression efforts, and the explosive nature of resent fires.

HAYMAN FIRE ASSESSMENT

The Hayman Fire was reported on June 8, 2002 at 4:55p.m. at an approximate size of 20 feet by 20 feet. On June 8, the winds were reported to be from the south-southwest, ranging from 15 to 20 miles per hour (mph). The temperature was approximately 85˚F, the humidity ranged from 5 to 8 percent with a Haynes Index of 5. Within 18 minutes of initial attack, the fire was 10 acres and exhibiting erratic fire behavior.

The Hayman Fire is a good example of a fire burning under the influence of all the extreme factors that affect fire behavior. Fuels were flashy and dry from a 3-year drought, with all time live and dead fuel moisture lows and abundant, continuous and available fuels. Terrain was very steep and inaccessible once the fire crossed County Road 77 and the Tarryall River. The terrain was on a west and very dry aspect and oriented to a south and southwest wind direction. The area was prime for the large fire spread event that occurred on June 8, 9 and 10. Under these conditions, the homes in the area were in a poor position from a fire behavior standpoint and many were minimally defensible.

Fire weather on June 9 had high temperatures, low relative humidity, (9% at 8:00 a.m., down to 5% that afternoon) a National Weather Service Red Flag Warning for high winds 30 to 40mph throughout the day, and the highest possible index for extreme fire conditions.

Response Time

During the first two hours of the initial attack of the Hayman Fire there were: 4 air tankers, 1 Type-1 helicopter that carried 2,000 gallons of water with each load, 2 Type-3 helicopters that could carry 85 gallons in each of their buckets, 1 Type-1 hotshot crew, 1 Type-2 handcrew, 2 5-person handcrews, 7 Type-6 fire engines, 2 water tenders, and miscellaneous supervisory and safety personnel. Fire managers determined that a total of 100 firefighters was the maximum amount to safely operate considering the confined area, very limited access and supervisory/employee span of control.

The Hayman Fire burned 19 miles on June 9th, and expanded to an estimated 62,000 acres. No firefighters would have been safe under these conditions. The terrain in which the fire was burning the first several days precluded use of fire engines and water tenders. No amount of people on this fire would have stopped it during the first several days. Because of the steep, inaccessible terrain and the fire's erratic behavior, firefighters were not ordered to attack this fire.

Suppression Strategy

The suppression strategy during the initial phases of the fire was to concentrate on evacuation of the residential areas because of the extreme fire spread. As wind directions changed and the wind driven aspect moderated, suppression strategies became more aggressive and successful in later days.

POST-FIRE ASSESSMENT

Suppression and Emergency Rehabilitation Costs

Suppression costs were estimated at over 32,000,000 dollars and Burned Area Emergency Rehab (BAER) expenditures were estimated at $21,500,000. Long-term restoration needs exceed $50,000,000. BAER accomplishments included:

Hillslope stabilization – 38,000 ac. completed at a cost of $19 million.
Noxious weed treatment and inventory – 4,500 acres completed at a cost of $1 million.
Road maintenance, repair and closures – 156 miles and 139 sites completed at a cost of $306K.
Hazard tree felling - around all private inholdings.
Site visits with private landowners - nearly 100 site visits with affected private landowners to address flooding and sediment damage.
Treatment effectiveness monitoring – $340,000 with RMRS and CSU.
Early warning system and flood hazard signing throughout burned area.

Watershed

A high erosion hazard existed on 95,900 acres or 70% of the burned area. Major ash flows impacted water quality at Cheeseman Reservoir. This reservoir continues to experience sedimentation.

Wildlife

Catastrophic wildfires such as the Hayman Fire have the capacity to drastically alter wildlife habitats and influence the persistence of species within and near the burned area. Some of the effects of the fire will likely be beneficial due to increased habitat diversity. However, those areas that experienced high severity burns may have negative impacts for years.

The fire affected at least five federally threatened and endangered species: the Canada lynx, Preble's meadow jumping mouse, bald eagle, Mexican spotted owl, and the Pawnee montane skipper.

Human Impacts

Approximately 5,430 people were evacuated during the Hayman Fire. Five firefighters lost their lives in a vehicle accident en route to fight the fire. Approximately 600 structures were destroyed in the path of the fire (133 homes, one business and 466 out buildings).

Air Quality

Smoke from the Hayman significantly degraded the air quality in the area surrounding the fire and throughout the Denver and Colorado Springs metropolitan areas. Health warnings were issued.

RESTORATION TEAM

The Forest Service formed a 13-member Hayman Restoration Team that continued working on rehabilitation efforts on the 137,760 acre Hayman Fire for two years. Many accomplished in a short period of time occurred thanks to the Team's efforts and the assistance of individual Forest district personnel, detailed personnel from other forests, other federal, state and county agencies, the Coalition of the Upper South Platte (CUSP) who organized volunteers and managed volunteer projects, and the vast amount of volunteers.

The team identified ten critical priority areas to be addressed:

1. Visitor Information/Control – Area closure, gating, information signing
2. Noxious Weed Control – Inventory and treatment
3. Road Repair – Road assessment, reconstruction, erosion control, culverts
4. Reforestation – Seed Collection
5. Landlines - reestablishment
6. Recreational Facilities – Decommission/Conversion
7. Habitat Restoration – Wildlife and watershed projects
8. Public Affairs – Media, Volunteer Coordination, Congress
9. Watershed Monitoring – Completion of TMDL, research coordination
10. Salvage & Hazard Tree Removal

Funding: Congress approved $7 million for all restoration work nation-wide in Fiscal Year 2003. The majority of this money went to California. As a result, the USFS Rocky Mountain Region carved appropriated dollars from its FY 2003 budget to supplement restoration funding of the Hayman Fire Restoration program at approximately $3 million for FY 2003. Funding from the Rocky Mountain Region of the Forest Service in FY 2004 was $1 million.

Using resource specialists from the Pike and San Isabel National Forests, Comanche and Cimarron National Grasslands (PSICC), the Hayman Restoration Team worked on the following projects:

Reopening the Hayman Fire burn area. The team completed assessments cleared hazard trees and posted signage to enable reopening of portions of the Hayman Fire burn area. As additional roads and trails were deemed safe, their opening was phased in.

A roads and trails analysis was completed to recommend what roads need to be reconstructed, closed or maintained in order to protect the watershed. An Environmental Assessment was prepared to implement those recommendations. Crews finished clearing hazard trees from the roads and trails. Periodically, roads that were re-opened had to be temporarily closed for additional restoration work resulting from flooding in the area

Noxious weed treatment & miscellaneous contracts. Contracts in excess of $450,000 were completed to spray for noxious weeds that invaded after the fire and $300,000 for inventory of additional weeds. Road maintenance and repair contracts totaling $500,000 and landline/fence reestablishment of $100,000. Four hundred fifty bushels of cones were collected for reforestation. Wildlife habitat surveys and watershed restoration projects were conducted.

Burn Area Timber Salvage Environmental Assessment (EA). The restoration team prepared an Environmental Assessment for harvesting up to 10,000 acres of trees that were killed by the fire. Contracts with timber companies were written and administered to remove the timber.

Monitoring research activities. The Hayman Restoration Team was responsible for monitoring and tracking the wide range of requests received, from educational institutions and USDA Forest Service Research, to conduct research and experiments within the Hayman burn area. The experiments include the study of fire's impacts on wildlife, watersheds and health of surviving trees, as well as the fire's effects on a range of other resources.

Impacts on forest facilities. The Rehabilitation Team initiated public scoping and the environmental assessment to determine which recreational facilities in the burn area should be decommissioned, repaired or reconstructed. They removed four developed camping sites that were adversely impacted by the fire.

Major Hayman Restoration Team Accomplishments

Aerial seeding and mulch applied covering over 4,500 acres in 2003

Third application of straw mulch on 2,000 acres was completed in 2004

Repair of 25 road washouts and 40 culverts were installed of repaired.

Seven hazardous tree roadside salvage sales completed totaling 1.4 million board feet.

Hazard trees were cleared on 290 miles of road

Cleared hazard trees along all trails within the burn.

Several timber salvage sales were completed, including the 3,582 acre Burnt Cedar salvage sale, 490 acre Flickenstein Timber Sale, 179 acre Painted Rocks Timber Sale, 460 acre Cemetary Timber Sale, and the 177 acre Molly Salvage Sale.

Approximately 20 research projects continue to examine the effects of the Hayman Fire

SOME LESSONS LEARNED

1. **From the time the fire ended, how long did it take to decide to take action (i.e., go forward with salvage sale) and develop a plan to move forward?**

Before the smoke cleared the intent to move ahead with salvage was always the plan. A categorical exclusion was done immediately to begin salvage along roads. The fire was controlled 7/18/02. The plan was to hire a Restoration Team to oversee the National Environmental Protection Act for the general forest area salvage. The permanent Team Leader arrived 10/4/02. The Forest Leadership Team debated the pros and cons of contracting out the Environmental Assessment (EA) for the first two weeks of October. A contract was written to do the EA. Bids received 11/05/02 were astronomical and the decision was made to do the EA in-house. Fifteen Core Team members were identified and the Salvage Sale Scoping Letter went out 11/22/02.

Lessons Learned:

a. If you don't have a million dollars to hire out the NEPA analysis, do it yourself. Your people would have to provide most of the on-the-ground knowledge anyway. If you don't already have good field data to provide a contractor, you are better off doing it yourself anyway.
b. Do not have a large EA Core Team. Smaller is better, five, six core members are able to do the most efficient work.

2. From the time you decided to take action, how long did it take to form an ID team?

ID team members to fill key positions were identified from local Forest employees. Certain positions were difficult to fill (wildlife biologist, fisheries and economics) because there are so few of them on the forest and they are in such high demand to complete existing projects. A letter of commitment from the Forest Supervisor solidified the team rather quickly, within two weeks.

Lessons Learned:

a. Identify potential team members early and have the Forest Supervisor sign a letter of commitment that states their participation on the Core Team is the highest priority.
b. Certain positions are not found on the Forest (Economist) and getting the same level of commitment from the Regional Office is just as important for the team to be complete.

3. How long did it take to complete the NEPA analysis and make a decision?

The timeline developed by the EA Core Team was followed as planned. The Scoping Letter went out 11/22/02 and the Decision Notice was issued 6/2/03.

Lessons Learned:

a. Commit to a timeline with due dates for specialist reports and periodic meetings of the core team to see that everything is progressing as planned.
b. Spend time early to define a clear and simple purpose and need. If you make the purpose and need to complex, you will have difficulty during the NEPA analysis and generate additional appeals.

4. How long did it take from making the NEPA decision to offering the first timber sale?

The Decision Notice was issued 6/2/03 and the sale could have been awarded on 10/9/03.

Lessons Learned:

a. Even so our timber salvage project was not appealed, things can be delayed from unexpected events. Teller County wanted the Forest Service to pay the county for use of the county roads and provide a bond for any damage as well as maintain these roads during the sale period. Even so Teller County was notified during the scooping process and during the comment period, their intentions to pass load limits on haul routes were unexpected. It took two months to work through these issues with the county. The sale contract was executed December 9, 2003. The operator started on the sale January 14, 2004.

5. **What lessons did you learn as a result of your experience with this project, good and bad – what worked well, what didn't work well, what were barriers to making speedy progress in project planning and implementation, what would you do differently?**

   Lessons Learned:

a. Adequate funding is very unlikely so choose the most economical method to accomplish the NEPA analysis. If you can do an EA and not an Environmental Impact Statement, do it in-house.
b. Line Officer involvement in the Core Team Meetings is a must. There are so many decisions the EA Core Team must make that will impact the Decision Notice that without the involvement of Line Officers you will encounter many setbacks.
c. Keep the Purpose and Need simple. Do not include anything more than necessary to get the salvage accomplished. Do not base the Purpose and Need on economics, do not include reforestation, do not include other projects or needs.
d. Have a good team leader that can lead the process and knows NEPA. It is best to have core team members that have done previous EA's and know the process.
e. Keep the ID core team small but include other auxiliary team members to fill the roles needed in specialist areas.
f. If pre-fire information would have been available on stand conditions, slope, roads, access, property lines, etc. it would have made initial identification of salvage areas easier. The team had to compensate with remote GIS information and then attempt to ground truth. Using remote GPS information allowed the team to speed up the salvage project but caused more adjustments to the sale areas on-the-ground.
g. In the event of a large catastrophic fire, the best way to accomplish the restoration is to identify a dozen specialists from the forest and reassign them to a Restoration Team, full-time. Advertise for temporary 1 year appointments to fill in behind these forest employees. This allows a restoration team to hit the ground running. They know the land, they know the policy and procedures for the area and they know who to contact in the community.

UNCOMPAHGRE PLATEAU PROJECT: THE EVOLUTION OF A COLLABORATIVE IN WESTEERN COLORADO

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ABSTRACT

This paper overviews the development of the Uncompahgre Plateau Project (UP) over a seven-year period and provides an in-depth discussion of the collaborative resource planning process for two priority watersheds on the Plateau, Spring Creek and Dry Creek, and an illustration of one implementation project. The purpose of this overview is to illustrate the formative stages or steps that collaborative groups commonly go through – from initial assessment of the issue to implementation of on-the-ground projects – as a framework for understanding how to develop a collaborative process. The discussion on collaborative planning and implementation explains the process the UP group pioneered for restoring a vegetation mosaic at a landscape scale and provides an illustration of an implementation project. These examples are intended to illustrate the benefits of working collaboratively and planning holistically on a landscape scale.

INTRODUCTION

Collaboration has emerged over the last few decades as a new environmental movement that is having a decided impact on public land management. Weber characterizes it as "Grass Roots Ecosystem Management" (GREM), and differentiates it from three earlier environmental movements that shaped the development of federal resource management agencies: preservationism, conservationism, and contemporary environmentalism. GREM shares some features with these earlier movements. They all grew out of the public's concern over the ecological degradation that accompanied rapid industrialization, and they all share a belief in "limits to growth" – "that there are biophysical constraints on the ability of industrial societies to continue historical trajectories of population growth and resource consumption (Weber, 2000)." However, GREM differs from these earlier movements on key issues, such as the role of science and government in defining the balance between nature and culture, the appropriate geographic and temporal scale for management, how best to manage for complexity and uncertainty, and how to evaluate success.

While representing distinctively different world views and approaches, all three of these earlier movements were primarily concerned with the economic market's failure to regulate resource use in a sustainable manner, so they concentrated on expanding the government's role in resource management to limit or regulate industries' uses of resources. They promoted top-down approaches that focused on defining how best government could limit or regulate industry. In contrast, GREM supports a more democratic, grass roots, results oriented approach to resource management that emphasizes on-the-ground improvements. The focus in GREM is on community. It grew out of a desire to find a more fruitful way of negotiating environmental issues that did not polarize communities and lead to gridlock and to find more creative ways of using market-based mechanisms to address environmental issues. At the same time, GREM calls for a more holistic, integrated approach that demands a broader geographic and temporal scale for resource management, and an adaptive approach to manage for the complexity and uncertainty of ecological systems.

Collaboration Defined

As it is practiced, collaboration as a process can take a wide range of forms, but it can be generally defined as a cooperative form of communication, problem solving, and decision making (Wondelleck and Yaffee 2000). More successful groups tend to be "place-based" – or local to the area of interest. They are multiparty, with members holding different, and often time, competing interests, but they share a sense of interdependence and shared responsibility for the outcomes of their management decisions. Through deliberation, they are able to define "common ground" and build a shared vision that provides a coherent, holistic approach for managing across jurisdictional boundaries.

Groups can employ a range of strategies for developing action plans and working together, but more successful groups generally use a collaborative learning framework to develop a rich body of site-specific knowledge and ground their decisions in high quality science and local knowledge. They emphasize on-the-ground ecological conditions – both in defining the problem and in developing the solutions. Their decision making process is generally much more democratic than the traditional technocratic process of land management agencies. By engaging different perspectives and bringing multiple resources to the table, they are able to develop and implement more creative solutions.

Because collaborative processes must be adapted to the context, there are no definitive guidelines for developing a collaborative process. However, researchers have synthesized the formative process stages or steps that collaborative groups tend to go through based on their analysis of multiple groups. The framework employed in this overview was adapted from Wondelleck and Yaffee (2000). The steps they outline include:

1. Make an Initial Assessment
2. Develop a Common Purpose
3. Develop a Process
4. Learn Collaboratively
5. Develop an Action Plan
6. Implement and Manage Adaptively

The following section traces a seven-year period in the evolution of the UP in western Colorado to illustrate these formative process stages UP went through from conception to project implementation.

UNCOMPAHGRE PLATEAU PROJECT

UP is a landscape level ecosystem restoration project that consists of 6 formal partners, USDA Forest Service (USFS) - Grand Mesa, Uncompahgre, Gunnison National Forest (GMUG), Bureau of Land Management (BLM) - Uncompahgre Field Office and Grand Junction Field Office; Colorado Division of Wildlife (CDOW), Western Area Power Administration (Western), Tri-State Generation and Transmission Association, Inc. (Tri-State), and the Public Lands Partnership (PLP). PLP is a consortium of local officials and interest groups such as loggers, recreationists, environmentalists, and ranchers from four local counties: San Miguel, Ouray, Montrose, and Delta.

The purpose of the UP partnership is to assist in planning, coordinating, funding, and facilitating restoration activities across jurisdictional boundaries. It supports, but does not supersede, management authority on any federal, state, or private lands. The jurisdictional boundaries for the project encompass roughly 1.5 million acres on the Uncompahgre Plateau, which runs northwest to southeast from Grand Junction down to Ridgway (Figure 1). Roughly 75 percent of the Plateau is public lands (1,126,359 acres), and 25 percent is private (387,552 acres). The USFS (544,777 acres) and BLM (571,992 acres) are the two largest land management agencies, with an additional 9,590 acres under State management.

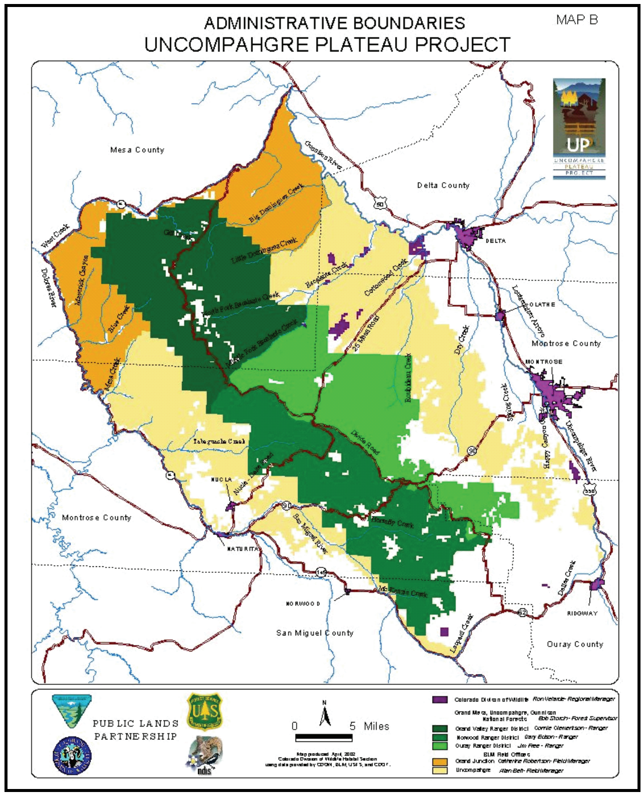

Figure 1: Administrative Boundaries Uncompahgre Plateau Project

STEPS IN THE DEVELOPMENT OF UP

Step One (Part 1): Initial Assessment

- Identify and analyze the principle issues, key stakeholders, interests, and positions
- Gather/develop baseline information on areas of common ground versus potential controversy
- Define the issue/problem inclusively

UP began rather informally with a concern for single species management. The CDOW had been tracking mule deer populations on the Plateau since the 1980's and had seen a significant decline over a 20-year period. In 1998, a CDOW wildlife biologist began discussing his concerns with the mule deer population decline with the BLM wildlife biologist and USFS range conservationist. They resolved to work cooperatively to manage mule deer across jurisdictional boundaries. The three agency specialists continued to discuss the issues more widely and tried to build support for their idea. In an informal meeting with CDOW biologists and some local concerned sportsmen, the CDOW director promised $500,000 for mule deer habitat projects if they group could develop a collaborative process for managing the deer.

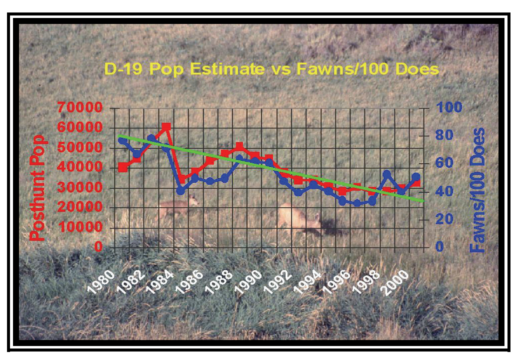

Figure 2: Colorado Division of Wildlife. Mule Deer Population Studies (1980-2000)

Step One (Part 2): Internal Convening

- Convene internal discussions to help address and clarify key issues, philosophy, approach, roles of key players in the agencies
- Use the opportunity to build staff capacity
- Create political and institutional support for the process

This informal promise of money for treatment projects galvanized the group, and they expanded their efforts to garner support for their vision of working together across jurisdictional boundaries. They formed a group called the Uncompahgre Mule Deer Project (UMDP) to discuss how they might collaboratively manage for mule deer. They also engaged more agency staff in the discussions, and started meeting regularly to establish a strategy for cross-jurisdictional management.

In 1999, this larger group developed a more formal process, with facilitated public meetings with PLP members to discuss a collaborative management structure. Through the course of their discussions, the group came to realize that they could not effectively address mule deer decline as just a single species management issue. Much of the ongoing research pointed to declining habitat as a major factor in the mule deer decline. Additionally, there were other indicators that the ecosystem was in decline, such as rapidly declining Gunnison sage-grouse populations, declining range conditions, and poor overall vegetation condition, to name just a few. Consequently, the group decided that they needed to broaden the focus of their collaborative management effort from single species management to landscape health and ecosystem restoration. They changed the name of the group from the Uncompahgre Mule Deer Project (UMDP) to the Uncompahgre Ecosystem Restoration Project (UERP).

Step Two: *Collectively*…Develop a Common Purpose

- Define the problem inclusively
- Develop a vision statement that identifies what you want to achieve
- Develop a purpose statement that defines the issue to be addressed
- Clarify your relationships

Throughout 1999, the group continued to meet to develop a formal collaborative structure. They created a purpose statement and began to more formally define their relationships in the management process.

> *Purpose Statement: To restore and maintain the ecosystem health of the Uncompahgre Plateau in Western Colorado, using best available science and a collaborative approach between communities and agencies.*

Their main concern was the overall decline of the ecosystem of the Plateau and its social and economic impacts. However, the initial concern with mule deer habitat still loomed large as a primary consideration. The commitment of dollars for habitat treatment provided additional incentive to put a collaborative plan in place to address habitat.

Step Three: *Collectively*…Develop the Process

- Establish a charter or operating agreement
- Clearly define ground rules, roles, responsibilities, authorities, legal and political sideboards, other constraints
- Strategically use facilitation
- Design the process to fit the situation

The coalition got a boost in 2000, when the PLP received a $750,000 grant from the Ford Foundation for a community-based forestry restoration project. PLP created Uncompahgre/Com Inc.(Un-Com), a 501(c)3 non-profit to administer the Ford Foundation funds. PLP made a portion of this money available for staff positions for UERP, and it provided a means to pass through money from the agencies for collaborative management. With this infusion of money, the four partners developed and signed a Cooperative Agreement (CA) and a Memorandum of Understanding (MOU) to formalize their working relations.

They developed a formal structure, with a technical committee at the core consisting of a technical representative from each of the four partners. This committee had final decision-making authority and gave direction to the project. Oversight came from the Executive Committee, which was made up of top level managers from each of the partners. Un-Com was the financial arm of the partnership, and they funded and managed four staff positions: technical coordinator, educational coordinator, financial coordinator, and grant writer. These coordinators took their direction from the technical committee and were responsible for developing a collaborative interface between the technical committee and what was called the collaborative council – which represented the collaborative forums for exchanges and learning among the agency managers, scientists, modelers, and public.

Step Four (Part 1): Emphasize Collaborative Learning and Sharing of Information

- Share information
- Clarify differences
- Identify knowledge gaps
- Jointly gather/assess information
- Develop a common knowledge base
- Educational Forums/Field Trips, Research Conferences

In 2001, Un-Com hired a technical coordinator and educational coordinator. With the addition of these staff positions, the coalition began to take on a public identity. The decision was made to change the name of the collaborative from Uncompahgre Ecosystem Restoration Project to Uncompahgre Plateau Project because some of the members felt the acronym UERP did not lend itself to creating a positive public image. Hence the name was changed to UP, and the group developed logos to solidify their new identity. The two coordinators immediately began an aggressive outreach program. They raised public awareness by presenting at multiple civic group and interest group meetings. They developed websites and newsletters, and began to host field trips and public meetings.

Many of the initial public meetings were co-hosted with the GMUG. The forest was beginning public outreach for their forest plan revision, and had identified the Plateau area as one of their "landscape working group" areas. The GMUG initiated their first public outreach meetings in the communities around the Plateau, and UP continued to co-host public meetings with the GMUG for about nine months, until the forest shifted their focus to their remaining landscape working group areas.

At the same time, the collaborative began laying the foundation for joint management. They contracted a joint BLM/USFS landscape assessment to provide baseline data for planning, and they contracted research to address some of the questions that had been raised in the discussions about how to most effectively management that ecosystem. Specifically, they solicited fire history research projects to better understand the local fire and disturbance regime in the pinyon-juniper woodland and Ponderosa pine forest ecosystems. They also funded additional mule deer studies.

Step Four (Part 2): Provide Multiple Opportunities to Participate

- Public meetings, open houses, workshops
- Strategic planning meetings
- Ongoing consultation with key constituency groups
- Educational forums/field trips, research conferences
- Websites and interactive tools
- Surveys, polls and questionnaires

One of the challenges that UP faced was the wide range in technical knowledge among the participants. Some of the public participants were very knowledgeable about ecosystem functions and forest and/or range management, in general. Others had little understanding of basic ecosystem functions, but they had a strong attachment to the land and a definite interest and stake in the outcome of the management decisions. After the initial meetings to raise awareness about the collaborative, the outreach focus shifted to providing meaningful venues for discussions about the major issues and ongoing research and for building an understanding about the area and natural processes. The purpose was to help develop a common vocabulary and understanding among the participants. For example, UP hosted field trips to burn sites to illustrate succession in various vegetation types or to discuss burn area rehabilitation plans. Other field trips provided a forum to explain and discuss monitoring. The researchers provided public presentations on the results of their studies.

Not all participants want to continually attend meetings or field trips, however, so information was also made available on the website and in newsletters. Participants could ask questions or provide responses by contacting one of the coordinators.

In 2002, the final building blocks of the collaborative structure were put into place. The financial coordinator and the grant writer were hired, and the UP Plan was completed, which spelled out the working relations of the partnership. A monitoring program and a native seed program were also initiated to identify native seeds on the Plateau and develop local sources.

Step Five: *Collectively*…Develop an Action Plan

- Explore and decide on management objectives to achieve group's goals
- Develop and explore options that meet multiple interests
- Select criteria for choosing among options
- Develop clear and measurable action items
- Specify who/what/by when/if not/what

There was increased pressure from within and without the group to move toward implementation – as the whole goal of the project was to put treatments on the ground. Much of the initial funding that had been committed was going to be redirected if it was not spent, and people were tiring of meeting to develop a collaborative structure and build a common vision and shared learning base. So the group worked intensively through 2002 on pioneering a process for interagency planning and implementation.

As a foundation for joint management, agency specialists built compatible GIS (Geographic Information System) vegetation data bases to create a seamless vegetation layer for planning and analysis. They began working with modelers to develop a landscape dynamics model for the Plateau, which was a spatially explicit model of disturbance regimes from fire, insect, disease, and drought in the major vegetation communities. The goal of the modeling was to help refine the assumptions about the historic range of variation in these communities.

Over the course of the year, working groups and subgroups - including scientists, agency employees, and interested public members – met to develop the process. The groups ranged from forty to five people, depending on the degree of technical expertise required. They developed a process that became the foundation for the Spring Creek/Dry Creek Vegetation Strategy Plan. This process will be described in greater detail in the following section.

In 2003, the Interagency Spring Creek/Dry Creek Vegetative Strategy Plan was completed, and the group began intensive project planning for implementation.

Step Six*: Collaboratively…*Implement and Manage Adaptively

- Establish implementation structures that facilitate coordinated actions
- Develop adaptive management protocols to ensure continued learning
- Establish mechanisms to ensure accountability and conflict resolution
- Develop feedback loops and share with the broader public

In 2004, the collaborative had over 3 million dollars in funding. Western and Tri-State joined as formal partners after participating informally for over a year. The group was able to complete nearly 30,000 acres of treatments. The primary and secondary objectives for these treatments were for wildland urban interface – including power line and private property protection, Gunnison sage-grouse habitat restoration, mule deer habitat restoration, and overall ecosystem restoration. In addition, they were also able to expand the native seed program and continue to work with the modelers to complete the historic range modeling.

PLANNING PROCESS

The ultimate goal of the collaborative was to restore the ecosystem of the entire Plateau, but they decided the most judicious approach was to start with complete restoration of a sub watershed as a pilot. They felt it would be necessary to first pioneer a process for restoration and then replicate it in other areas.

1) Develop Objectives to Meet Goals

To develop this process, the group had to first decide on quantifiable, measurable objectives to define "restoration" or a desired future condition (dfc) that would provide a basis for developing and evaluating restoration management actions. They began by identifying the multiple issues of concern for the partners, which included:

- Wildland Urban Interface (WUI)/Fuels management
- Endangered species
- Mule deer decline
- Power line protection
- Livestock forage
- Habitat for woodland dependent birds
- Natural and functioning ecosystems

They identified the *vegetation mosaic* as a common connection among these issues that was quantifiable and directly related to restoration and could be influenced by management activities. Vegetation mosaic is the range of vegetation types and age classes, and their spatial and temporal distribution on the landscape. It is composed of patches within a matrix and characterized by the proportion of age classes and patch shape, size, edge, and arrangement. Since different age classes and their arrangement on the landscape provide different human uses and natural values, many of these issues could be simultaneously addressed across the landscape.

The basic premise of the planning process was that the dfc could best be characterized by identifying the range of desired vegetation mosaics across the landscape that most efficiently and cost effectively met these multiple interests. The foundation of the strategy was a set of mosaic objectives that help identify what and where the vegetation problems are and how to address them. These vegetation mosaic objectives and vegetation condition objectives provide a framework for a management strategy, including:

Designing Treatments
Implementing Treatments
Establishing Monitoring Criteria
Evaluating Treatments
Assessing Progress

2) Prioritize Watersheds

Next, the group had to decide which sub-watersheds would best serve as a pilot area. They evaluated all the watersheds within the project area in terms of overall degradation, potential for restoration, and social and economic concerns. They selected Spring Creek and Dry Creek as priority watersheds for treatment. The two watersheds combined totaled approximately 220,000 acres, with an intermix of USFS, BLM, State, and private land ownership. They embodied some of the Plateau's more pressing issues, such as wildland urban interface (WUI) risks from fire, rare species, mule deer habitat, and forest health problems.

3) Identify Landscape Units

The group then identified areas that would provide a foundation for developing mosaic objectives. They divided up the Dry and Spring Creek Watersheds into what were called "landscape units" – which were units that share common vegetation communities and natural disturbance patterns (Figure 3). The six major landscape units included:

Aspen/Spruce Fir
Mountain Shrub
Ponderosa Pine
High-elevation pinyon-juniper/shrub
Low-elevation pinyon-juniper/sagebrush
Saltdesert shrub

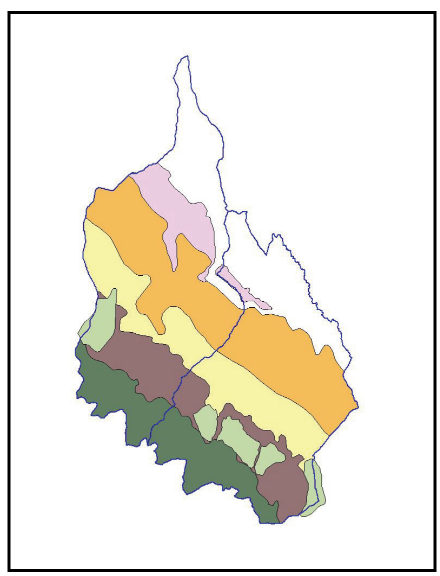

Figure 3: Spring Creek/Dry Creek Landscape Units

4) Develop Mosaic Objectives

Using available science and local management expertise, they developed a desired "natural" vegetation mosaic objective (Figure 4) for each landscape unit (see Appendix A). These mosaic objectives were based on the groups' best understanding of historic disturbance regimes and ranges of variability for each of the major vegetation types.

What became clear as the group proceeded is that the disturbance regimes would be different in these watersheds, which were on the eastern slope of the Plateau, from those on the western slope of the Plateau. Through further discussion, the group decided to break the Plateau up into four quadrants along a north/south and an east/west axis. There was a slight difference in elevation between north and south and a distinct difference in moisture and fuels. The storm patterns, winds, aspect, and slope from east to west was varied enough to have a pronounced affect on the disturbance regimes.

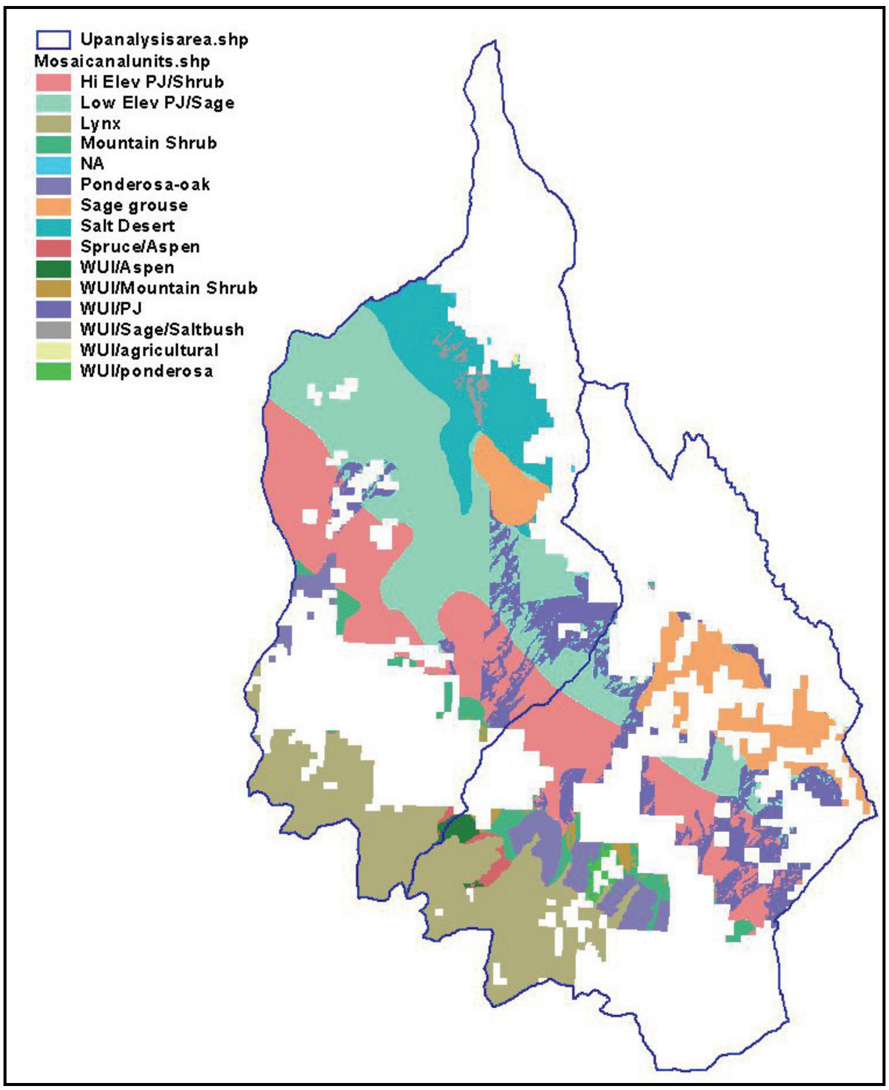

Figure 4: Spring Creek/Dry Creek Mosaic Objectives Map

4) Identify Management Issues

Once the group had developed mosaic objectives for the landscape units, they identified the land management issues in the two watersheds that were dependent on vegetation mosaic. The list included all the issues that had been raised in relation to the decline of the ecosystem:

Fire management – including safety and protection of life and property
Transmission line protection
Mule deer habitat
Woodland dependent bird species habitat
Threatened and endangered species habitat
Overall forest/range health

5) Prioritize Management Issues

The group then prioritized these management issues to determine which would dominate or be "drivers" where issues overlapped. There was a great deal of discussion over the competing interests. Many felt that the best strategy was to restore a naturally functioning system in order to support all the interests. However, agency specialists pointed out that they had no latitude in managing for wildland urban interface issues and threatened and endangered species because of national directives.

As a result, the issues identified as "drivers" for selecting vegetation mosaic objectives on the landscape included in order of priority: urban interface and power line protection, Gunnison sage grouse and potential lynx habitat, and overall forest and range health. The remaining issues became "modifiers" for projects during the design phase, instead of driving an objective.

6) Create Mosaic Objectives Map

To identify where these "drivers" were on the landscape, a mosaic objectives map (Figure 4) was created by overlaying a GIS layer for each of the management issues onto the landscape units. The resulting map has large areas dedicated for management as natural vegetation mosaics, with smaller proportions in sage-grouse, lynx, wildland urban interface, and powerline protection categories. This was a surprising result given the number and scope of management issues in these particular watersheds. The result was one benefit of the collaborative process, as competing interests were brought together and found overlap among their issues.

7) Create "Driver" Mosaic Objectives

The group developed a vegetation mosaic objective for each of the "drivers". These objectives were based on a combination of local conservation plans, appropriate science, and management experience that was used to best describe amounts and arrangement of each age class in order to optimize vegetation for that particular driver. In some of the areas, the mosaic objectives were very similar to mosaic objectives for natural landscape units. For example, in the objectives for wildland urban interface in Ponderosa pine, the landscape unit objectives for early and early-mid seral stages would be similar to the "driver" objectives for fire management, with a few modifications of stand structure. In other areas, such as pinyon-juniper and sagebrush, the "driver" objectives called for more early and early-mid seral stages and smaller patch sizes.

8) Create Current Vegetation Mosaic Map

As a baseline for evaluation, they created an existing vegetation mosaic map (Figure 5) to profile the current mosaic. They created this vegetation age class map by cross-walking the best available GIS vegetation data from each of the agencies. Then they analyzed the map in GIS for the age class proportions and patch sizes.

9) Identify Departures

They evaluated the existing vegetation mosaic (Figure 5) against the desired mosaic objectives to identify areas of departure from the desired mosaic objectives. This analysis identified where there were departures from historic vegetation ranges or sage grouse and lynx habitat and urban interface objectives.

Based on the map, the group knew where they had departures and generally what action they needed to take to move the landscape toward the desired mosaic condition – e.g. recruit more early seral in small to medium size patches. Then they developed a list of potential management actions to accomplish the desired condition. For example, in Ponderosa pine where they wanted to move the landscape toward a later seral stage, they could use mechanical and/or prescribed burning to improve the stand structure and recruit more older-age trees.

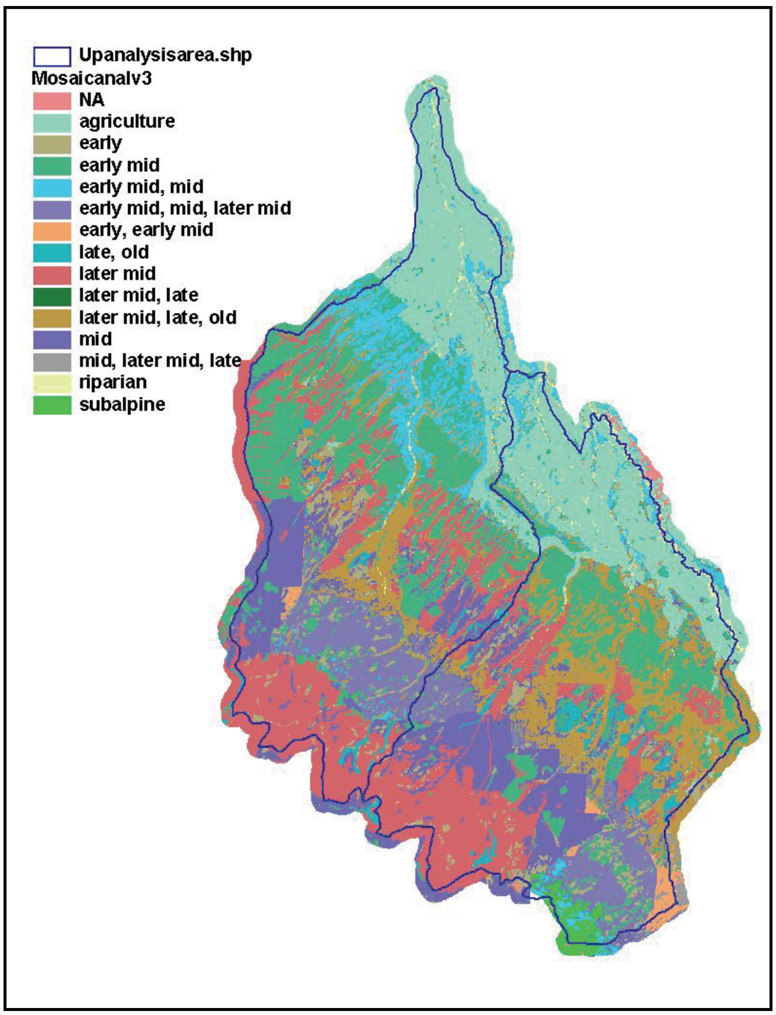

Figure 5: Spring Creek/Dry Creek Current Vegetation Mosaic Map.

10) Identify Projects

Proposed treatment projects were identified that could meet the goal of restoration of the watersheds. Many projects also addressed vegetation condition problems along with mosaic restoration. All the partners submitted potential projects developed to meet their objectives. These projects were screened to see if they would meet the desired mosaic objectives, and modified as necessary. Project designs were also modified where it was possible to address multiple interests. For example, the project outlined in the next section discusses a project designed primarily for the purpose of power line protection that was modified to also accomplish mule deer habitat enhancement. Additional projects were developed to bring project acreage in line with the objectives for the watersheds (see Attachment B for a sample project table).

Once the projects were identified, different partners selected the projects best suited for their specific interests. Progress is tracked through a web-based interactive map that shows progress to date and the responsible party. The Vegetation Strategy includes a monitoring component, and identifies a committee to evaluate progress and determine if changes are needed. The UP technical committee has the larger oversight role on monitoring and evaluation.

IMPLEMENTATION

Highway 90/Government Springs Power Line Protection

The power line protection treatments in the Highway 90 and Government Springs areas provide an example of the collaborative nature of the implementation. A total of about 1,000 acres were treated primarily for power line protection in the two areas. The federal and state partners all contributed to varying degrees on these treatments. Tri-State paid for the cultural clearance. Western provided linemen to help lay out projects. BLM designed and implemented protection projects on BLM lands adjacent to the power lines, and CDOW designed and implemented mule deer habitat projects on BLM lands close to the power lines and residences.

Primary Objectives

The primary objectives for the power line treatment were:

- Create a safe environment for fire fighters
- Reduce potential for heat impacts to facility
- Reduce potential for smoke impacts to facility
- Remove hazardous trees that might impact facility

Create a safe environment for fire fighters

To create a safe environment for fire fighters, it was imperative to provide for safety zones and escape routes. It was also necessary to create defensible space along the power line to give fire fighters an area in which to fight the fire. And finally, it was important to minimize the risk of arcing due to smoke impacts in this defensible space area to ensure the safety of the fire fighters. To accomplish this objective, the treatment was designed with a mosaic of "tiled" seral stages.

Reduce potential for heat impacts to facility

To reduce the potential for heat impacts directly to the towers and power lines, the potential fire intensity and fire residence time had to be reduced. To accomplish this objective, the stand structure was modified to reduce the overall amount of vegetation and reduce the heavy fuels in the stand.

Reduce potential for smoke impacts to facility

Reducing the total amount of vegetation, and reducing vegetation with inefficient combustion that smoldered or produced a lot of smoke, such as standing pinyon–juniper and oakbrush and heavy dead and down woody fuels, further reduced the potential for smoke impacts to the lines.

Secondary Objectives

Reduce current visual impact of the power line corridor
Improve mule deer winter range

Reduce visual impact

Creating variation among the size and shape of treatment units and curving or feathering the edges of the units reduced the visual impact of the power line corridor (Figure 6). Modifying the tiling pattern away from the line and leaving some trees and islands within the units also served to minimize the visual impact.

Figure 6: Spring Creek/Dry Creek Post-Treatment Power Line Corridor

Improve mule deer winter range

In conjunction with reducing the visual impacts, modifying the size of treatments to less than fifty acres and providing savannah and leave islands within the units also met the objectives for improving mule deer habitat. The treatments were also designed to leave enough distance between treatments to provide cover.

BENEFITS OF COLLABORATION

Working collaboratively at a landscape scale changed the group's management vision. When the collaborative was initially conceived as a single species management concern, agency specialists had talked about large scale prescribed burns and mechanical treatments to improve mule deer habitat. After the intensive planning process for landscape scale restoration, that vision was no longer admissible.

In fact, working collaboratively at a landscape scale provided a means to overcome some obstacles and avoid repeating some past mistakes, which looked a lot like the large scale treatments people were initially proposing. It was typical for vegetation treatments to be planned within a single jurisdiction for a single purpose. They were generally planned and implemented project by project, with no overall mechanism to monitor spatial and temporal cumulative impacts. As a result, there was no way to monitor or learn from mistakes, and there was no broader vision for working with natural processes and within historic bounds.

By contrast, the broader vision and more refined planning process from this landscape scale collaborative effort provided numerous benefits. There was a much greater shared knowledge base and understanding of natural systems, and a mechanism in place to monitor and learn from treatments. All the projects were compatible with the bigger picture and were designed across administrative boundaries to have cumulative benefits, both spatially and temporally. Projects could also be planned to meet multiple needs and could be tailored for a specific area, and multiple resources could be pooled to accomplish projects. This effort exponentially improved each of the partners' ability to manage for healthier landscapes.

BARRIERS TO COLLABORATION

The UP project is just one of many examples of how a collaborative approach can lead to more creative solutions and provide a foundation for landscape scale ecosystem management. While it is not hard to sing the praises of this new approach to resource management, it is advisable to sound a note of caution about the potential barriers to engaging in such an undertaking and the long-term viability of this approach.

The greatest barriers UP participants faced were inconsistent resources and shifting priorities, which is typical of other collaborative groups. A recently published synthesis of research on collaboration points to these exact same issues as barriers that other collaborative efforts have faced (Sturtevant et. al., 2005). As they note, there are many factors that influence the success of a collaborative effort, but the most critical are stable, long-term funding; strong leadership and management support; and effective group facilitators and coordinators. "Sixty percent of studies analyzed in one survey mentioned the importance of funding and effective leaders, facilitators, and coordinators (Sturtevant et al., 2005)." Other critical factors cited include community and agency capacity to collaborate, dedication of agency staff time to collaborative projects, and effective monitoring and evaluation programs.

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Attachment A

South-East Quadrant of Plateau
High Elevation Pinyon-Juniper/Shrub Mosaic Objectives

Unit #/Name High Elevation Pinyon-Juniper/Shrub –SE Plateau

Written Description of Unit: Pinyon dominated stands with or without Juniper, mountain shrub, grass/forb in understory.

Likely Presuppression Fire Regime and Mosaic: This type of vegetation can burn in some landscape positions, burns are stand replacement events (observation from Karen E., has seen stands of this with burn evidence in them on other areas of Plateau). 200-300 yr fire intervals reported by Romme in Mesa Verde, substantiated here by some of Karen's data collected locally. Fire behavior tends to be short duration, large scale, high intensity, or little (one tree) fires. Herbaceous fine fuels probably not a factor in PJ fire…instead they are wind driven crown fires. We assume a reverse J distribution—with a hump at the big end of the x axis—lots of small events, very few larger events, a bit more big events (this based on fire behavior and size distribution of pj fires in region over last 25 years). Patchiness effect on fire spread—debatable. Inspection of vegetation map shows little fragmentation in this zone, indicating large fire events probably rare. The infrequent, large fire likely to be wind driven, long thin patch like Fruitland fire—constrained by topography, especially the drainages. Mosaic likely to be few large patches. Little known about Ips beetle Small patches of trees killed common, rare large patch events correlated with drought cycles—(Karen assumption), probably their biggest effects on trees growing on fine, deep soils or marginal sites.

Desired Mosaic: Fire and disturbance size distribution—assume patches very small or quite big

Patch Size	Early Grass-forb 0-10 yrs	Early Mid Shrub-grass 10-50 yrs	Mid Shrub-tree infilling 50-150 y	Later Mid Tree stem exclusion, touching canopies 150~350y	Late Tree stand thinning-even size/age, indiv. Dropping out 250~400y	Old Understory reinitiation, starts old growth characteristics 350~600y
% of Unit	5	5	15	15	30	30
Small 0-5 acres	30	30	20	M	M	M
Medium 5-100 acres	10	10	20	M	M	M
Large 100+ acres	60	60	60	M	M	M

*Use M to designate which stage is the Matrix

** If important, use A to designate abrupt edge, F to designate feathered edge, U for undulating edge

List of references or data from which mosaic objective derived

See narrative above, get specific references

List of Uncertainties/Assumptions/that should be addressed or may cause objectives to be changed: We assume a reverse J distribution—with a hump at the big end of the x axis; long thin patches; Mosaic likely to be few large patches; Ips beetle effects with rare large patch events correlated with drought cycles—(Karen assumption), probably their biggest effects on trees growing on fine, deep soils or marginal sites. Assumption on time of seral stages. Assumption that proportions of transition times of stages give us idea of amount of area needed of each.

Attachment B

Sample Project Table from Spring Creek/Dry Creek Vegetation Strategy

UP Project Number	Mosaic Area	Acres in Project Area	Treatment Type	Project Details	Project Modifiers
35	High Elevation PJ/Shrub Dry Creek 35	704.8	Maturing	Allow 80% of early mid and 90% of mid to mature to later stages	Pinyon jay, BTG warbler, Brewers sparrow, Mule deer, G flycatcher
35	WUI/High Elevation PJ/Shrub Dry Creek 35	492.9	Mechanical	Create 3-10 ac patches of early seral on 30% of mid and later seral stages	Pinyon jay, BTG warbler, Brewers sparrow, Mule deer, G flycatcher
42	High Elevation PJ/Shrub Dry Creek 42	283.5	Rx Burn	Burn small patches of old projects that are in early and early mid stage w/in project boundary. Retain all mid stages for recruitment to later mid.	Pinyon jay, BTG warbler, Brewers sparrow, G flycatcher
43	High Elevation PJ/Shrub Dry Creek 43	232.2	Mechanical	Remove invading PJ, improve sage condition, seed natives, avoid treating later stages	Pinyon jay, Aberts squirrel, Brewers sparrow, Black Bear, Lewis woodpecker
44	High Elevation PJ/Shrub Dry Creek 44	618.6	Mechanical	Break up old treatment block by actively thinning 60% of mid seral stage in patches to advance seral stage	Pinyon jay, Brewers sparrow, Black Bear, Lewis woodpecker, G flycatcher

TUNDRA RELOCATION ON THE
GUANELLA PASS CONSTRUCTION PROJECT

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ABSTRACT

The revegetation of a high altitude road reconstruction site is challenging work. Harvesting, transporting, and transplanting of high altitude sod in large quantities was accomplished on an old parking lot on Guanella Pass along the road connecting Georgetown and Grant, Colorado during the late summer of 2004. New ideas and state of the art equipment enhanced the success of this revegetation project. Approximately one acre of tundra sod was transplanted as far away as one mile, with virtually no mortality. Slope revegetation work outside the parking lot boundary will also be discussed.

INTRODUCTION

American Civil Constructors was awarded a $20 million contract to repair and upgrade the road from Georgetown to Grant, Colorado. The contract, administered by the Federal Highway Administration, includes the reconstruction of 8.8 miles of roadway over the summit of Guanella Pass, where the elevation is over 11,000 feet above sea level. Work began in the spring of 2004 and will finish in the summer of 2006. The project includes mechanically stabilized earth walls to support the widened road and drainage improvements, including 121 structures and 100 new culvert crossings.

Revegetation components of the contract include: (1) installation of 2,400 sq. meters of erosion blanket, (2) hydraulically seeding and mulching 80 acres, (3) Biosol and humate treatments, (4) application of bonded fiber matrix on 8 acres, and (5) relocating 4,400 sq meters of tundra sod. Also the plan calls for the planting of 21,000 willow poles, 4,400 containerized plants, and 1,167 transplants. The erosion and sediment control portion of the contract includes: (1) installation of 8,000 meters of silt fence and (2) installation of 1,450 meters of erosion logs. The monetary value of the *entire* revegetation portion of the contract is approximately $880,000.

HISTORY

The Guanella Pass scenic byway follows an historic 23 mile long wagon trail, which linked the mining towns of Grant and Georgetown, Colorado. These two communities were centers of the silver mining era that boomed in the mid to late 1800's. This area is situated in the Mineral Belt which runs from near Ward, CO, through Leadville, CO, terminating in the San Juan Mountains in the southwest part of Colorado. Guanella Pass is located in Clear Creek County and is locally known as the "Silver Heritage Region—a historic designation that allows the area to be culturally, historically, and naturally enhanced for the benefit of all.

PROJECT DESCRIPTION AND RESULTS

The existing older parking areas on Guanella Pass were developed and expanded for the benefit of hikers and other outdoor enthusiasts. Unfortunately, older parking lots were not graded for drainage, and some were susceptible to significant erosion, which, in turn, affected some of the receiving streams and down-gradient lakes in the Guanella Pass area. American Civil Constructors' first task was obliterating and re-contouring the existing parking lots. Boulders were then placed in random fashion to discourage further degradation and use.

A new paved parking lot was engineered and staked out to replace the other lots described above. Tundra sod was harvested from the new lot in large pieces--approximately 4 feet square "sheets", with 6 to 9 inches of soil supporting the rooting portion of the sod. See Figure 1. Good quality soil existed in the 6-9 inch layer beneath the sod. The remaining layer of soil below about 9 inches was very coarse sand that did not transport or facilitate transplanting very well, so it was discarded. Salvaged tundra sod pieces were very heavy--up to 1000 pounds each. On shorter hauls (up to 400 feet) from harvest site (new parking lot) to placement site (old parking lots), tracked skid steer loaders with special attachment buckets were utilized to harvest and transport the sod. One machine had a 4 foot cutting edge bucket, which provided effective harvesting of the sod. The other skid steer had a flat steel plate mounted on the bucket to receive and move the harvested sod to the planting area. Arctic Willows in the sod harvesting areas were transplanted in-place along with the sod. Willows clumps were used along the road to discourage people from disturbing the reclaimed site and a rail fence was also constructed around the reclaimed area to restrict foot traffic.

Some of the tundra sod planting sites were a mile or more from the sod harvest sites. Because of these longer distances, transport with conventional skid steer loaders was not practical. For the longer hauls, John Deere Diesel Pro Gators with dump beds were used to transport the tundra sod. See Figure 2. Harvesting sod involved dumping a 4 foot square piece onto the Skid steer's flat plate and then it was transferred to the Pro Gators which negotiated the longer haul more efficiently. During sod transport operations haul roads, traveled by the skid steer loaders, were graded regularly to eliminate bouncing and jarring that tended to break sod into pieces. Transplanting took place as described above. Low ground pressure balloon tires on the Pro Gators were gentle on the terrain, and cushioned the ride so that sod pieces did not break apart.

Prior to placing the sod, topsoil was imported from the sod harvest sites and spread evenly on the obliterated parking lots. Topsoil replacement areas were hand tilled and smoothed out to afford good contact between tundra sod and topsoil. Rough edges on the sod pieces were hand trimmed with spades to provide a tight fit for the sod "puzzle". After placement, sod joints were "top-dressed" with loose topsoil salvaged specifically for that purpose. Boulders were placed randomly within the transplanted sod areas to establish a more "natural" appearance.

Following the harvest, transport and placement of sod on old parking areas, the next challenge was to place remaining harvested sod on 2H to 1V cut and fill slopes. Obviously, sod placement on steep slopes complicated the transplanting operation. Skid steer loaders were utilized to gently place the sod on slopes and a lot of hand work was required to fit it all together.

Following sod transplanting efforts, a dry weather period ensued. A truck mounted hydroseeder was left on-site, so the sod could be watered regularly Because of the high altitude and the short growing season, fall arrived within a short time of transplanting. Concern was expressed by project managers that transplanted sod would not have time to establish before the growing season was over. American Civil Constructors personnel theorized that the deeper layer of soil under the plants would allow the sod to survive as though it had not been moved. The following spring, there were signs of stress in the willows and transplanted sod right at the time plants were starting to "green up". However, after two more weeks, observations indicated the transplanted sod and willows were similar in appearance to the surrounding undisturbed vegetation. Wildflowers in the newly transplanted sod bloomed beautifully that spring—signaling a successful tundra transplanting operation.

Figure 1. Conventional Track Mounted Skid Steer Loader with 4 Feet Square Piece of Tundra Sod

Figure 2. Pro Gator Unit Used to Transport Tundra Sod on Long Hauls

ALTERNATIVES TO TOPSOIL IN MINE SITE RESTORATION: A CASE STUDY IN THE COEUR D'ALENE (IDAHO) MINING DISTRICT

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ABSTRACT

A series of demonstration plots was installed to evaluate soil amendments that facilitate revegetation of waste rock piles in the Coeur d'Alene Mining District of northern Idaho. The amendments included biosolids, composts, log yard wastes, and two liquid soil restoration treatments. The plots were evaluated for revegetation success, runoff, and soil stabilization during the 2003, 2004, and 2005 field seasons. A stable plant cover was established on each plot, although the extent of coverage varied significantly among the amendments. Fertility status had a strong impact on species distribution and extent of unseeded vegetation. For example, high fertility amendments promoted a grass-dominated/low forb profile with a low content of unseeded vegetation, while the low fertility amendments promoted a more diverse grass-forb mixture with greater susceptibility to unseeded vegetation establishment. The species distribution within most plots also changed over time. In grass-dominated plots, wheatgrass declined with a concurrent increase in brome and fescue. The remaining plots exhibited significant increases in yarrow, white clover, and milkvetch. Large increases in unseeded vegetation were also observed, particularly in numbers of black medic, sweet clover, and knapweed.

INTRODUCTION

Remediation projects throughout the Coeur d'Alene Mining District consume large quantities of topsoil. The State's RI/FS team has estimated that approximately 600,000 cubic yards of growth media will be needed to complete all anticipated remedial actions. Available topsoil resources are insufficient to meet this need. Hence, there is an ever present need to identify topsoil alternatives that are locally available and can promote a self sustaining vegetative cover.

A variety of materials have been investigated for their suitability as soil amendments, or as components of a manufactured soil. For example, surface applications of composts, consisting of municipal solid waste and biosolids, have been used to control erosion and revegetate steep slopes on the Quall Cherokee Reservation of North Carolina (EPA 1997). Li et al. (2000) found that composted, limed biosolids were superior to conventional fertilizer plus lime treatments in revegetating a Cd/Zn contaminated soil near Palmerton, PA. A biosolids/yard waste compost was used by Glanville et al. (2004) to establish a vegetative cover on new highway embankments. A related study showed that these compost treatments provided erosion control prior to plant establishment (Persyn et al. 2002) but also resulted in increased runoff P concentrations (Glanville et al. 2002).

Zeng et al. (1993) reported that log yard fines (LYF) improved soil physical properties including water holding capacity, bulk density, and porosity. Although orchard grass biomass decreased with increasing LYF application rates, alfalfa growth improved, suggesting that the high C/N impeded grass growth but did not adversely affect legumes. Brown et al. (2005) reported that municipal biosolids mixed with agricultural limestone were beneficial for establishing a plant cover on mine tailings near Leadville, CO. This study also observed decreased plant diversity (i.e. a high percentage of grasses relative to forbs) on the biosolids-amended plots. This effect was attributed to the high N content of the biosolids.

A range of solid materials, including municipal biosolids, woody debris, wood ash, pulp and paper sludge, and compost was tested for reclamation of metal-contaminated mine wastes at the Bunker Hill Superfund site in northern Idaho (Brown et al. 2003). These researchers found that biosolids in combination with wood ash, with or without the other materials, were able to promote a vegetative cover in acidic soils with elevated Pb and Zn. This study also tested two commercially available soil restoration products, Biosol and Kiwi Power, which are designed to enhance microbial activity and stimulate nutrient cycling. While Brown et al. (2003) found superior initial germination in the Biosol and Kiwi Power treatments, these amendments were not effective in sustaining plant growth.

Other studies suggest that enhanced bioactivity can be a key factor in successful reclamation of severely disturbed soils. Noyd et al. (1996) reported that plant cover and biomass, percent seeded species, and mycorrhizal infectivity were positively associated in reclamation of iron ore tailings. Furthermore, decreased mycorrizal activity was identified as a major factor in poor plant establishment following long-term stockpiling of topsoil during mining operations (Schuman 1999).

Even this cursory look at the literature makes two points very clear – first, each individual waste material, or combination of materials, possesses unique properties (e.g. nutrient content, cation exchange capacity, microbial activity) that can be exploited for reclamation purposes. Second, each reclamation project exhibits specific challenges (e.g. heavy metals, low fertility, steep slopes, limited accessibility) that must be overcome for successful revegetation. Thus, additional research - particularly field studies conducted under differing environmental and topographic conditions - can only lead to an improved understanding of what works, what doesn't, and why.

The objective of this study was to use a series of demonstration plots to examine a variety of amendments, including biosolids, composts, log yard wastes, plus the Biosol and Kiwi Power treatments, for efficacy in site revegetation, as well as erosion control and impacts on nutrient runoff.

METHODS

Site Description

The Silver Dollar Mine site is located west of Osburn, Idaho (47º 30.22' N; 115º 59.39' W). The site is dominated by a waste rock pile produced during mine development and sorted from the ore during the mining process. Milling and smelting activities took place off-site so heavy metal concentrations are a minor issue for plant growth relative to low fertility. The waste rock pile rests on a north-facing slope at an elevation of about 2500 feet. Average total monthly precipitation ranges from 1.5 inches in July to 4.5 inches in November, with a total annual precipitation of 38 inches. Average monthly temperatures are 32.9/21.3ºF (max/min) in January and 78.6/47.2ºF in August.

Site Preparation/Plot Installation

The waste rock pile was regraded to a 2:1 (H:V) slope and ten plots (20' X 100') were installed with a berm (3' X 2') separating each plot. Runoff flumes and an erosion trap were installed at the bottom of each plot. The western- and eastern-most plots were reserved for controls; the remaining plots were assigned to participants on a random basis. Project participants (Table 1) were solicited and selected by IDEQ.

Installation of the plots began 25 September 2002 and concluded 23 October 2002. A brief description of amendment materials and application rates is listed in Table 1. Each participant selected their amendment rate and method of application and each plot was seeded, either by hand or by hydroseeding, using a standardized seed mix (Table 2).

Table 1. Demonstration plot amendments, rates, and project participants.

Plot	Amendment	Affiliation	Rate
A	Control (topsoil)	IDEQ	40 yd^3 of topsoil was spread to a depth of approximately 6”
B	Biosolid + Woodash (0.75:1)	Coeur d'Alene Wastewater Treatment Plant	26 yd^3 of Class B biosolids mixed with wood ash (0.75:1) was spread to a depth of approximately 4”
C	Potlatch Log Yard Waste	Potlatch Corp.	Log yard fines (<3/4”) were mixed with urea fertilizer (10 % v/v); 48 yd^3 was spread to a depth of approximately 6”
D	Kiwi Power	Quattro Environmental, Inc.	Fertile Fibers Plus, Kiwi Power, Strong Hold + Tacker and Atlas Soil Lock was mixed and applied using the hydroseeder
E	Eko Compost	Eko Compost	20 yd^3 of compost was spread to a depth of approximately 4”
F	Glacier Gold Compost	Glacier Gold, LLC	20 yd^3 of compost was spread to a depth of approximately 4”
G	Biosol	Rocky Mountain Bio Products	83 lb Biosol Mix (7-2-3) plus 5 lb Wood Fiber Mulch seed mix was applied using the hydroseeder. Wheat straw was spread over plot and 4 lb Guardian Tackifier applied.
H	Glacier Gold Log Yard Waste	Glacier Gold, LLC	20 yd^3 of log yard waste was spread to a depth of approximately 4”
I	Biosolid + Woodash (1:1)	Coeur d'Alene Wastewater Treatment Plant	26 yd^3 of Class B biosolids mixed with wood ash (1:1) was spread to a depth of approximately 4”
J	Control (fertilizer)	IDEQ	50 lb of fertilizer (16-16-16), seed mix, and tackifier were applied with the hydroseeder. Bluegrass straw was applied as a mulch on bottom-half of plot

Table 2. Seed mix used on the Silver Dollar Demonstration Plots.

Common Name	Scientific Name	Rate (lb PLS/ac)	Weight (%)	Minimum (%)
Slender wheatgrass	*Elymus trachycaulus* ssp. *trachycaulus* var. Revenue	14 lbs	22.3	21.9
Idaho fescue	*Festuca idahoensis* var. Joseph	8 lbs. 7 oz	13.4	13.2
Sheep fescue	*Festuca ovina* var. Covar	7 lbs	11.1	10.9
Mountain brome	*Bromus marginatus* var. Bromar	7 lbs. 11 oz	12.2	12.0
Meadow brome	*Bromus biebersteinii* var. Paddock	8 lbs. 7 oz	13.4	13.2
White Yarrow	*Achillea millefolium*	11 oz	1.1	1.1
Blue flax	*Linum lewisii* var. Appar	4 lbs. 3 oz	6.7	6.6
Rocky Mountain penstemon	*Penstemon strictus*	1 lb. 6 oz	2.2	2.2
White dutch clover	*Trifolium repens* L.	8 oz	0.8	0.8
Canada bluegrass	*Poa compressa*	11 oz	1.1	1.1
Big bluegrass	*Poa ampla* var. Sherman	1 lb. 7 oz	2.3	2.3
Canby bluegrass	*Poa canbyi* var. Canbar	1 lb. 6 oz	2.2	2.2
Cicer milkvetch	*Astragalus cicer*	7 lbs.	11.1	10.9
Fireweed	*Epilobium angustifolium*	1 oz	0.1	0.1
Weed seed				0.5 (Max)
Inert and other crop				1.5 (Max)

Plot Assessment

The plots were inspected monthly from April through August during 2003, 2004, and 2005. Percent germination and relative growth (leaf stage) were evaluated at the beginning of each growing season. In addition, a qualitative assessment of leaf color was made as this can provide clues to nutrient sufficiency/deficiency and plant stress due to diseases and pests. Uniformity of coverage was also noted for each plot.

Plant coverage was assessed using two methods. Plant frequency was determined using a Cover-point optical projection scope (ESCO Associates, Boulder CO). One hundred points were recorded at 1 m intervals along a randomly located transect in each plot. Each point identified an individual plant, rock, bare soil, or litter. Plant density was assessed at two sampling points per plot, 10 m in from the bottom and top of the plot. The specific location of the sampling point was randomly selected - the observer faced away from the plot and tossed a 1-m^2 PVC hoop over their head into the plot. Each individual plant within the hoop was tallied and identified, including plants that were not a component of the original seed mix. Hence, unseeded species were included in the calculation of plant frequency and density. The mean value of the replicate density assessments is reported in the following tables and figures.

Surface runoff was collected monthly in 2003, 2004, and 2005, and each sample analyzed for ammonia-N, nitrate-N, and orthophosphate. In 2003 and 2005, a composite (3x) soil sample was collected from each plot. A standard fertility test (ammonia-N, nitrate-N, available P and K, organic matter, and pH) was determined for each sample. All laboratory work was conducted at the University of Idaho Analytical Sciences Laboratory.

RESULTS

Growth Media Properties

The pH of the unamended waste rock was 8.3 and the amended plots ranged from 6.3 to 8.3 with the 1:1 woodash/biosolid mixture exhibiting the highest pH. Overall, the pH was relatively consistent within a given plot throughout the study period. The organic matter content varied from ~1% in the controls and liquid-based soil treatments to 15-34% in the solid-based amendments. Each of the amendments increased the available P and K content, with the extent of increase being strongly dependent on the nature of the amendment. Available P values ranged from 2 to >600 ug/g while available K (sodium acetate extractable) ranged from 80 to 1000 ug/g. Thus, each of the amended plots contained adequate to excessive P and K relative to typical plant requirements. In several plots, the application rates are well in excess of vegetation needs, thereby increasing the potential for nutrient runoff. The nitrate-N level in the unamended soil-waste rock was 0.7 ug/g while the amended plots exhibited nitrate-N concentrations ranging from <5 to >800 ug/g. Similarly, ammonia-N was initially low and varied significantly among the amended plots, ranging from <4 to >60 ug/g. These values are equivalent to 20 - 3500 lb available N per acre [available N (lb/ac) = (ug/g ammonia N + ug/g nitrate-N) x 4]. A more detailed discussion of growth media properties may be found in a related paper in these proceedings (McGeehan 2006).

Nitrogen and Phosphorus in Surface Runoff

As would be expected, runoff nitrogen concentrations closely reflected the available ammonia and nitrate content of the amendments. The highest runoff ammonia- and nitrate-N concentrations (5.3 and 34 mg/L, respectively) were observed in the Potlatch Log Yard Waste plus fertilizer. This is undoubtedly due to the very high rate of urea fertilizer (10% v/v) applied to the log yard waste, which resulted in an extremely high available N content (~3500 lb/ac) in 2003. Significant N runoff was also observed in the

Eko Compost and Biosolids + Woodash II plots. Runoff N from the Kiwi Power, Glacier Gold Compost, Glacier Gold LYW, and Biosol plots was much lower in comparison. However, the Glacier Gold Compost plot exhibited the highest P runoff values, particularly in 2003 and 2004. Additional discussion of surface runoff from the demonstration plots may be found in McGeehan (2006).

Erosion Control

Sediment runoff and surface erosion was minimal on the demonstration plots throughout the 3 year study. A minor amount (5-10 kg) of eroded sediment was observed in early spring 2003, and small rills were evident in the lower-half of the Kiwi Power, Eko Compost, and Glacier Gold Compost plots. In addition, one large rill was observed on the upper-half of the Control-Fertilizer plot in early spring 2003. It is noteworthy that the eroded sediment from this plot was retained by the berms, per the plot design, and no sediment was observed in the trap. No additional erosion was observed in 2004 or 2005 on any of the plots.

Vegetation Coverage: Frequency vs. Density

Each of the amendments supported a plant cover during the first year of the study (2003), with one exception. The Potlatch LYW + Urea Fertilizer plot exhibited almost zero plant growth during 2003. This is likely the result of over-application of urea fertilizer, which resulted in ammonia phytotoxicity and minimal germination.

Plant frequencies were relatively low (<50%) in 6 out of 10 plots in 2003 (Figure 1A). These relatively low frequencies were consistently associated with the lower fertility plots (e.g. Kiwi Power, Glacier Gold Compost, Biosol, and Glacier Gold Log Yard Waste). Available N ranged from 20 to 50 lb/ac in these plots while available P was low in some plots and adequate in others (data from McGeehan, 2006). Plant frequencies were higher (>70%) on the high fertility plots (e.g. Biosolids, Eko Compost, and Potlatch LYW + Urea in 2004 and 2005). These plots contained very high available N (>900 lb/ac) and P (>120 lb/ac) contents at the onset of the study.

Total plant frequency increased between 2003 and 2004 but did not change significantly in most plots (6 of 10) between 2004 and 2005 (Figure 1A). Instead, plant frequency tended to level off in the 75 to 95% range. The total frequency data also indicate that the proportion of grasses did not change significantly between 2004 and 2005. That is, the frequency with which of grasses were encountered remained relatively constant with a given plot. However, it is important to make the distinction between relatively constant total grass frequencies vs. changes in the distribution of individual grass species. The detailed frequency plots (Figures 2, 3, 4, 5, and 6) clearly show significant changes in species distribution (i.e. the relative contents of wheatgrass vs. brome vs. fescue) over the course of the study. This will be discussed in greater detail later in the report.

The total frequency of forbs was more variable between 2004 and 2005, particularly in the Kiwi Power and Glacier Gold Compost plots (Figure 1A). And, as was observed with the grasses, dicot vegetation also exhibited significant changes in species distribution. In particular, the growth of white clover, cicer milkvetch, and yarrow increased in 2005.

A somewhat different trend was observed in the plant density data (Figure 1B). As with the frequency results, plant density responded positively to each amendment, with density values ranging from slightly less than 200 to over 300 plants/m^2 in the first year of the study. In contrast to the frequency results, the high fertility plots exhibited low plant densities. Furthermore, several of the lower fertility plots (particularly Kiwi Power and Biosol) exhibited some of the highest plant densities in 2003. Field investigations confirmed that the lower fertility plots are supporting large numbers of plants; albeit much smaller in stature as compared with neighboring high N plots. Thus, in terms of sheer numbers of plants per unit area, these plots exhibit relatively high plant densities. Plots receiving high N amendments supported grasses with larger growth characteristics (i.e. much taller, larger leaf area). These characteristics also suppressed growth of most forbs, most likely as a result of light competition. The net result was the growth of very large grasses on the high fertility plots but at lower densities. The high fertility, grass-dominated plots also exhibited very low numbers of unseeded species, while just the opposite was observed on the low N plots.

The total plant frequency and density results suggest that most of the plots fall into one of two generalized frequency/density profiles. Although this categorization grossly oversimplifies the very complex nature of plot assessment, it is worthwhile as a beginning step in understanding the interrelationships between vegetation and the properties of the growth media:

<u>Profile #1: Higher Plant Frequency/Lower Plant Density</u>

- frequency >80%, density <300 plants/m^2
- characterized by large vegetation and very dense stands
- higher fertility amendments
- lower diversity, dominated by grasses, low density of weeds
- e.g. Biosolids/Wood Ash, Eko Compost, Potlatch Log Yard Waste plus fertilizer

<u>Profile #2: Lower Plant Frequency/Higher Plant Density</u>:

- frequency <80%, density >300 plants/ m^2
- characterized by small thrifty vegetation and sparse stands
- lower fertility amendments
- higher species diversity, but also greater density of weeds
- e.g. Kiwi Power, Biosol, Glacier Gold Compost and Log Yard Waste

Species Distribution on Selected Plots

The topsoil-control exhibited a total plant frequency of 38% and total density of 266 plants/m^2 in 2003 (Figures 2A,B). Bluegrasses and bromes were the most commonly observed grass species. Yarrow and, to a lesser extent, mountain penstemon, comprised the majority of forbs. It is also clear from Figures 2A,B that unseeded vegetation (primarily hare's foot clover and black medic) comprised a significant proportion of the vegetation on this plot (also see Table 3). This observation illustrates the role that imported topsoil often plays in reclamation projects – serving as a seed bank for both desirable and undesirable vegetation. Total plant frequencies and densities increased significantly in this plot in 2004 and 2005, with greater numbers of white dutch clover and very large increases in unseeded vegetation.

The Eko Compost plot is illustrative of the seed mix's response to a high N amendment. As can be seen in the summary graphs (Figures 1A,B), similar plant frequency profiles were observed for the Biosolids, Potlatch LYW + Urea Fertlizer, and Eko Compost plots. Each of these plots was dominated by grass species with slender wheatgrass being most extensive, followed by the bromes, fescues, and bluegrass sp. (Figures 3A,B). The only forb encountered on a regular basis in 2003 was yarrow. Both plant frequency and density increased slightly between 2003 and 2005. The most noteworthy change in vegetation on these plots was a general decline in wheatgrass numbers with concurrent increases in the brome sp., fescues, and bluegrass sp. Another distinguishing feature of each of the high fertility plots was a very low incidence of unseeded vegetation.

A contrasting vegetative profile was observed on the lower fertility plots, such as Glacier Gold Compost (Figure 4). Although the total plant frequencies were low in comparison to the higher fertility amendments, large numbers of forbs were intermixed with the grasses. None of the grass species in the seed mix appeared to dominate. Yarrow was commonly encountered as was white dutch clover, mountain penstemon, and cicer milkvetch. In addition, most of the forbs increased in numbers between 2003 and 2005. It should also be noted that the low fertility plots tended to support the largest numbers of unseeded species, and these numbers increased over the course of the study. The greatest increase in unseeded vegetation was due to establishment of black medic, sweet clover, and spotted knapweed (Table 3).

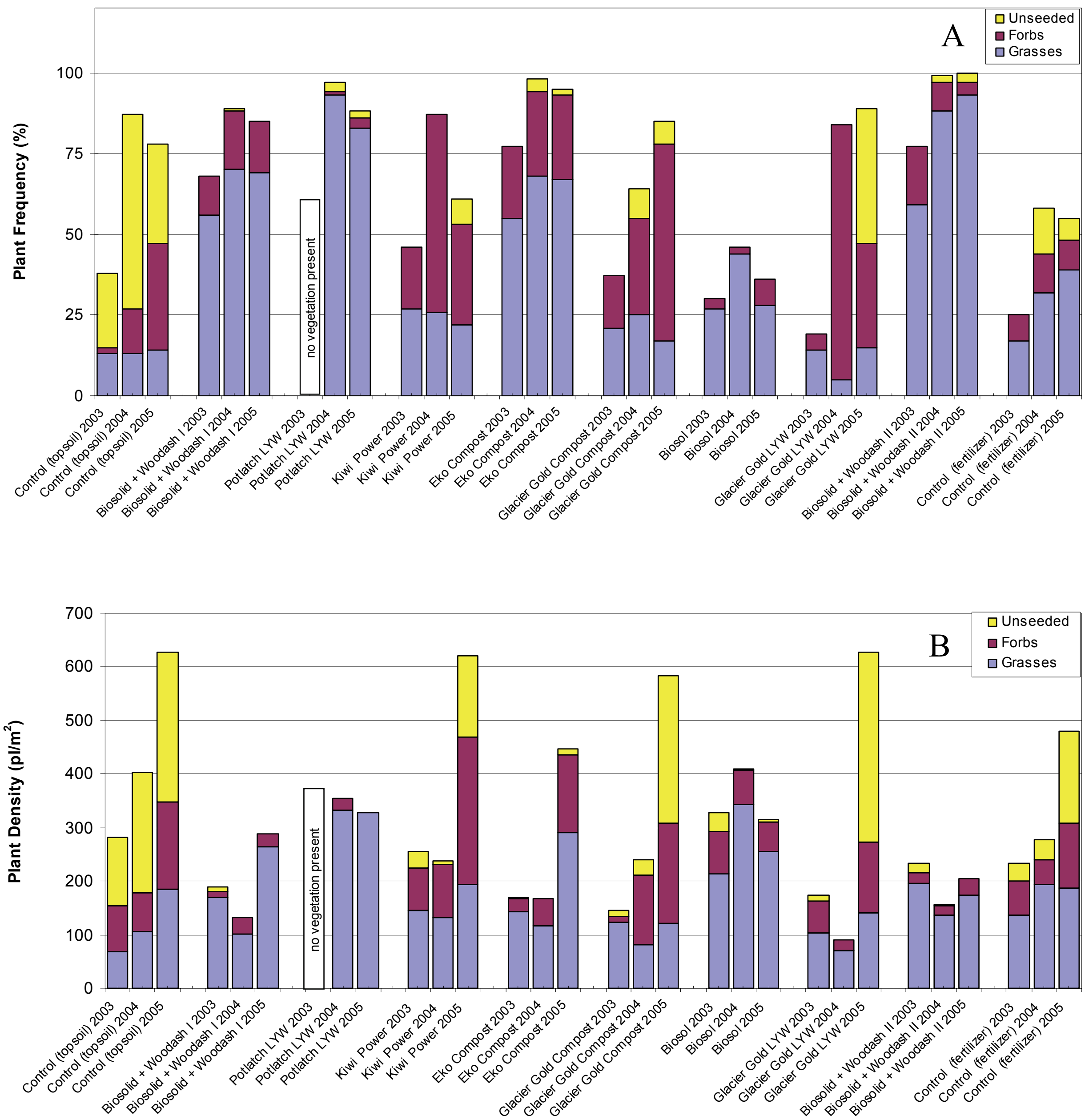

Figure 1. Comparison of (A) plant frequencies and (B) densities across all plots in 2003, 2004, and 2005.

The Kiwi Power and Biosol plots exhibited plant cover that was similar to the low fertility profile; that is, a relatively diverse mix of grasses and forbs as opposed to a grass-dominated plot. The Kiwi Power plot exhibited a plant frequency of 54% and plant density of 283 plants/m^2 in 2003 (Figure 5A,B). A mixture of grasses was evident with good establishment of wheatgrass, bromes, fescues, and bluegrass. Yarrow was dominant among the forbs, with much lower numbers of white dutch clover, cicer milkvetch, and mountain penstemon. Yarrow grew very well on the Kiwi plot with significant increases in 2004 and 2005. Unseeded species were present in low numbers in 2003 and 2004, but increased significantly in 2005, due to the establishment of black medic and sweet clover (Table 3).

The Biosol plot also exhibited a diverse vegetative profile, although the grasses were more frequently encountered over forbs (Figure 6). Each of the grass species was growing well with no single species dominating. It should be noted that a wheatstraw mulch was applied immediately following seeding, and this was tallied as litter during the plant frequency assessment (Figure 6A). Several forbs, including yarrow, penstemon, and white clover, were also established on this plot. It is noteworthy that the incidence of unseeded vegetation is relatively low on this plot, in comparison with other lower fertility plots (Table 3).

Changes in Species Distribution

While the total frequency and density of grasses in many plots did not change significantly between 2004 and 2005, significant changes in individual species did take place. For example, wheatgrass was clearly dominant in the Eko Compost plot in 2003. However, the 2004 and 2005 data show a more equal distribution between wheatgrass, bromes, and fescues (Figure 3A). This general trend of declining wheatgrass with concurrent increases in bromes and fescues was evident in most of the grass-dominated (higher fertility) plots. Yarrow was present in relatively high numbers in 2003, and increased steadily in most of the lower fertility plots throughout the study. In contrast, while white clover and cicer milkvetch were rarely encountered in 2003, these species increased significantly in 2004 and 2005. These changes were clearly evident in the Glacier Gold Compost plot (Figure 4). Both yarrow and white clover produce many profuse seed heads, which suggests the frequency and density of these species is likely to increase in the future.

Unseeded Vegetation

Significant increases in both the frequency and density of unseeded vegetation were observed over the course of the study (Figures 1A,B and Table 3). The vast majority of unseeded species can be classified as broadleaf weeds commonly encountered in the northwest (Whitson, 1999). It is important to note that several plots (i.e. Biosolids, Potlatch Log Yard Waste, Eko Compost, and Biosol) exhibited little to no growth of weed species. In contrast, the Control plots as well as Kiwi Power, Glacier Gold Compost, and Glacier Gold Log Yard Waste plots exhibited substantial increases (Table 3). For example, the weed density in the Topsoil Control increased from 128 to 280 plants/m^2 between 2003 and 2005 (Figure 2B). The increased weed densities observed in the other plots were also substantial with 25-50% of the total plant density occupied by unseeded species (Figure 1B).

Given the disproportionately high percentage of unseeded vegetation present in the topsoil control in Year 1, it is likely that many weed seeds were transported to the site in the topsoil amendment. However,

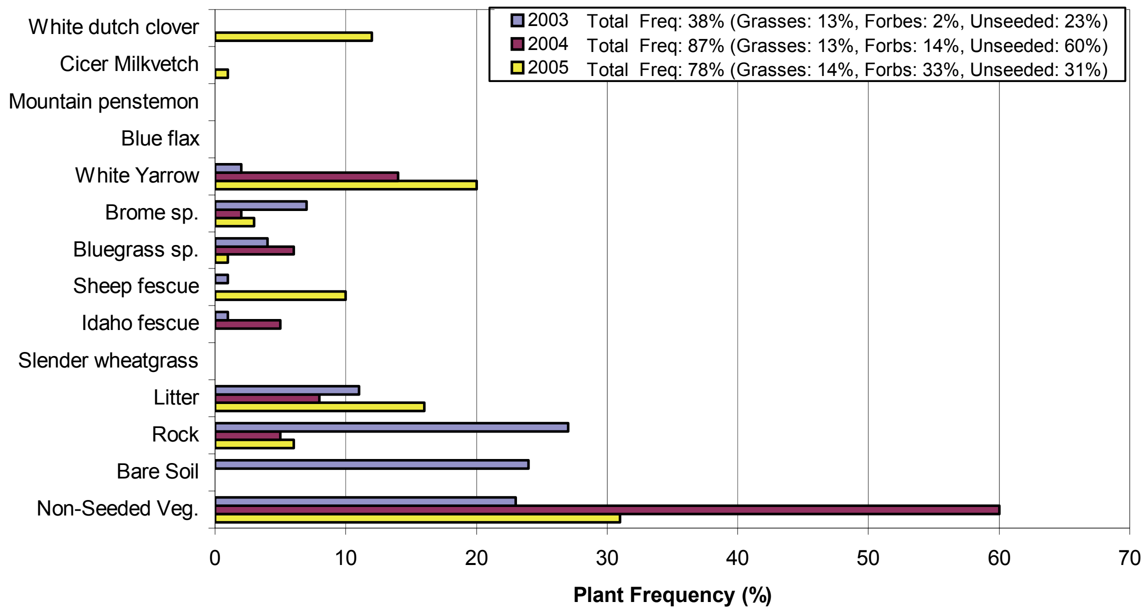

Figure 2A. Plant frequency on the Control-Topsoil plot in 2003, 2004, and 2005.

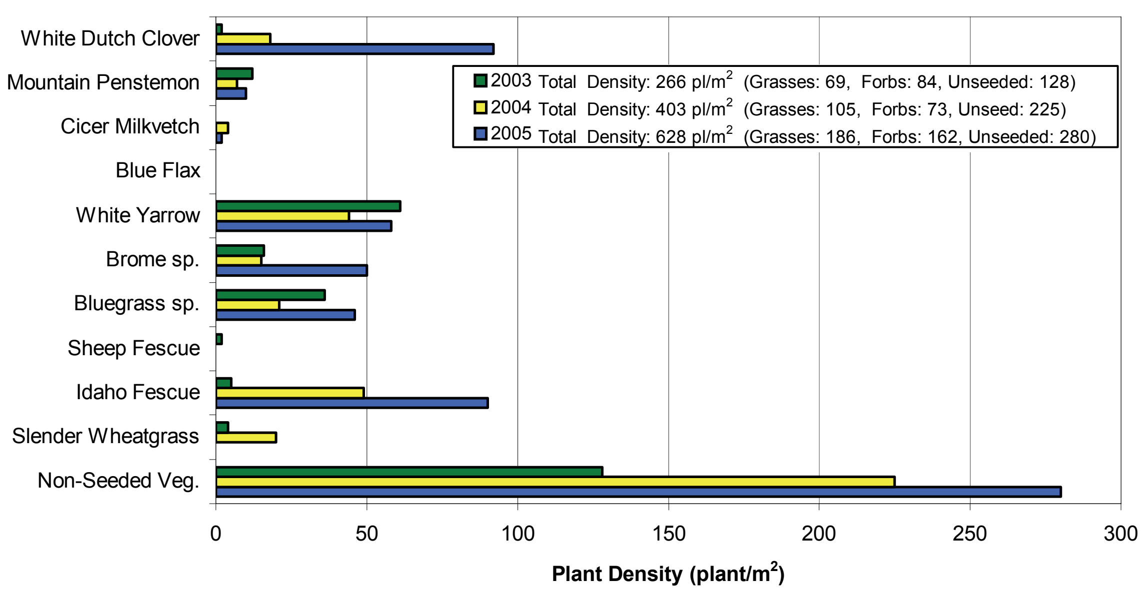

Figure 2B. Plant density on the Control-Topsoil plot in 2003, 2004, and 2005.

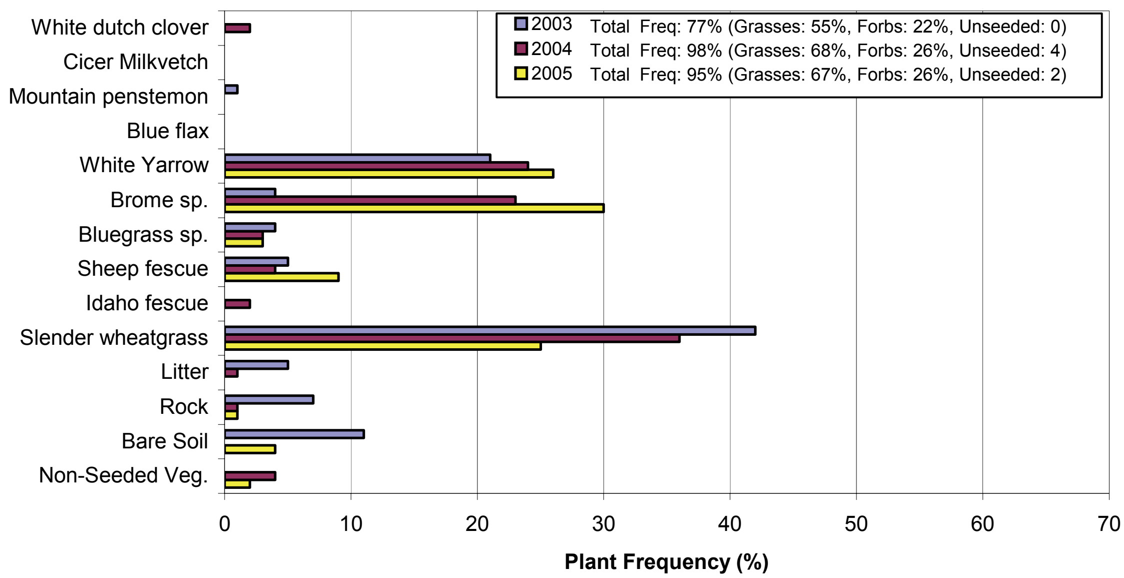

Figure 3A. Plant frequency on the Eko Compost plot in 2003, 2004, and 2005.

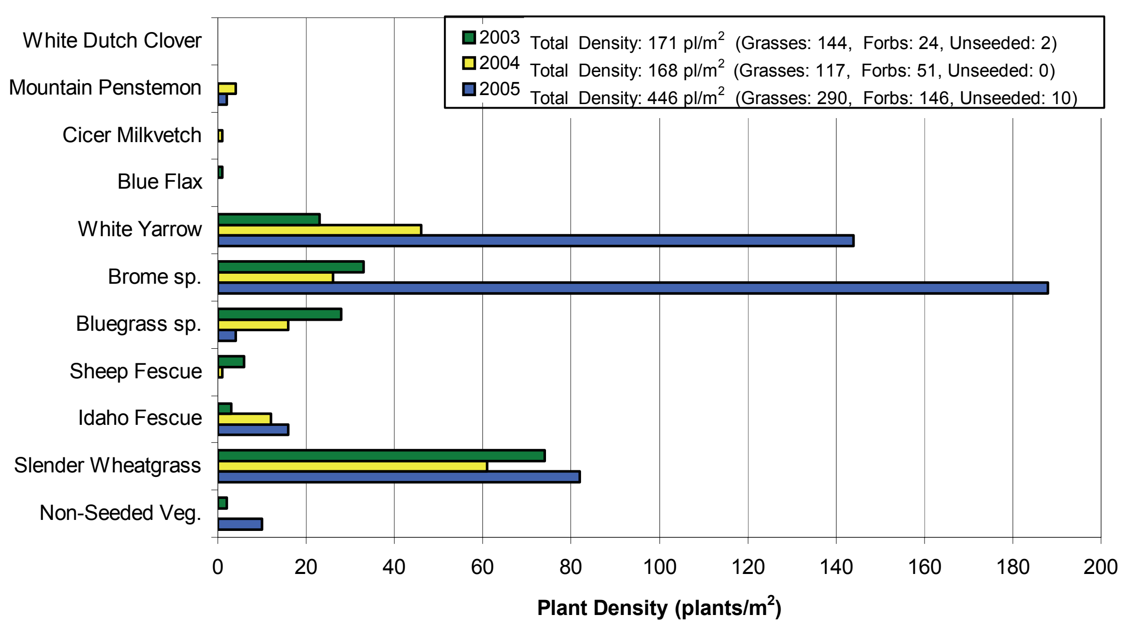

Figure 3B. Plant density on the Eko Compost plot in 2003, 2004, and 2005.

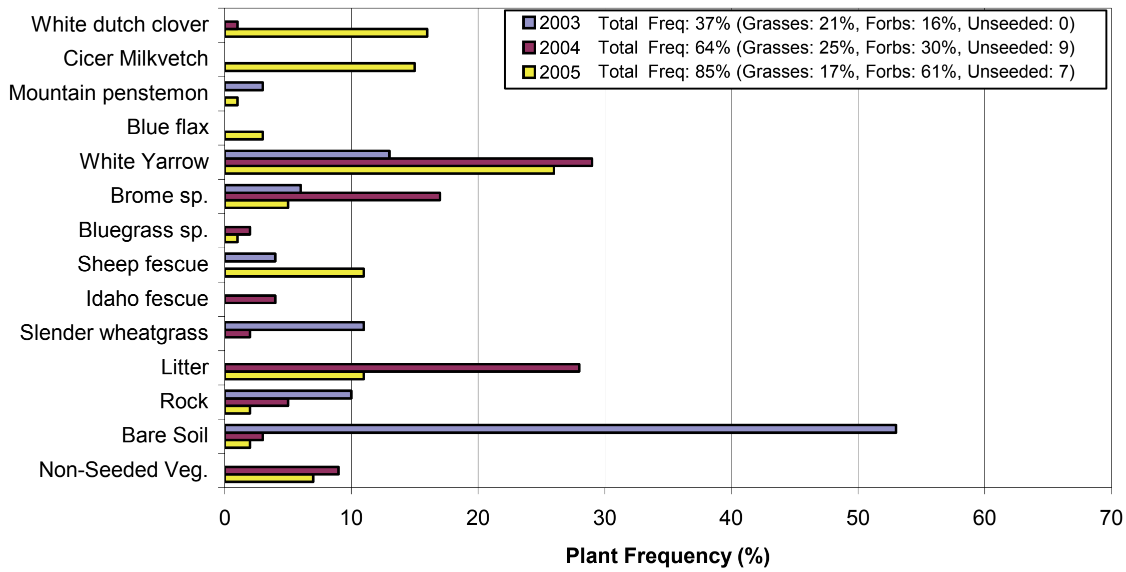

Figure 4A. Plant frequency on the Glacier Gold Compost plot in 2003, 2004, and 2005.

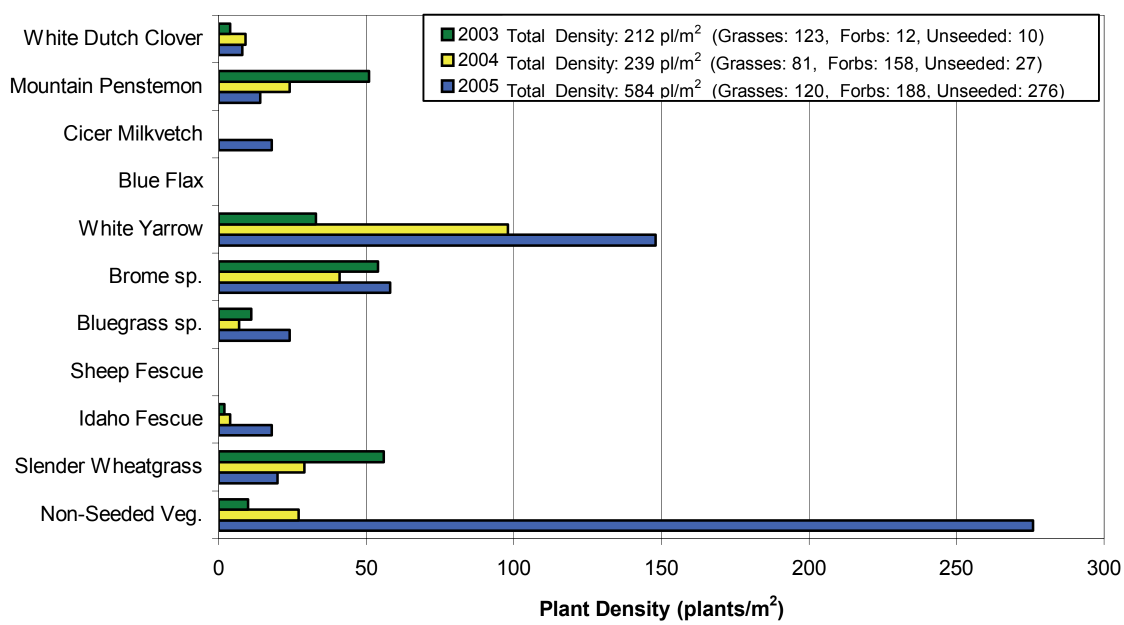

Figure 4B. Plant density on the Glacier Gold Compost plot 2003, 2004, and 2005.

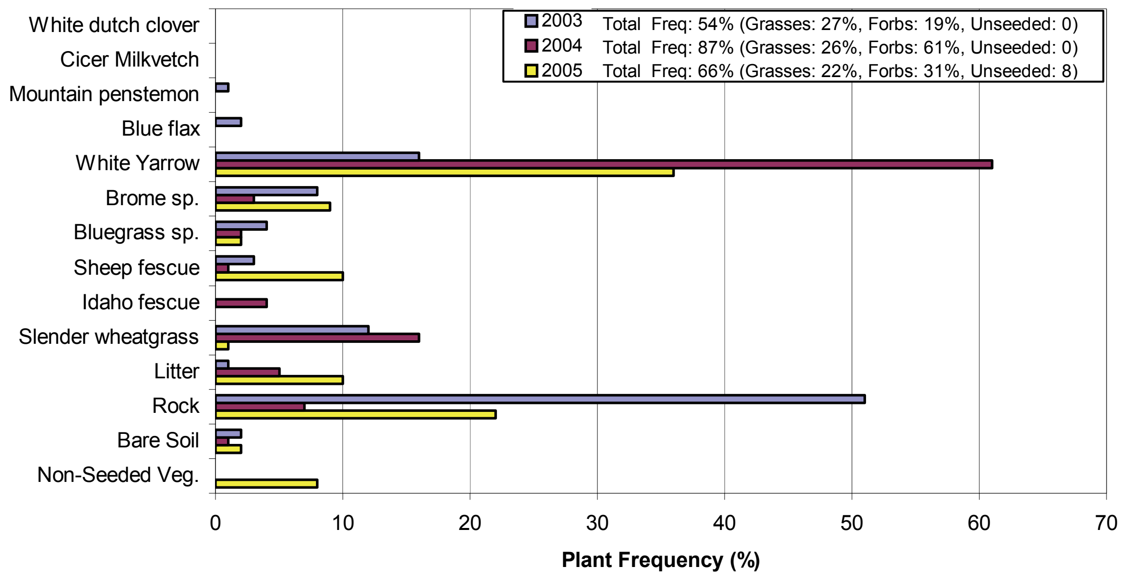

Figure 5A. Plant frequency on the Kiwi Power plot in 2003, 2004, and 2005.

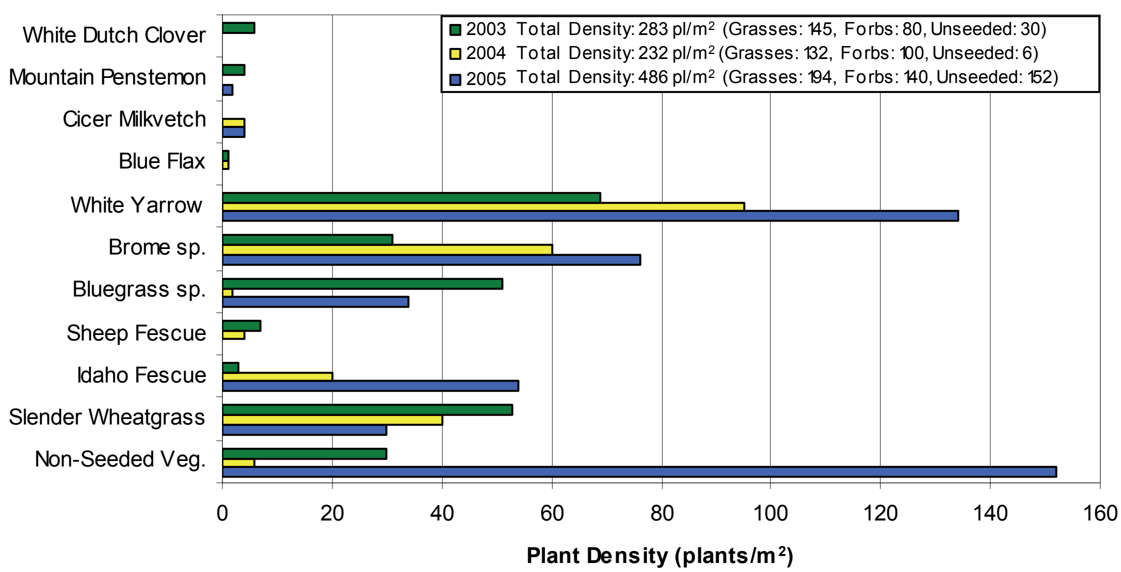

Figure 5B. Plant density on the Kiwi Power plot in 2003, 2004, and 2005.

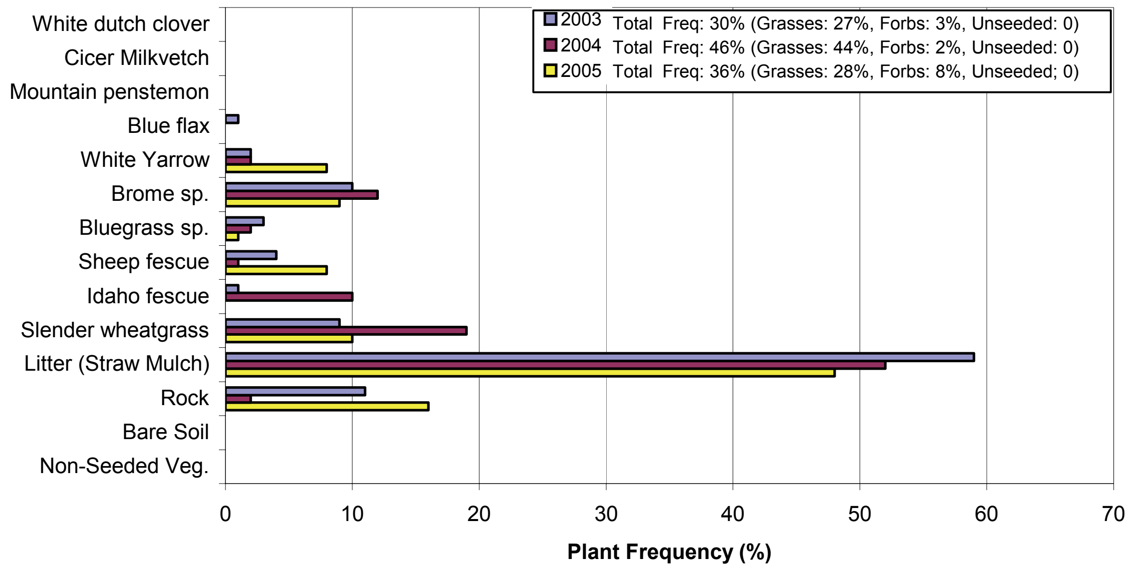

Figure 6A. Plant frequency on the Biosol plot in 2003, 2004, and 2005.

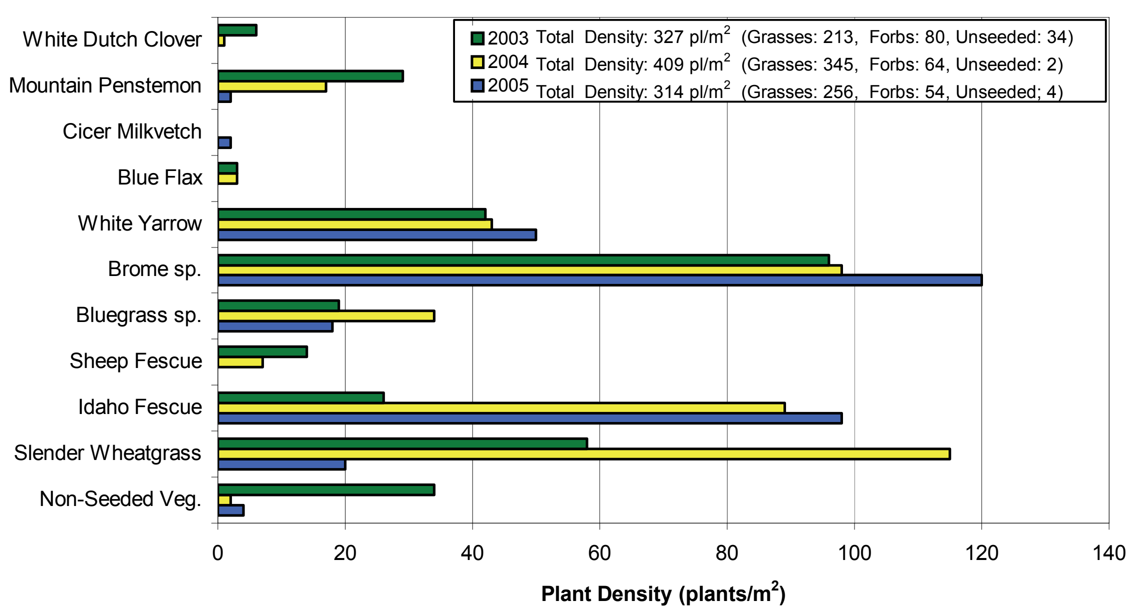

Figure 6B. Plant density on the Biosol plot in 2003, 2004, and 2005.

the 2004 and 2005 data indicate that weeds endemic to the surrounding landscape are beginning to invade the plots. In particular, sweet clover, black medic, and knapweed numbers increased significantly. Although knapweed was not perceived to be a major problem during the July 2004 plot assessments, a significant invasion was observed by the end of August. Project personnel decided to cut and remove the aboveground knapweed plants in an effort to reduce reseeding. Knapweed was judged to be a continuing problem in 2005 and, as such, the plots and borders were spot-treated with Reedem herbicide.

One additional note regarding unseeded vegetation – moss (of an unknown species) was observed to actively growing on every plot. The extent of moss coverage varied with the plot amendment and tended to be more extensive on heavily vegetated plots. These plots maintained relatively high surface moisture which might create favorable conditions for the moss. It is unclear as to the significance of moss growth in the overall revegetation picture.

Table 3. Density of unseeded vegetation on the demonstration plots.

Plot	Common Name	Scientific Name	Density (plants/m^2)		
			2003	2004	2005
A	Sedge	*Carex sp.*	53	0	0
	Black Clover (black medic)	*Medicago lupulina*	31	175	154
	Hare's Foot Clover	*Trifolium arvense*	9	29	102
	*encountered infrequently: toadflax, potentilla, knapweed, chickweed, mullin, oxeye daisy				
B	Sedge	*Carex sp.*	8	0	0
C	Cheatgrass	*Bromus tectorum.*	NA	2	0
D	Sedge	*Carex sp.*	22	0	0
	Black Clover	*Medicago lupulina*	5	2	54
	Sweet Clover	*Melilotus albus*	0	0	76
	*encountered infrequently: knapweed, lambsquarter				
E	Sedge	*Carex sp.*	2	0	0
	Sweet Clover	*Melilotus albus*	0	0	10
F	Sedge	*Carex sp.*	10	1	0
	Black Clover	*Medicago lupulina*	0	19	0
	Sweet Clover	*Melilotus albus*	0	0	276
G	Sedge	*Carex sp.*	27	4	0
	*encountered infrequently: horsetail, black clover, common tansy				
H	Sedge	*Carex sp.*	7	0	0
	Black Clover	*Medicago lupulina*	0	0	6
	Sweet Clover	*Melilotus albus*	0	0	348
	*encountered infrequently: red clover, prickly lettuce, maple				
I	Sedge	*Carex sp.*	14	0	0
	*encountered infrequently: moss, knapweed				
J	Sedge	*Carex sp.*	24	0	0
	Spotted Knapweed	*Centaurea maculosa*	0	0	42
	Sweet Clover	*Melilotus albus*	0	0	68
	*encountered infrequently: red clover, lotus clover, oxeye daisy, knapweed				

Nitrogen Impacts on Grass:Forb Distribution and Establishment of Unseeded Species

As discussed earlier, between 75 and 90% of the total vegetation is accounted for by grass species in the high N plots (e.g. Biosolids, Potlatch LYW, and Eko Compost). The correlation of high N availability and grass dominance is clearly shown by plotting the grass:forb ratio vs. available N for each plot (Figure 7). A grass-dominated plant community results in a high grass:forb ratio; for example, the high N amendments in the demonstration plots exhibited grass:forb ratios exceeding 5:1 In contrast, plots receiving lower N inputs (i.e. Controls, Kiwi Power, and Glacier Gold plots) exhibited a greater numbers of forbs mixed with the grasses, and exhibited ,uch lower ratios (<1:1). While many factors contribute to plant diversity and grass:forb distribution, N availability has been shown to be a key consideration in other reclamation/revegetation studies (Willems and van Nieuwstadt 1996; Baer et al. 2003; and Brown et al., 2005). Available N values less than 25 lb/ac are likely to restrict the growth of grasses and favor legumes, while available N >50 lb/ac favors grasses over legumes (Mahle, 2005).

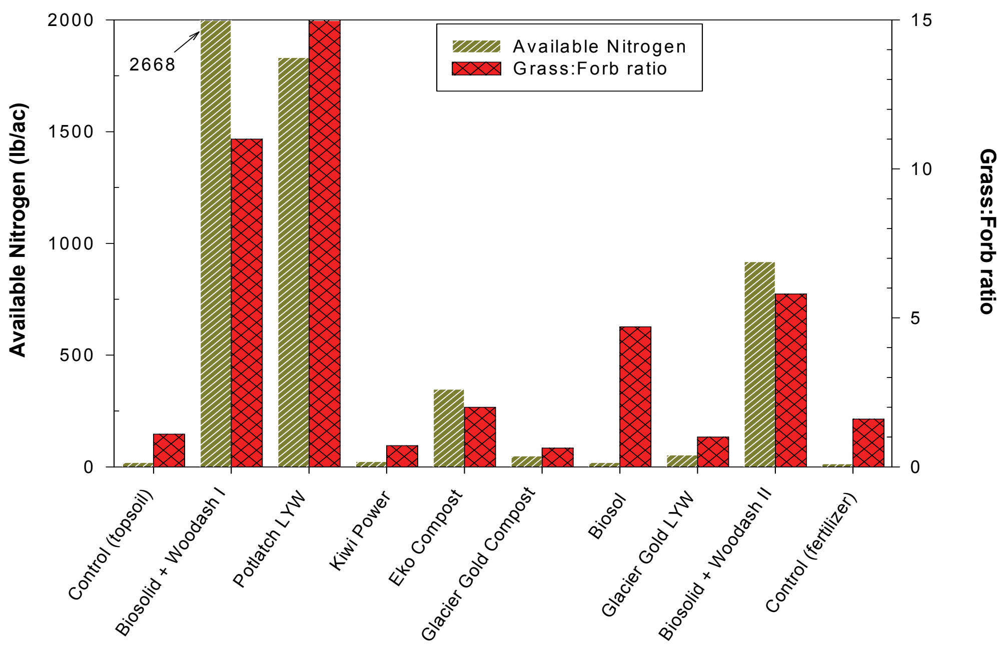

Figure 7. Available nitrogen vs. grass:forb ratio in the demonstration plots (2005 data).

The incidence of unseeded vegetation also varied significantly between the plots, with low weed densities consistently observed on plots with high available N (Figure 8). Conversely, much higher weed densities were observed on the low N plots. The mechanisms responsible for this trend are likely to be similar to those explaining N impacts on grass vs forb densities. High N availability promotes growth of tall, dense grass stands which out compete forbs and other broadleaf species for light (Grime 1973; Wilson and Tilman 1991; Rajaniemi 2002). Competition for nutrients replaces light competition as a determining factor in low N environments. Plant communities growing under these conditions are more likely to support small, sometimes stunted, stands with patchy coverage. These characteristics are favorable for establishment of invasive weed species.

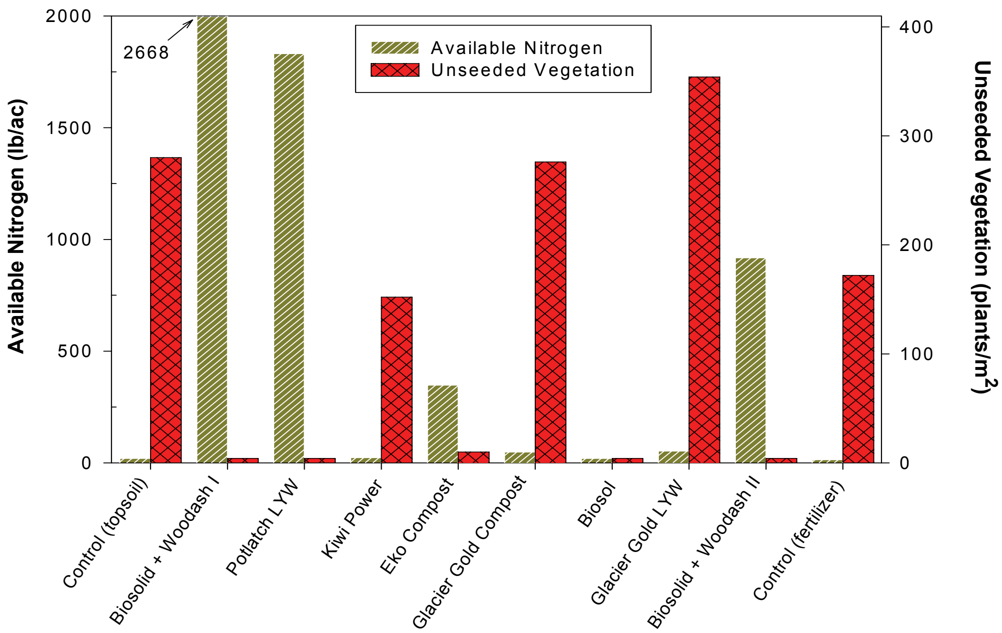

Figure 8. Available nitrogen vs. unseeded vegetation in the demonstration plots (2005 data).

General Ecology: High Fertility Plots

As discussed above, the high fertility plots were dominated by grasses. In most cases, slender wheatgrass (*Agropyron trachycaulum*) was the dominant species during the first two years of the study. Wheatgrass is a valuable component of reclamation seed mixes for revegetation and ersosion control due to its rapid development, ability to increase soil organic matter, and extensive root system. Wheatgrass responds extremely well to high nutrient availability and is know for its ability to sequester excess available nitrogen. This not only decreases the potential for runoff N, but also helps reduce weed problems. Decreasing the available nitrogen pool is also believed to assist plant succession by improving conditions for late seral species that are typically low N tolerant (Ogle et al. 2003). Slender wheatgrass is relatively short-lived (3-5 years) and, as the vegetation assessments show (Figure 3A), is beginning to diminish in favor of the bromes.

The brome species, mountain brome (*Bromus marginatus*) and meadow brome (*Bromus biebersteinii*) reach full productivity in 1-3 years and are both shade and nitrogen tolerant, making these grasses well suited to replace wheatgrass in the succession of high fertility plots. Mountain brome is short-lived and will be replaced by long-lived species over time including meadow brome, sheep fescue (*Festuca ovina*), and Idaho fescue (*Festuca idahoensis*). Declines in available N facilitate this succession, creating favorable conditions for the late seral grasses. The fescues take several years to develop but, once established, provide excellent cover, erosion control, and weed suppression (Ogle et al. 2003). Both species grow well in 10+ inch precipitation zones and can tolerate steep north-facing slopes. The three bluegrass species, Canada bluegrass (*Poa compressa*), big bluegrass (*P. ampla*), and canby bluegrass (*P.

canbyi) also possess growth characteristics that will fill niches in the demonstration plot plant communities. Canada bluegrass is slow to establish and tolerant of shade, making it another likely species to increase as wheatgrass declines. Growth occurs early in the spring providing good ground cover and, once established, is very persistent. Big bluegrass is very slow to establish, requiring as much as 4 to 8 years, but does well in mixed vegetation sites at 2000 to 6000 feet. Canby bluegrass is a long-lived species that is commonly crowded out when season-long moisture is available. In sites with dry summers, this species thrives on early season moisture and goes dormant quickly to resist drought.

General Ecology: Low Fertility Plots

The low fertility plots exhibited a more diverse mixture of forb and grass species. White yarrow (*Achillea millefolium*) was the most commonly encountered forb. This observation is not surprising as white yarrow is one of the most widely distributed forbs in the western United States, and is well adapted to disturbed and depleted sites (Ogle et al. 2003). White clover (*Trifolium repens var. Landino*) is a long-lived perennial legume that is well-suited to the shallow soils and slightly acidic to medium alkaline conditions of the demonstration plots. It is an effective erosion control plant on cool, moist, winter snow-covered mountain slopes. Cicer milkvetch (*Astragalus cicer*) is a long-lived, late maturing legume that is slow to establish due to very hard seed. It is well adapted to cold temperatures and will substitute for the other legumes when winterkill is a problem. Although Rocky Mountain penstemon (*Penstemon strictus*) and blue flax (*Linum lewisii*) were not encountered at the same frequency as the other forbs, both have been shown to do well when seeded in mixtures on disturbed seedbeds. Both penstemon and blue flax will tolerate some competition from grasses, but their production improves in more open communities (Ogle et al. 2003). Each of these forbs provides forage for grazing wildlife and the seeds are a good food source for birds.

General Ecology: Expected Future Trends

Slender wheatgrass should rapidly decline during Years 4 and 5, regardless of amendment. The bromes and Canada bluegrass are expected to replace the wheatgrass on plots with higher available N. Low available N will favor Idaho fescue, sheep fescue, and canby bluegrass. Big bluegrass is not likely to persist as this species is prone to leaf rust in higher moisture environments. Yarrow is likely to persist, with stands stabilizing in Years 4 and 5.

Fairly stable plant communities are expected on the solid-based amendments (biosolids, composts, and log yard wastes) by Year 5. These plots should maintain 4-6 inches of topsoil-like growth media with good water holding capacity and >3% organic matter. Under these conditions, nutrient levels should become more cyclical, thereby promoting a self-sustaining plant cover. A key question will be grass performance on plots with the highest available N. If conditions favor too much grass growth, lodging and disease problems could occur. This also could create an opportunity for establishment of weed species (Mark Stannard, NRCS, personal communication).

Organic matter content will be a critical feature on some of the low fertility plots. A degrading plant community is expected on plots with weak organic matter development (e.g. <1%). Under these conditions, grasses will comprise no more than 25% of the above ground biomass. Milkvetch and clover productivity should fluctuate year to year, and conditions will be favorable for broadleaf weed invasions (Mark Stannard, NRCS, personal communication).

CONCLUSIONS

Each amendment investigated in this study was successful in promoting a vegetative cover on the waste rock media. The amended plots exhibited a range in available nutrient levels and this had compelling impacts on plant frequency, plant density, species distribution, and the extent of unseeded vegetation. High N amendments promoted a grass-dominated, low forb profile, while the low N amendments promoted a more diverse grass-forb mixture. Furthermore, available N was inversely correlated with the extent of unseeded vegetation. Thus, it appears that high levels of available N provide grasses with a competitive advantage relative to forbs, resulting in high grass:forb ratios. This has the desirable outcome of controlling unseeded vegetation but at a cost of low species diversity. Furthermore, significant changes in plant communities were observed over the course of this three year study. Wheatgrass dominated the high N plots during Years 1 and 2, but is beginning to decline in favor of the bromes, fescues, and bluegrass. Yarrow was common to all low N plots in Year 1 and its numbers increased in each of the subsequent years. In Years 2 and 3, milvetch and white clover increased significantly, with lesser gains in pestemon and blue flax. In addition, a substantial increase in unseeded vegetation, primarily broadleaf weeds, was observed on the low N plots in Years 2 and 3.

As was expected, runoff nutrient levels were closely related to the available nutrient contents of the various amendments. However, even the plots with the highest available N and P concentrations exhibited significant reductions in runoff by Year 3. It is likely that the combined effects of leaching and plant uptake will result in relatively low nutrient levels in future runoff. Small amounts of soil erosion and sediment runoff were observed in Year 1 of the study, but these processes were negligible in subsequent years.

Overall, the results of this study show that each amendment investigated possesses characteristics and properties that are useful for reclamation of waste rock piles. Each material was capable of promoting a plant cover that was self-sustaining over the three year study period. The specific attributes of a given amendment resulted in unique characteristics for each plot. When considering the selection of a particular amendment for a remedial action, the extent of vegetative success (in terms of plant frequency and density) should be weighed against factors such as species diversity, nutrient runoff potential, and susceptibility to weed invasion. Finding the correct balance between these factors is the challenge of every restoration project.

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STATUS OF SPRUCE BEETLE AND MOUNTAIN PINE BEETLE OUTBREAKS IN HIGH ELEVATION FORESTS OF COLORADO

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ABSTRACT

Mountain pine beetle, *Dendroctonus ponderosae*, and spruce beetle, *Dendroctonus rufipennis,* are two of the most important mortality agents in Colorado forests. These beetles are native insects that often exist at background population levels, infesting primarily stressed or injured trees. However, under the right conditions, populations can build up to outbreak levels. In 2005 alone, mountain pine beetles and spruce beetles killed approximately 2.2 million trees on 550,000 acres in Colorado's high-elevation forests. While the majority of mortality occurs in lodgepole pine and spruce-fir forests, other tree species such as limber pine are also impacted. When such large-scale outbreaks occur the landscape is and will continue to be dramatically affected. Areas of once mature forests are opened-up for regeneration and the subsequent increase in forage production can be beneficial for a wide variety of wildlife. In total, the recent spruce beetle and mountain pine beetle outbreaks affect all of Colorado national forests. As a result, large-scale changes in high elevation forests will continue in the future.

PRESENTATION SUMMARY

Current bark beetle outbreaks in western North America extend from Alaska to the southern Rockies. Outbreaks are occurring from low elevation piñon-juniper ecosystems up to high elevation spruce-fir forests. In Colorado, mountain pine beetle and spruce beetle are responsible for tree mortality on thousands of acres in high elevation lodgepole pine and spruce forests. These native bark beetle species play important roles as agents of change by recycling mature forests and providing space for young trees to become established. The fact that tree mortality is occurring across large areas of very visible high elevation forests and where people recreate and live, is cause for concern by many of the public and land managers.

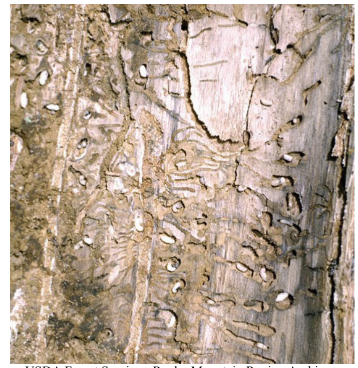
Beetle Biology

Mountain pine beetle, *Dendroctonus ponderosae*, and spruce beetle, *Dendroctonus rufipennis,* are members of the bark beetle family, Scolytidae. They feed under the bark in the phloem layer of the tree and their feeding coupled with the introduction of staining fungi lead to the death of the tree. Both beetles are similar in appearance, about ¼ inch long, and thousands can be present in a single tree (Figure 1). Mountain pine beetle adults attack lodgepole pine trees in the summer and lay eggs in the phloem layer that hatch into larvae. Larvae feed throughout the fall, enter a resting state during the winter and resume feeding in the spring.

Figure 1. Mountain Pine Beetle, *Dendroctonus ponderosae,* adult

They then pupate and turn into adults ready for the summer flight season (Figure 2). Spruce beetles have a similar life cycle with the exception that although a one-year life cycle occurs, they most commonly take two years instead of one to reach maturity.

Beetle infested trees show signs of infestation shortly after being attacked. A popcorn like pitch-tube is formed when the beetle tunnels into a tree and the tree attempts to push the beetle out. If the trees defenses in the form of resin production are strong enough the beetle's attack will not be successful. This is referred to as a pitch-out. If this happens beetles can often be seen entombed in the pitch-tube. However, if the beetle is successful then sawdust mixed with beetle excrement is present in bark crevices and at the base of the tree (Figure 3.). Even from a distance bark beetle caused tree mortality frequently is very conspicuous. For example, lodgepole pines killed by beetles in the previous year are visible by the bright yellow to orange colored needles.



USDA Forest Service - Rocky Mountain Region Archives, USDA Forest Service, www.forestryimages.org

Figure 2. Mountain pine beetle galleries, larva and pupa on underside of bark.

Kenneth E. Gibson, USDA Forest Service, www.forestryimages.org

Figure 3. Pitch-tubes on bark and frass around base of lodgepole pine tree.

Mountain Pine Beetle

Under low or endemic population levels mountain pine beetles usually infest individual trees that are stressed by such agents as disease or lightning. When forest conditions are conducive, beetle populations can build from endemic levels to outbreak or epidemic levels. On the landscape scale, these building populations first appear as small clumps of dead trees. During the following years, entire hillsides of lodgepole pine surrounding these clumps can be infested and die. In 2005 alone, aerial surveys mapped approximately 1 million lodgepole pine trees killed on 430,000 acres in Colorado (Figure 4). Aerial surveys are conducted each year over many of the forested areas in Colorado by the U.S. Forest Service and Colorado State Forest Service and provide an estimate of the numbers of acres and trees killed by insects and disease.

Mountain pine beetle populations in north-central Colorado lodgepole pine forests have dramatically increased in the last 10 years. In order for such outbreaks to occur, trees and stands must be favorable for beetle development. Since beetles feed on the phloem layer of trees, studies have demonstrated that in lodgepole pine they prefer larger diameter trees and trees with thicker phloem. Other studies have demonstrated that unthinned stands are more susceptible to attack than thinned stands. Additionally, drought and warmer temperatures have had a significant impact on mountain pine beetle development in lodgepole pine at elevations exceeding 9,500 feet where mountain pine beetles normally would have been expected to have only a modest impact. With the vast acres of mature lodgepole pine forests and climate conditions, Colorado forests were primed for the current widespread outbreaks we are experiencing. Areas that have been severely impacted by mountain pine beetles include Winter Park, Granby and the west side of Rocky Mountain National Park, Dillon, Keystone, Vail and many other forested areas that are popular recreation sites.

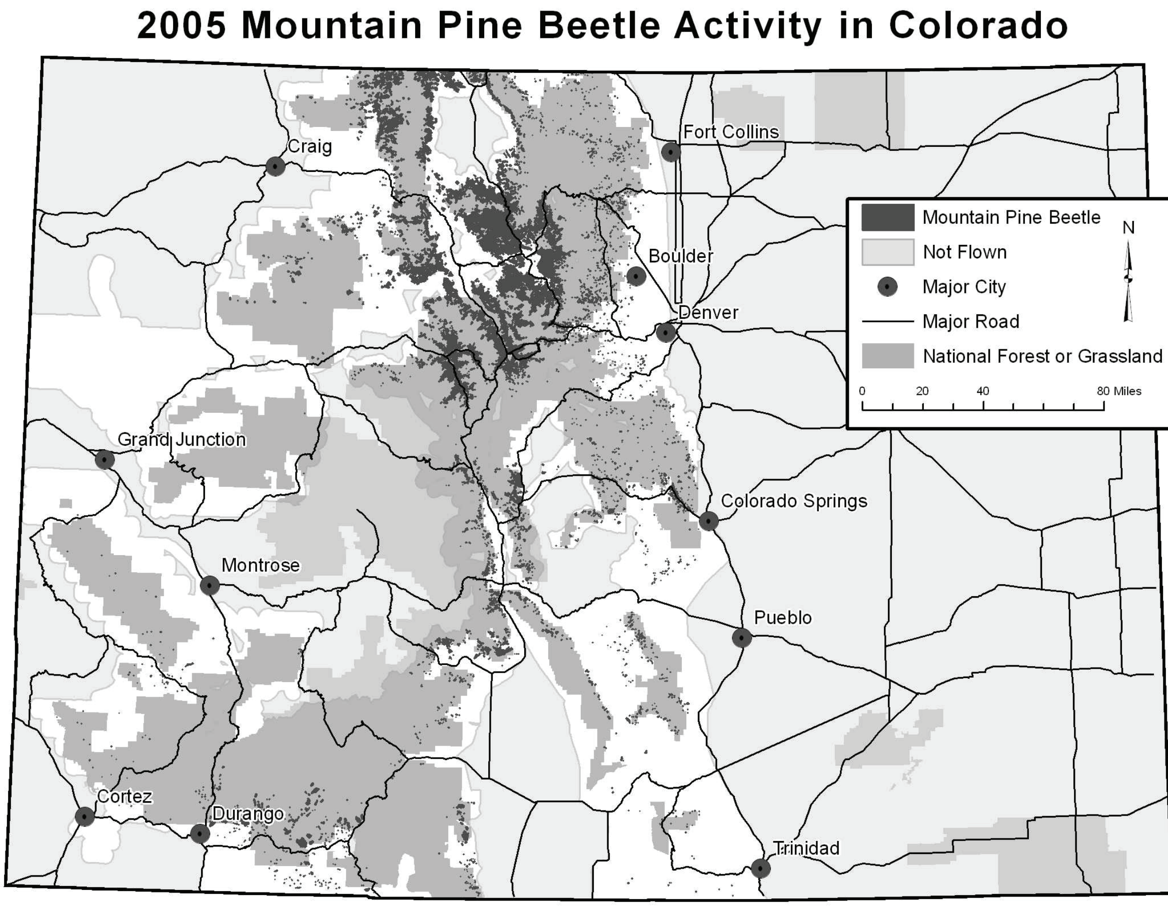


Figure 4. Mountain pine beetle activity detected by aerial surveys in 2005 in Colorado.

***Disclaimer**: Due to the nature of aerial surveys, the data on this map will only provide rough estimates of location, and the resulting trend information. These data should only be used as an indicator of insect and disease activity, and should be validated on the ground for actual location and casual agent. Shaded areas show locations where trees were killed. Intensity of damage is variable and not all trees in shaded areas are dead. The data represented on this map are available digitally from the USDA Forest Service, R2 FHP. The cooperators reserve the right to correct, update, modify or replace GIS products. Using this map for purposes other than those for which it was intended may yield inaccurate or misleading results.

Spruce Beetle

Spruce beetles also exist at low population levels and like mountain pine beetles can increase dramatically if the conditions are right. Unlike mountain pine beetles, during low population levels spruce beetles usually exist in freshly fallen trees. They prefer the underside of a tree, trees in creek bottoms, large trees and dense stands. Outbreaks are often initiated by large windthrow events, like the windthrow event that happened on the Medicine Bow-Routt National Forest, east of Steamboat Springs, Colorado, in 1997, that blew down a significant number of spruce trees on 13,000 acres. The abundance of big downed trees initiated one of the larger spruce beetle outbreaks in Colorado recorded history. From approximately 1999 to 2003, spruce beetles killed most of the standing spruce within the blowdown area before beetle populations began to decline. However, the effects of this blowdown triggered epidemic continue to be evident throughout the Park Range and the Sierra Madre (Routt and Jackson Counties). In addition, spruce beetle populations are increasing rapidly along the Medicine Bow Range in Colorado. This epidemic appears to have developed independently from the epidemic in the Park Range and Sierra Madre. Other large epidemics are found on the San Juan, Grand Mesa, Uncompahgre, and Gunnison

National Forests. In 2005, aerial survey estimated 1 million trees were killed on approximately 120,000 acres (Figure 5). Furthermore, unlike lodgepole pine needles that fade to bright orange and can remain on the tree for a couple of years, the needles on dying spruce trees fade to light green and shortly after fall to the ground, making them more difficult to spot from the air. This harder aerial signature coupled with the two-year life cycle makes it probable that the aerial surveys underestimate the extent and severity of spruce mortality significantly.

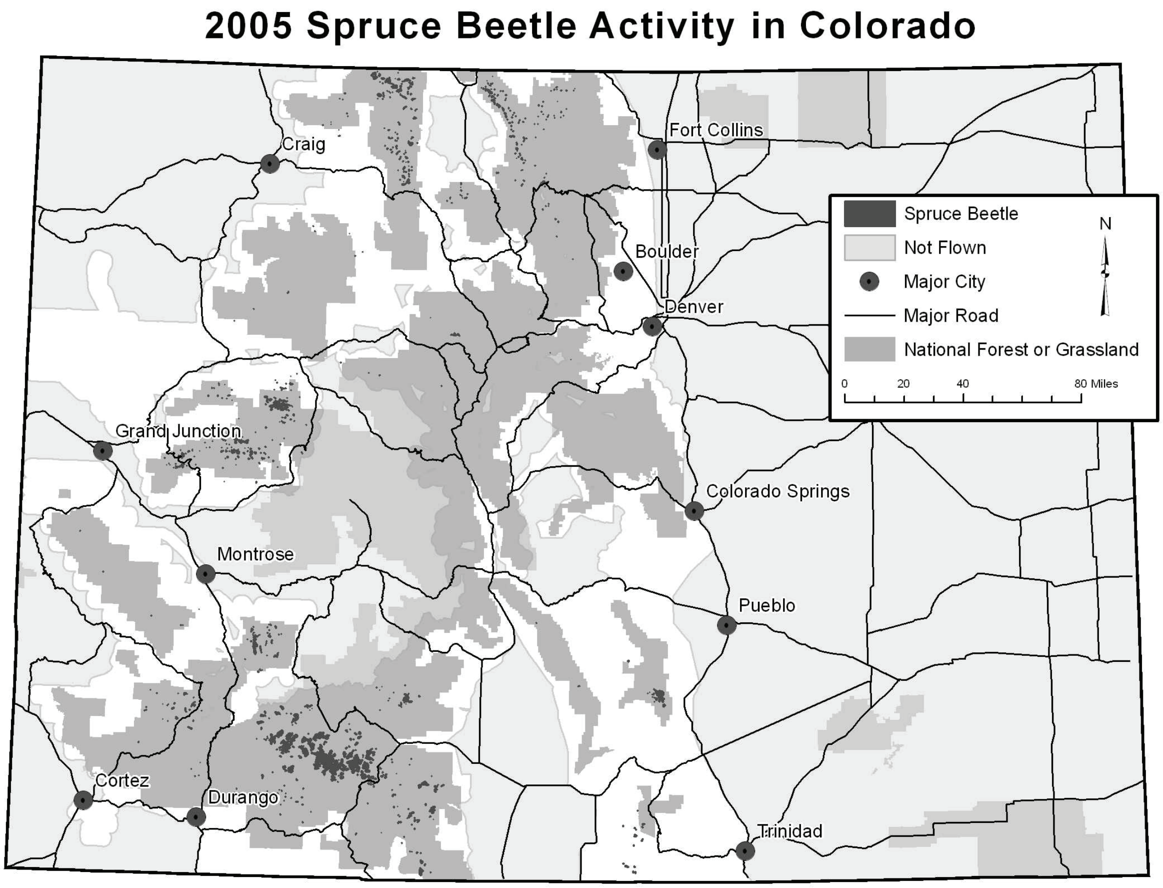


Figure 5. Spruce beetle activity detected by aerial surveys in 2005 in Colorado.

***Disclaimer**: Due to the nature of aerial surveys, the data on this map will only provide rough estimates of location, and the resulting trend information. These data should only be used as an indicator of insect and disease activity, and should be validated on the ground for actual location and casual agent. Shaded areas show locations where trees were killed. Intensity of damage is variable and not all trees in shaded areas are dead. The data represented on this map are available digitally from the USDA Forest Service, R2 FHP. The cooperators reserve the right to correct, update, modify or replace GIS products. Using this map for purposes other than those for which it was intended may yield inaccurate or misleading results.

Outbreak Implications

Each outbreak has its own "personality". Certain outbreaks slowly kill some of the larger trees in an area and having a thinning effect on the forest. Other outbreaks like the ones in lodgepole pine in Grand County, Colorado, are intense. The majority of trees within these outbreak areas are killed during the course of an outbreak, which may last six to eight years, or longer. Regardless of how an outbreak proceeds, spruce and mountain pine beetles in high elevation forests cause many changes to forests. A consequence of an outbreak of beetles is that they themselves become a food source for certain animals, including the three-toed woodpecker, often causing a local and temporary increase in this woodpecker population. Many different wildlife species, including elk and snowshoe hares, can benefit from the increase in forage after trees are killed. As well as, other shade tolerant tree species like aspen can become re-established and provide additional forage. Conversely, species like goshawks and pine martens rely on mature forests and winter cover for deer and elk may be diminished considerably. In addition to implications for wildlife, these outbreaks have raised public concern. Many land owners are faced with the disposal of beetle infested trees and are concerned with the possibility for high severity fires in these outbreak areas. Additionally, these outbreaks directly impact high-value areas like timber production sites, ski-areas and campgrounds that depend on the trees for revenue, wind-protection and aesthetics.

The question remains whether the large scale of these outbreaks is following a normal cycle or whether humans have modified the forest in these high elevations through fire suppression and forest applications, dating back to techniques employed in the early 1900's. Large outbreaks have been recorded in the past, but recent outbreaks are on a much larger scale. Apart from the historical records, current climate conditions and mature stands have set the stage for continued bark beetle outbreaks and changes to come. In addition, other important factors such as non-native insects and pathogens are currently affecting Colorado forests. The significant threat of invasive species coupled with beetle outbreaks could have devastating impacts on forest ecosystems. For example, another high elevation tree species, limber pine, is susceptible to a non-native pathogen, white pine blister rust that is spreading in Colorado's 5-needle pine species. In some areas of Colorado mountain pine beetles are also attacking limber pine, which coupled with white pine blister rust could have a dreadful effect on this tree species.

Lastly, it is encouraging to note that most lodgepole pine and spruce stands affected by mountain pine and spruce beetles will regenerate. For example, regenerating spruce forest can be seen in the Flat Tops area on the White River National Forest, where spruce beetles killed many of the spruce trees in the late 1940's and early 1950's. However, regeneration to a mature forest takes 100 to 300 years, especially in the case of spruce that live between 250 to 500 years before a major disturbance event like fire or a spruce beetle outbreak kills the majority of a stand. Still it is important to remember that despite the severity of these outbreaks mountain pine beetles and spruce beetles are native insects and one agent of change in constantly changing forest ecosystems.

BIOSOLID APPLICATIONS AT THE CLIMAX MINE: REVEGETATION AND SOIL RESULTS

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ABSTRACT

The Phelps Dodge Climax Molybdenum Company's Climax Molybdenum Mine (Climax) is a high elevation (3,200m) mine near Leadville, Colorado. Since 1918, Climax operations have affected approximately 3,000 acres of land. Reclamation of mine tailings is challenging due to extreme climatic conditions and the high coarse fragment and low organic matter content of overburden cover materials on site that have high acid production potential. The purpose of this study was to evaluate the enhancement of vegetative growth and soil development after a one-time application of lime, biosolids, and seed mixture to capped mine tailings in seven discrete units between 1997 and 2003. A field study was conducted in 2004 to evaluate vegetation and soil parameters, including nutrient and metal concentrations, across biosolids application units (BAUs). The findings of this study support previous research: biosolids are an effective means of establishing soil microbe and vegetation communities on capped mine tailings. This study suggests that over seven years and in extreme conditions, biosolid amendments reduced soil toxicity, neutralized acidity, improved wildlife habitat, and introduced constituents necessary to sustain vegetation communities on tailings capped with overburden material.

INTRODUCTION

As public demand for mineral resources continually increases, so does the extent of severe land disturbance due to mining operations. These impacts are particularly evident in mountainous regions of the western United States, where historically the rise and fall of communities has been tied to mineral exploration and mining. Reclamation of high elevation mining disturbances requires land managers to modify traditional reclamation techniques to account for the high acid production potential, phytotoxicity, limited topsoil availability, and extreme climatic conditions characteristic of these sites (Syndor and Redente 2002).

Regardless of elevation or latitude, successful revegetation of fields covered in mine tailings, which are often dry, acidic, and metal-contaminated, depends on amelioration of the soil environment (Noyd, Pfleger et al. 1995). Common mine waste amelioration strategies include neutralization and the addition of organic matter, including biosolids. Biosolids offer a cost-effective source of organics and nutrients necessary for successful reclamation, and tailings sites offer an economical and environmentally sound solution to the disposal of biosolids. Previous research shows that biosolids chelate and bind the pyrites of mine wastes and tie up heavy metals, and that with the application of biosolids there is a marked increase in organic matter, microbial activity, nutrient cycling, and decomposition of organics (Sopper 1993; Bengson 2000). High levels of biosolids have been shown to improve the physical characteristics of the mine wastes by decreasing bulk density, improving water holding capacity, and increasing infiltration (Jenness 2001). Biosolids also improve hydraulic conductivity and water saturation percentage and increase cation exchange capacity, which allows greater nutrient holding capacity, immobilizes heavy metals, and improves surface temperatures. Previous studies in alpine environments have demonstrated that biosolids are an effective means of incorporating organic matter, improving the fertility and physical and chemical properties of mine tailings, and facilitating reclamation. However the effects of a one-time application of biosolids and vegetation community structure and pedogenesis at 3,200m have been an unknown to land managers and researchers.

Climax requested that Habitat Management, Inc. (HMI) conduct soil and vegetation evaluations of the seven BAUs that were treated with 20-30 dry metric tons per acre of biosolids, limed at a rate of 30 tons per acre, and seeded between 1997 and 2003. Table 1 contains the Climax seed mix applied to the site at a rate of 25 pounds of bulk seed per acre as required by the site's reclamation permit. The purpose of this initiative was to evaluate the enhancement of vegetative growth and soil development after a one-time application of lime, biosolids, and seed mixture to capped mining tailings near Leadville, Colorado at 3,200m. This report contains vegetation and soil monitoring results collected in September, 2004. This data has been used to characterize the nature and vigor of plant communities established on the seven BAUs treated with biosolids.

Table 1. Permanent Reclamation Seed Mixture

Species	Common Name	Desired Species Composition (%)	PLS/Square Foot
Graminoids			
Agrostis gigantean	Red Top: Streamer	25.1%	106.8
Alopecurus arundinaceus	Creeping Foxtail	11.0%	47.0
Bromus anomalus	Nodding Brome: VNS	0.4%	1.5
Bromus marginatus	Mountain Brome: VNS	2.9%	12.4
Dactylis glomerata	Orchardgrass: Potomac	4.7%	19.8
Festuca arizonica	Arizona fescue: Redondo	3.6%	15.5
Festuca ovina btrachyphylla)	Hard Fescue: Brigade	6.4%	27.2
Festuca rubra	Creeping Fescue: VNS	3.4%	14.4
Phleum pretense	Timothy: Climax	10.0%	42.7
Poa compressa	Canada Bluegrass: Ruebens	16.5%	70.2
Poa pratensis	Kentucky Bluegrass: Troy	7.2%	30.6
Poa secunda	Big Bluegrass: Sherman	5.4%	23.0
Graminoid Subtotals (%, PLS/Acre, PLS Pounds/Acre, PLS/Square Foot)		96.5%	411.0
Forbs			
Achillea millifolium occidentalis	White Yarrow	2.0%	8.3
Astragalus cicer	Cicer Milkvetch: Monarch	0.2%	0.9
Trifolium repens	Clover: White Dutch	1.3%	5.7
Forb Totals (%, PLS/Acre, PLS Pounds/Acre, PLS/Square Foot)		3.5%	15.0
Combined Totals (%, PLS/Acre, PLS Pounds/Acre, PLS/Square Foot)		100.0%	426.0

METHODS

Habitat Management, Inc. staff evaluated species diversity, production, and vegetative cover across BAUs. Plant tissue and soil samples were collected to evaluate pH, organic matter content, and metal and plant nutrient concentrations. Qualitative inspections were also conducted on the reclaimed areas to identify plant species occurrence, erosion, and monitor other factors that may have affected vegetation success.

Site Description and Location

The Climax Mine, located near Leadville, Colorado at 3,200m, is one of the largest molybdenum mines in the world. The average growing season at Climax is six to eight weeks, the average annual temperature is 1°C, and the average snowfall is 6.8-m. The hectares (ha) of the seven BAUs monitored during 2004 are listed in Table 2.

Table 2. Climax Mine Biosolids Application Units

Year of Biosolids Application	ha
1997	2.25
1998	0.65
1999	10.19
2000	7.70
2001	2.72
2002	9.59
2003	16.50
Total	49.6

Vegetation and Soil Monitoring Methods

Habitat Management, Inc. collected the vegetation parameters of ground cover, species presence and frequency, and production from each BAU. The following sections detail the vegetation and soil sampling methods, respectively, that were used during the evaluation. The methods selected for the monitoring of post-mining revegetation success discussed in this section are accepted by the Colorado Division of Minerals and Geology (DMG) for a variety of hard rock, industrial mineral, and coal reclamation evaluations.

Sampling Point Selection and Transect Location

Across BAUs a total of 35 vegetation cover, shrub density, and production samples were taken. Five vegetation transects were located to best represent the average conditions of each BAU. Each 50-m transect began at a randomly selected coordinate intersection, and from this intersection the random transect direction was determined by the surveyor.

Statistical Analysis

The general linear model procedure for ANOVA was used to identify trends in the vegetation and soil parameters measured within each BAU. The year of biosolids application was regressed with measurements of ground cover, density and distribution of shrub and subshrubs, annual vegetation production, species diversity, soil pH, crude protein, and life form composition. The significance of the effects of biosolids amendments on vegetation and soil characteristics was evaluated at the 0.2 probability level.

Percent Vegetation Cover by Species

Line-transect point-intercept methods were used to collect ground cover data on each of the seven BAUs. A 10-point laser frame (with the points set apart 10 cm horizontally) was used to take ground cover measurements at 1-m intervals along a 50-m line-transect (Figure 1). Ground cover measurements recorded "first-hit" point-intercepts by vegetation species, litter, rock, or bare ground. Litter included all organic material that was either dead or did not represent the current year's growth. Rock fragments were recorded when particle size was equal to or greater than 1 cm^2.

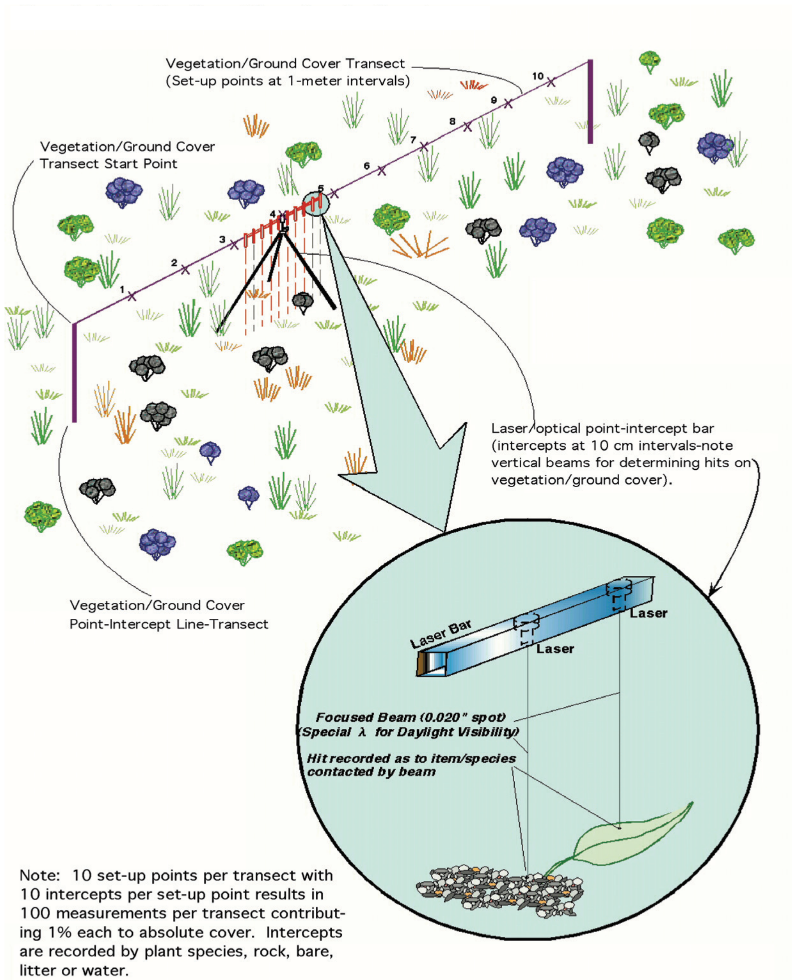

Figure 1. Vegetation/Ground Cover Sampling Procedures

Density and Distribution of Full Shrubs and Sub Shrubs

Each sample site was evaluated to determine the aerial distribution of trees and shrubs. Shrub density data was collected using 50-m square belt transects (1m x 50m) from the BAU. The line transects previously established for cover sampling were used for belt transects with data collected at a 1m distance from the right side of the 50-m line transect. Each full shrub and sub shrub encountered within each sampling strip was recorded by species.

Production Monitoring

Vegetation production data was analyzed to document the total annual growth of vegetation in each BAU. Above-ground biomass estimates were obtained by hand-clipping 5 randomly located ½-m rectangular quadrats from each plot. A production quadrat was located at the beginning of each of the 5 randomly located 50-m vegetation transects, with a 1-m offset to the right to avoid trampling the vegetation during cover and shrub density measurements. Plants within each quadrat were clipped to approximately 1 cm above the soil surface, bagged, and sent to the laboratory. There they were dried to constant weight in a forced-air oven at 140˚F for 24 hours, and weighed to obtain dry weights. Annual production was estimated to be the dry weights of the samples.

Vegetation Tissue Testing

Sub-samples from each of the 5 production samples collected from a particular BAU were composited and ground for the nutrient and metals analysis. The sub-sample of the oven-dried vegetation was analyzed for nutrients, protein, and the metal analytical parameters required in 40 CFR Part 503 including arsenic, cadmium, copper, lead, mercury, molybdenum, nickel, selenium, and zinc.

Soil Sampling

All soils contain trace elements or metals. Some metals are products of rock decomposition, some are atmospherically deposited, and some are anthropogenic in origin. Biosolids contain metals (often the so-called heavy metals) that enter sewage systems from homes, storm water, and industry. Several of these are "essential" plant nutrients, meaning plants must have them to thrive. Plants require copper, iron, phosphorus, manganese, molybdenum, nickel, and zinc. However some metals are toxic at high concentrations, and excessive quantities of metals including arsenic, cadmium, chromium, copper, lead, manganese, mercury, nickel, selenium, and zinc limit biotic productivity. Because of the potential environmental impacts of metal toxicity, agencies have developed stringent regulations for metal inputs to ensure environmental safety.

A purpose of this study was to make certain that the BAUs provide for an environmentally safe method of disposal for biosolids. The Colorado Department of Public Health and Environment (CDPHE) and U.S. Environmental Protection Agency (EPA) have developed requirements to reduce or eliminate the risk land applied biosolids pose to human health and the environment according to 40 CFR Part 503. Analytical parameters required in 40 CFR Part 503 for both organic and mineral soils include pH, arsenic, cadmium, copper, lead, mercury, molybdenum, nickel, selenium, zinc, nitrate, nitrite, total Kjeldahl N, total phosphorous, water soluble phosphorous, total carbon (Leco), and C to N ratio.

Fourteen soil samples from 35 cores were taken from the 7 BAUs. Soil sampling sites were located at the 0-m point along each of the 5 randomly located 50-m vegetation transect within each BAU. Soil cores were extracted from approximate depth increments of 0 to 6 inches and 6 to 12 inches. The surface depth increment was taken to the total applied depth of biosolids, and the second depth increment was taken from the biosolid/soil interface to a final depth of 12 inches to measure the lime and rock cover materials or subsoil. Each particular depth interval of the 5 sample points was composited and bagged for analysis. Acid/base accounting analysis was performed on the rock cover depth increment to evaluate both the effectiveness of liming and suitability for root penetration.

RESULTS AND DISCUSSION

Vegetation Cover

The more time that had passed since biosolids application, the greater the total vegetative cover (R^2=0.32, p-value≤0.18 – Figures 2 and 3). The exception was the 2002 BAU, which contained significantly less vegetation and more coarse woody litter compared to the other BAUs. Given the significant increase in dead and decaying vegetation observed in the 2002 BAU, and the relatively low above-ground biomass production observed in this area, this site most likely received organic amendment with greater concentrations of woody debris compared to the other sites. While no formal records of wood to sewage sludge ratios have been maintained at Climax, significant logging took place Peak 7 in Colorado's Tenmile Range in 2002. Consequently, higher concentrations of wood may have been mixed with the sewage sludge to produce compost and applied to the 2002 BAU. More woody debris in the organic amendment may have caused greater competition between plants and soil microbes for available nitrogen (N), thereby explaining the lower productivity and greater standing litter. This theory is supported by the higher carbon (C) to N ratios observed in the organic soil layer. In addition, 2002 was a severe drought year, which likely affected vegetation establishment and growth.

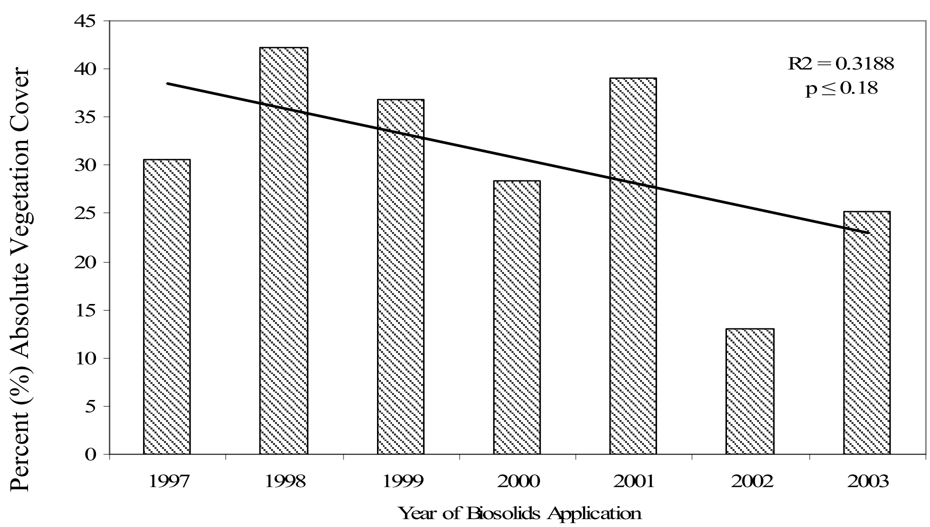

Figure 2. Vegetation Cover.

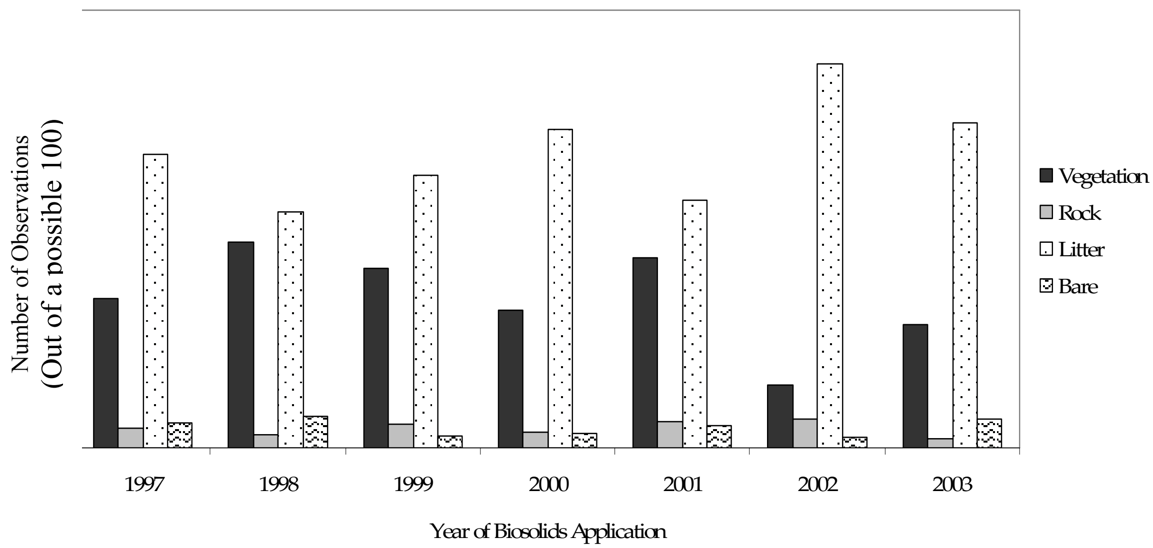

Figure 3. Total Cover across BAUs.

Full Shrub and Subshrub Density

Relative life form density observations were analyzed to identify any relationships between biosolids application and vegetation community structure. Greater densities of grasses were observed in more recent BAUs (R^2=0.47, p-value≤0.01). Greater densities of forbs (R^2=0.46, p-value≤0.01) and subshrubs (R^2=0.22, p-value≤0.01) were observed in older BAUs (Figure 4).

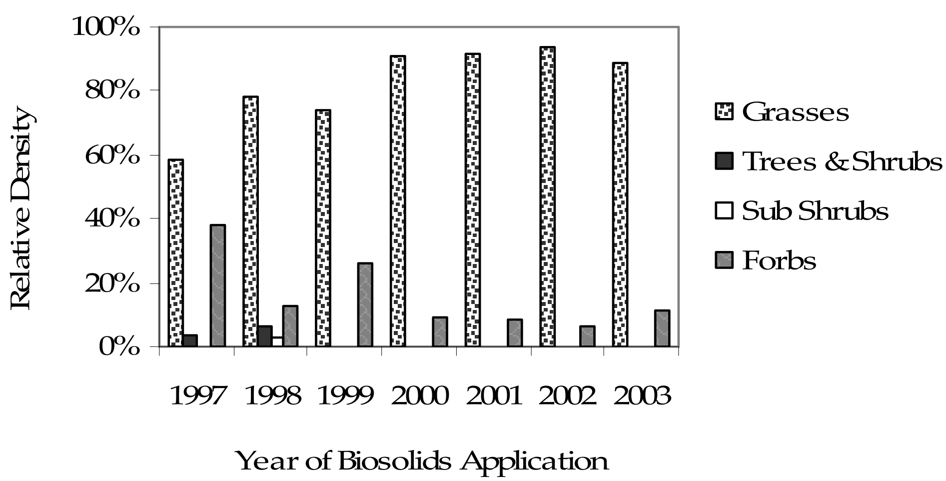

Figure 4. Relative Life Form Densities

Table 3. Life Form Frequency and Density

Total Life Form Density Observations					***Relative Cover (%)***				
Year	Grasses	Trees & Shrubs	Sub Shrubs	Forbs	Grasses	Trees & Shrubs	Sub Shrubs	Forbs	Total Observations
1997	17	1	0	11	59	3	0	38	29
1998	25	2	1	4	78	6	3	13	32
1999	20	0	0	7	74	0	0	26	27
2000	20	0	0	2	91	0	0	9	22
2001	21	0	0	2	91	0	0	9	23
2002	30	0	0	2	94	0	0	6	32
2003	16	0	0	2	89	0	0	11	18

Previous research demonstrates that the application of biosolids exacerbates to the early-successional species, such as grasses, and results in low establishment of woody and volunteer species (Halofsky and McCormick 2005). The DMG Hard Rock/Metal Mining Rules and Regulations require that lands "shall be revegetated in such a way as to establish a diverse, effective, and long-lasting vegetative cover that is capable of self-regeneration." Promoting species richness is not only environmentally responsible, but the DMG requires that a "diverse" plant community be established on reclaimed mine sites. While it may be preliminary to draw any conclusions based on observations made in 2004, future monitoring efforts should investigate the observed progression of shrub and tree establishment to evaluate reclamation success.

Vegetation Production

Vegetation production averaged 1,399.9 pounds per acre across BAUs (Figure 5). As mentioned previously, the 2002 BAU expressed significantly less above-ground biomass production compared to the other sites. Laboratory tests of nutrient concentrations in vegetation samples revealed that there were significantly lower concentrations of zinc, phosphorus, iron, potassium, magnesium, sodium, and sulfur compared to other BAU vegetation samples. However, the lower mean level of biomass production for the 2002 BAU was probably caused by the lack of precipitation and the higher rates of nitrate utilization by microbes to mineralize the additional carbon supplied by the wood residues (Figure 6).

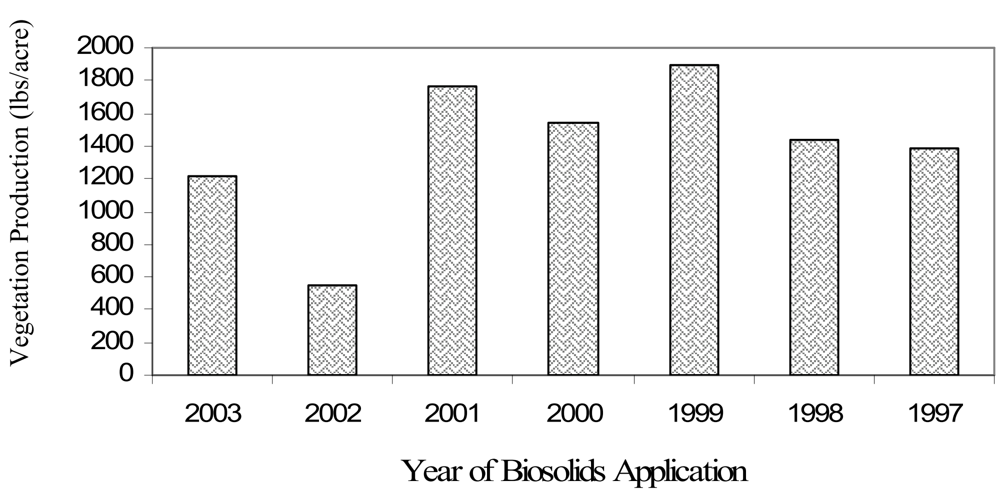

Figure 5. Vegetation Production

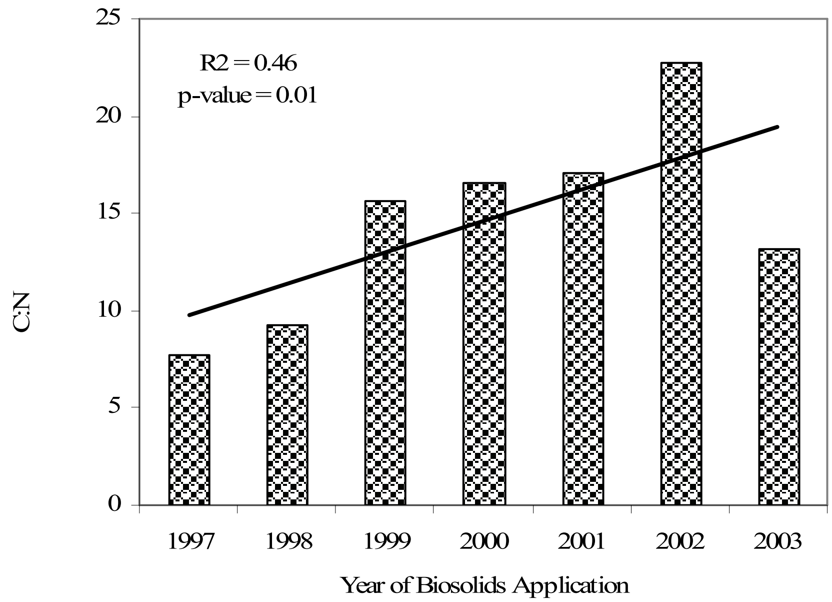

Figure 6. Soil Carbon to Nitrogen Ratios

Vegetation Tissue Composition

In accordance with 40 CFR Part 503, a sub-sample of oven-dried vegetation was tested for various metal parameters, including arsenic, cadmium, chromium, copper, lead, mercury, molybdenum, nickel, and selenium. No mercury was detected at the 0.05 mg/kg detection threshold; neither arsenic nor selenium was detected at the 0.5 mg/kg detection threshold; no chromium was detected at the 1 mg/kg detection threshold; and no lead was detected at the 5 mg/kg detection threshold. The concentrations of observed metals in vegetation composites are presented in Table 4; the results are average values of all plots for each test site.

Food, feed, or fiber crops may not be grown on active biosolids units unless the owner or operator of the surface disposal site can demonstrate to the permitting authority that through management practices public health, wildlife, and the environment are protected from any reasonably anticipated adverse effects of certain pollutants. Permits for grazing on areas treated with biosolids are granted by state authorities, and are approved on a case by case basis. Thus far, no permitting authority or regulatory agency has defined at what level metals in vegetation grown on biosolids amended soils pose a threat to human and wildlife health and the environment in Colorado. Reclaimed lands at Climax are available for use by resident and migratory wildlife populations with no anticipated agricultural land uses requiring grazing permits.

Vegetation samples were also analyzed to determine nutrient content. Specifically, samples were analyzed for N, phosphorus, potassium, magnesium, calcium, sulfur, sodium, iron, manganese, boron, copper, and zinc concentrations. Generally, the values of nutrient concentrations stabilized over time, indicating that nutrient cycling has been initiated on the older biosolids-amended sites.

Table 4. Metal Concentrations in Vegetation Samples

Metal	*Year of Sample*	*Level Found (mg/kg)*	*Metal*	*Year of Sample*	*Level Found (mg/kg)*
Molybdenum**	2004	41.8	Copper**	2004	21.2
	2003	18.7		2003	23.5
	2002	37.8		2002	8.6
	2001	18.8		2001	9.7
	2000	26		2000	8.6
	1999	12		1999	9.3
	1998	12.4		1998	9.9
	1997	15.4		1997	12.2
Nickel**	2004	n.d.	Cadmium*	2004	0.57
	2003	n.d.		2003	0.94
	2002	1.2		2002	n.d.
	2001	1		2001	1
	2000	n.d.		2000	0.73
	1999	1.1		1999	1.31
	1998	1.6		1998	0.72
	1997	1.4		1997	0.59

** 1.0 detection limit

* 0.5 detection limit

Nutrient content also provides a basis to judge vegetation quality. One measure of the quality of vegetation is Crude Protein percentage (Figure 7). Crude protein can be calculated by multiplying the N value by 6.25. Vegetation high in crude protein is typically low in fiber and thus has greatly increased digestibility by ruminant animals, such as cattle, deer, and elk.

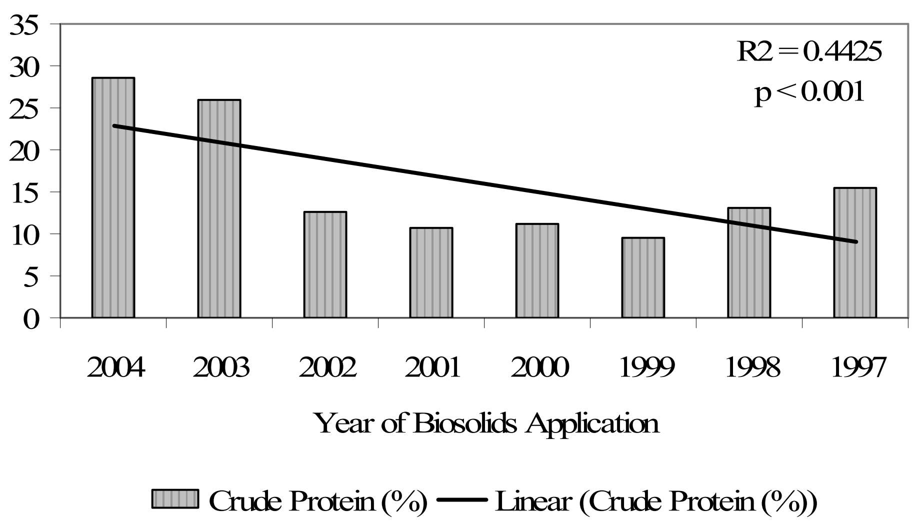

Figure 7. Crude Protein (%) across BAUs

From the vegetation data we know that the two dominate grass species in 2003 were *Phleum pretense*, and *Dactylis glomerata*. Typical crude protein book values from "NRC Nutrient Requirements of Beef Cattle, 2000" for these two species in a late vegetative state, are 14.0% and 8.4%, respectively. The average crude protein for the BAUs in 2003 was 25.9%. The overall average crude protein in the BAU's for all years is 15.9%. A 450kg cow nursing a calf, for the first 3-4 months postpartum has a dietary requirement for about 10% crude protein. Given the benign vegetation metal concentrations and crude protein observations, the BAUs appear to be a beneficial pasture of highly nutritious forage that exceeds the dietary requirements of ruminant animals.

Species Diversity and Composition

Comparisons demonstrated that there were greater numbers of species in older BAUs. The number of grass species represented from the various years of reclamation remained fairly consistent over time with a total of seven species present in the 1997 BAU (Figure 8) and 5 species present in the 2003 BAU (Figure 9). There was greater variety in the number of forb species, with a high of 6 species observed in the 1997 BAU and only 1 species observed in the 2002 BAU. These data suggest that grasses are quick to establish and persist in the reclamation. *Phleum pretense*, *Festuca ovina,* and *Dactylis glomerata* were the grass species most observed in the reclaimed BAUs. *Achillea millefolium* was the only forb encountered in vegetation transects, however additional forbs included in the rehabilitation seed mixture have been observed in the BAUs (Table 1).

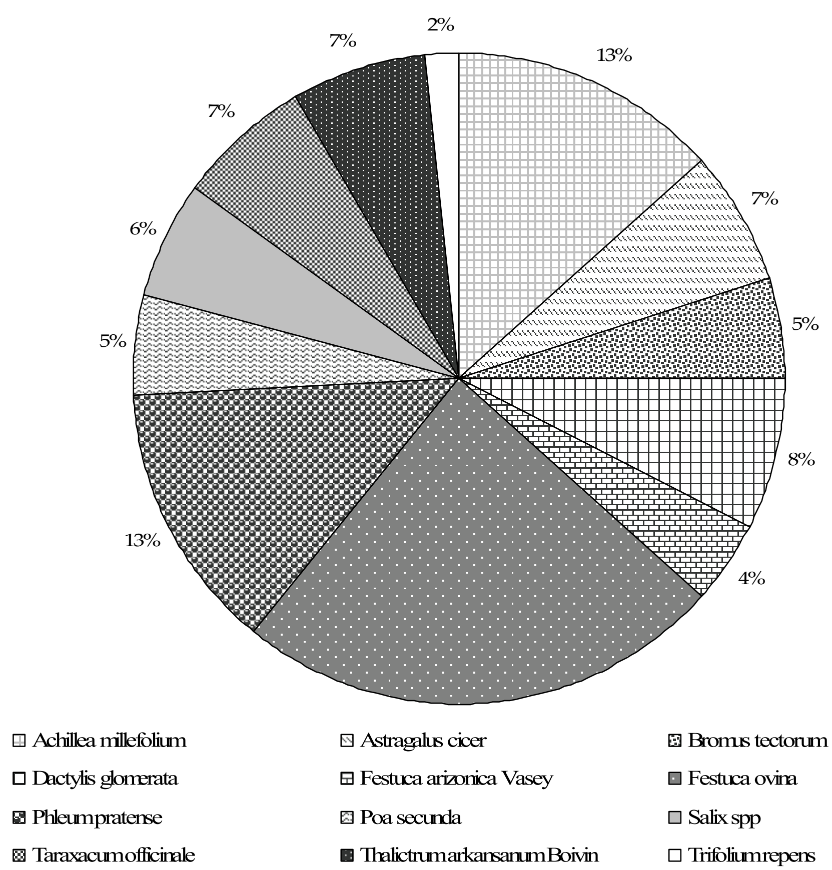

Figure 8. 1997 BAU Species Occurrence

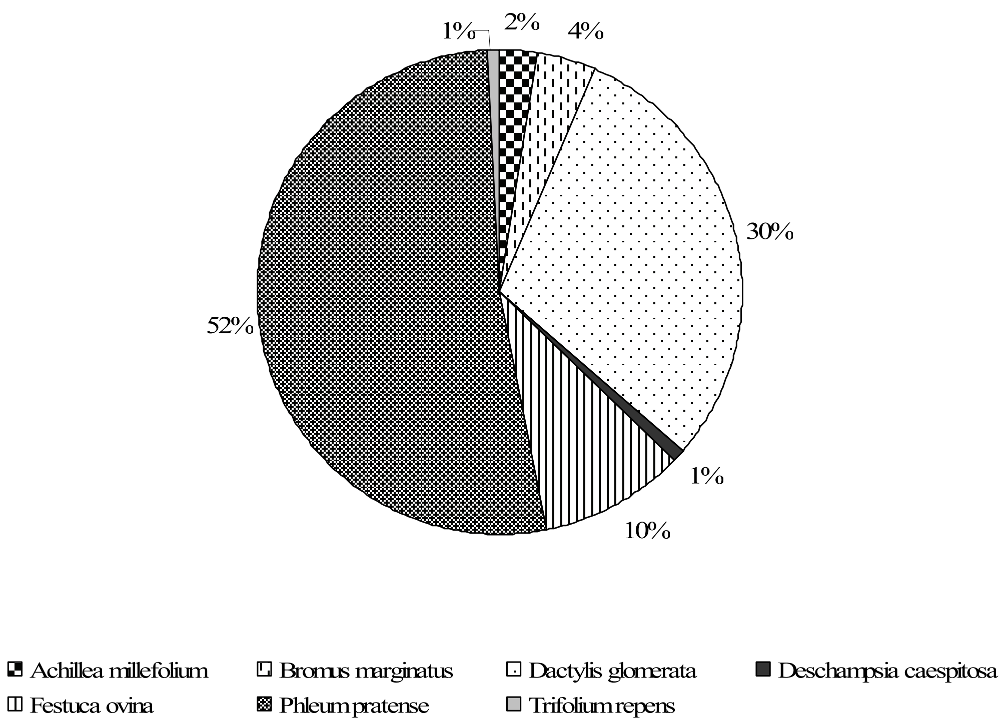

Figure 9. 2003 BAU Species Occurrence

Metal Concentrations in Organic Soils

The highest concentrations of the following metals were observed in the 1999, 2000, and 2001 BAUs: manganese, magnesium, zinc, silver, molybdenum, lead, iron, cadmium, and copper. The following metals remained fairly constant or there was no discernable pattern across BAUs: sulfur, sodium, selenium, barium, nickel, and chromium. Mercury and arsenic concentrations were greatest in the oldest BAUs, however these concentrations are far below limiting concentrations.

Metal and pH Concentrations in Mineral Soils

Minesoils consisted of run of mine waste rock that was placed as a cap over the tailings at a thickness of 12 to 18 inches thick. Ground agricultural and quick lime were applied on the surface of the cover prior to the application of composted biosolids at a rate of 24 tons per acres of agricultural lime and 6 tons of quick lime. The lime amendment was not incorporated into the cover. Minesoil samples reveal that lime and biosolids amendments were effective for treating large areas of highly impacted, base metal mining sites at Climax. A one-time application of lime and biosolids effectively increased pH to a suitable level for plant growth across BAUs from an initial pH of 4 to 5 (Figure 10).

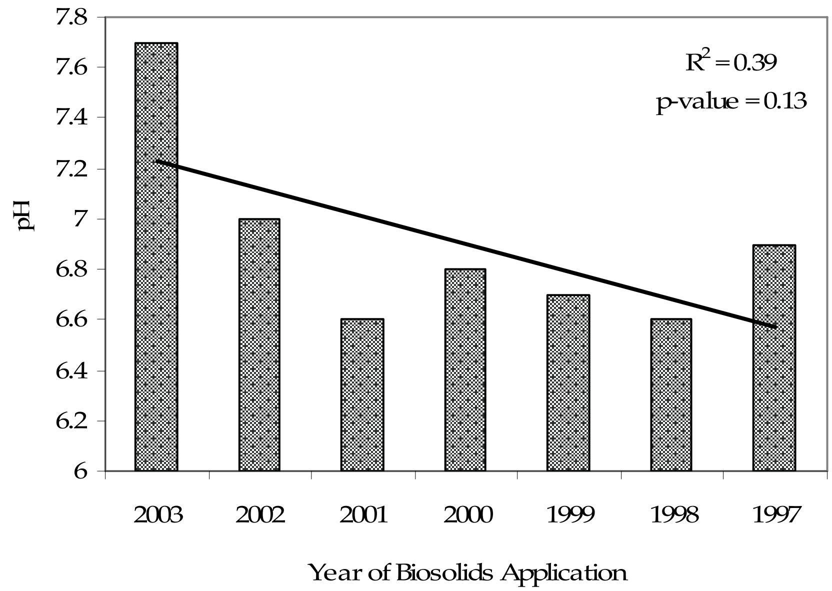

Figure 10. Mineral Soil pH Concentrations

Climax Biosolids Quality

Climax biosolid quality greatly exceeds the regulatory thresholds established by the EPA and CPDHE for the highest quality, pathogen free biosolids (

Table 5). Extensive data collected since 1920 show the use of biosolids, when in compliance with state and federal regulations, pose no known risks to human health or the environment (U.S. Environmental Protection Agency 1994). These results demonstrate that the use of biosolids in mine reclamation at Climax pose no risk to human or environmental health.

Table 5. Regulatory Requirements and Climax Biosolids Quality

Parameter	**Regulatory Limit**	**2004 Climax Biosolids Measurements**	**2003 Climax Biosolids Measurements**
	(mg/kg, dry weight basis)	(mg/kg, dry weight basis)	(mg/kg, dry weight basis)
Arsenic	41	1.5	1.02
Cadmium	39	0.6	n.d.
Chromium	1200 (EPA limit only)	6.8	7.6
Copper	1500	28.7	38.6
Lead	300	36.7	29.8
Mercury	17	n.d.	n.d.
Nickel	420	3.7	3.8
Selenium	100 (State limit)	n.d.	n.d.
Zinc	2800	115	39.3

Nutrient Concentrations and pH Levels in Organic Soil

Soil samples revealed that like vegetation nutrient concentrations, soil nutrient concentrations become more stable over time, as illustrated by the stabilizing of nutrients in older BAUs. These results suggest that nutrient cycling has been initiated in the biosolids-amended sites. Further, the biosolids appear to have had the anticipated effect of neutralizing the acidity of the tailings impoundments.

CONCLUSIONS

The findings of this study support conclusions drawn in previous investigations: biosolids are an effective means of establishing soil microbe and vegetation communities on mine tailings. The findings of this study suggest that over a 7 year span, Climax biosolids amendments reduced soil phytotoxicity, neutralized acidity, improved wildlife habitat, and introduced the necessary constituents to sustain vegetation communities. Further, the quality of the biosolids used at Climax far exceeds state and federal regulations.

Future monitoring efforts at Climax will enable mine staff and contractors to compare the effectiveness of using biosolids atop capped mine tailings with areas treated with biosolids ripped into the mineral soil layers. Anecdotal evidence suggests that ripping biosolids into the soil will reduce compaction and minimize lime stratification, which has formed an impenetrable layer to root systems, and further improve reclamation results.

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REVIEW OF THE EFFECTS OF CALCIUM CARBONATE-RICH SOILS ON PLANT ESTABLISHMENT DURING RESTORATION

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ABSTRACT

Soil factors can limit revegetation success. One such factor is the presence of soil calcium carbonate ($CaCO_3$). $CaCO_3$ is a salt commonly present in soils as a result of reactions between carbon dioxide, calcium cations, and water. Carbonate salts precipitate and can accumulate in soils as moisture sufficient to keep them in solution is lost by evaporation or transpiration. , $CaCO_3$ accumulations in lower soil profiles are called calcic horizons or a calcareous soil and are common in arid/semiarid regions. Exposure of subsurface accumulations of $CaCO_3$ can result from, agriculture, mining, and construction of landfills and/or evapotranspiration covers. The construction of soil profiles after disturbance with calcareous soil can be detrimental to the re-establishment of vegetation. Primary effects of high $CaCO_3$ are 1) inhibited absorption of macro/micronutrients, 2) increased potential for soil crusting, (thereby reducing water infiltration and root penetration) and 3) decrease in the abundance of soil microfauna that may decrease plants' ability to harvest water and nutrients. Information on what soil $CaCO_3$ level will decrease plant productivity is limited, but based on a review of current research, revegetation specialists should recommend limiting alkaline earth carbonates ($CaCO_3$ equivalent) to 10 percent or less.

INTRODUCTION

This work is a contribution to the understanding of how soil calcium carbonate may affect plant growth and was supported and funded by the U.S. Environmental Protection Agency, Region 8 (EPA). The EPA is tasked with overseeing the construction of evapotranspiration and capillary barrier soil covers at several locations at Rocky Mountain Arsenal Federal Facility Site (RMA). The functioning of the evapotranspiration covers requires an established plant community to transpire water. Several soil factors can limit revegetation success after disturbance and the presence of soil calcium carbonate ($CaCO_3$) in restored landscapes is one such factor. The soil to be used to construct the covers at RMA is from on-site sources that generally have subsurface layers of accumulated $CaCO_3$ at depth (most abundant accumulations are between 10- 14 feet) that could be placed at or near the surface. To evaluate potential thresholds of soil $CaCO_3$ abundance that might prove detrimental to vegetation growth, the EPA funded this work.

Other construction/mining projects may have similar concerns when restoring disturbed landscapes. In situations where soil is removed at depth, the most economical way to restore these landscapes post disturbance would be to invert the soil profile. This would potentially place soil that was at depth, with higher CACO3 accumulations, at or near the surface. This repositioning of the soil profile could adversely affect the establishment of plant communities.

The presence of $CaCO_3$ and its effects on plant growth is not confined to drastically disturbed lands. In some areas, $CaCO_3$ is present in the surface soil. For instance, in Northern Iowa 25 percent (2.6 million acres) of the topsoil contains $CaCO_3$. During glaciation, the parent material high in $CaCO_3$ was mixed into a till material. Later, when the ice receded, it left deposits of $CaCO_3$ at or near the surface. India also has many acres with topsoil rich in $CaCO_3$ and $CaCO_3$ accumulations at depth. In other situations, areas where concentrations of $CaCO_3$ are relatively close to the surface, topsoil can erode and expose lower soil profiles that were previously subsurface soil with $CaCO_3$ accumulations.

WHAT IS CALCIUM CARBONATE AND HOW DOES IT FORM?

$CaCO_3$ is a salt that can be formed by the reaction of carbon dioxide (an acid-forming oxide) and calcium oxide (a base-forming oxide). Carbon dioxide, produced by root (and soil microorganism) respiration in the presence of water, forms H_2CO_3 (carbonic acid) (Birkeland 1974). This formation tends to be most active in the upper soil where biological activity is highest. Calcium cations from weathering of primary minerals (parent material), or from windblown dust, or even entering the soil in rainwater, tend to stay dissociated in the upper soil where pH tends to be lower and water tends to be more abundant (Birkeland 1974; Jones and Suarez 1985; Monger and Gallegos 2000). As soil solutions pass to greater depth in the soil, increased pH and less abundant water drive the equilibrium toward precipitation of $CaCO_3$ (Birkeland 1974; Harden 1991; Pal et. al. 2000; Monger and Gallegos 2000). As this process continues in arid and semiarid regions, $CaCO_3$ accumulates in the lower soil profiles. Soil that contains $CaCO_3$ is called calcareous soil. Secondary accumulations of $CaCO_3$ in the subsoil, are referred to as calcic horizons. They may exist either as cemented layers, accretions, or concentrated horizons in lower soil profiles. These features are often colloquially but incorrectly termed caliche. Caliche (a geologic feature) forms on or very near the surface of soil in arid and semiarid regions), typically as a result of capillary rise and evaporation of $CaCO_3$-charged ground water.

Calcic soil horizons, by comparison, are a phenomenon of downward leaching. To a certain extent, the depth to calcic soil horizons depends on the amount of rainfall. Typically, as rainfall increases, so too does the depth to a calcic soil horizon. When annual rainfall exceeds 100 centimeters (~39 inches), calcic soil horizons disappear from the soil profile (Blatt et al. 1980).

Formation of $CaCO_3$ horizons or accumulations in soil of arid and semiarid regions in the world are common. In India, 54 percent of the total geographic area has soil that is calcareous (Pal et. al. 2000). In arid interior North America, such subsoil carbonate deposits are also widespread, including a large fraction of soils east of the Front Range on the high plains of Colorado.

As mentioned above, some CaCO3 rich soils are at or near the surface and are not confined to arid to semiarid regions. These surface soils high in CaCO3 are due to parent material erosion and/or the mixing of parent material during glaciation and then exposure after the glaciers receded.

The test used to identify the abundance of soil carbonates is called the Alkaline-Earth Carbonates from Acid Neutralization test and is detailed as Method 23c in U.S. Department of Agriculture (USDA) Agricultural Handbook No.60 (Richards 1954). Acid is added to the soil solution to neutralize the soil lime ($CaCO_3$). This test gives an equivalent percentage value, which is the amount of acid required to react with the lime. The test assesses the abundance of all alkaline earth carbonates present, and hence may, to varying degrees, overestimate the abundance of calcium carbonate.

CALCIUM CARBONATE EFFECTS ON NUTRIENT AVAILABILITY AND PLANT PRODUCTIVITY

One of the primary means by which $CaCO_3$ affects plant growth is by inhibiting the ability of plants to absorb nutrients from the soil. $CaCO_3$ affects plant uptake of both macronutrients (e.g., nitrogen and phosphorus) and micronutrients (e.g., zinc and boron).

The macronutrient most affected by the presence of $CaCO_3$ is phosphorus. Phosphorous is absorbed by plants in two forms, $H_2PO_4^-$, and HPO_4^{2-}. Of these, $H_2PO_4^-$ is most readily available to plants, whereas the other form (HPO_4^{2-}) is not readily absorbed by plants. In fact, McGeorge (1933) considered the monovalent ($H_2PO_4^-$) form, the only form of phosphorus that influenced plant growth and nutrition. In order for phosphorus to be absorbed by the root, the solution or film around the root must have a pH of 7.6, which is more difficult to attain in higher pH soil (McGeorge 1933). The abundance of the two phosphorous forms available to plants depends upon the overall pH of the soil (McGeorge 1933; Salisbury and Ross 1992). In low pH (acidic) soil, the most plant available form ($H_2PO_4^-$) is most abundant whereas the least plant available form (HPO_4^{2-}) is most abundant in high pH (alkaline) soil. The presence of the readily plant available phosphorous form ($H_2PO_4^-$) is greatly reduced in calcareous soil since they are generally higher in pH (alkaline) with pH ranges between 8.0 and 8.5 (McGeorge 1933; Sharma et al. 2001). In addition, phosphorus can react with $CaCO_3$ in the soil to form calcium carbonate phosphate (McGeorge 1933; Dominguez 2001), a phosphorous form unavailable to plants.

The uptake of micronutrients by plants is also affected by the presence of $CaCO_3$. The micronutrients whose absorption by plants is most affected by the presence of $CaCO_3$ are boron, zinc, iron, copper, and manganese (Brady and Weil 1994; Jones and Woltz 1996; Abdal et al. 2000). Reactions with $CaCO_3$, water, and carbon dioxide in soil can transform these micronutrients into forms unavailable for plants (Muramoto et al. 1991; Wang and Tzou 1995; Jones and Woltz 1996). One of the most common micronutrient deficiencies in plants is boron (Brady and Weil 1994). In calcareous soil, boron is fixed or bound by soil colloids (Brady and Weil 1994; Rahmatullah et al. 1998). For example, a study on sunflowers found that, as soil concentrations of $CaCO_3$ increased, the dry weight of sunflower shoots decreased, which directly correlated with decreasing concentrations of boron in the plant tissue (Rahmatullah et al. 1998). Iron is also made relatively unavailable to plants when $CaCO_3$ is present in the soil system. Plant iron deficiencies in leaf tissue can cause some plants to develop chlorosis (diminished abundance of chlorophyll in leaf tissue, recognized by yellowing of leaves) resulting in significant reductions in plant vigor (Kiloen and Miller 1992).

Concentrated $CaCO_3$ in soil also increases the potential for soil crusting, thereby reducing water infiltration and inhibiting root penetration (West et al. 1988; Abdal et al. 2000; Dominguez et al. 2001; Sharma et al. 2001). In other words, physical changes of the soil caused by higher concentrations of $CaCO_3$ can cause reductions in plant productivity.

In addition to inhibiting plant growth, increasing $CaCO_3$ concentrations in soil have also been linked to decreases in soil microfauna populations and mycorrhizal associations (Sharma et al. 200; Allen 1996). The affected microfauna include fungi, bacteria and actinomycetes, and azotobacter (Sharma et al. 2001). Microfauna are critical to the conversion of soil nitrogen into forms available to plants. Mycorrhizal associations (a symbiotic relationship between the root and fungi) can be critical for plants to increase uptake and harvesting of nutrients, especially phosphorus, and water.

As outlined above, the presence of $CaCO_3$ can decrease the availability of plant nutrients, reduce plant productivity and can produce physical changes to soil limiting water infiltration and root penetration. All these affects, can be detrimental to plant community establishment therefore the primary interest is in identifying a $CaCO_3$ percentage in restored soil profiles that would not be detrimental. While this is the goal, it is difficult to identify a soil $CaCO_3$ percentage that is not detrimental to plant growth without empirical data. An attempt has been made to glean information from the literature to target a $CaCO_3$ level that would not limit revegetation.

PREVIOUS STUDIES AND RECOMMENDATIONS FOR SOIL CaCO3 EQUIVALENCY PERCENTAGE USED AS A SOIL SUITABILITY CRITERION IN RESTORATION.

What is the maximum percentage of $CaCO_3$ in a soil that is acceptable for use in the restored plant growth media? Many authors have noted general decreases in plant productivity with increases in $CaCO_3$ percent (Lajtha and Schlesinger 1988; Westermann 1992; Pierce et al. 1999; Pal et al. 2000; Dollhopf and Mehlenbacher 2002). However, the specific level at which $CaCO_3$ begins to inhibit plant growth has not been well defined. A Kuwaiti vegetable production study showed very low crop biomass (89 to 400 lbs. per acre) when grown in soil with $CaCO_3$ ranging from 20 percent to 35 percent (Abdal et al. 2000). Hence $CaCO_3$ concentrations from 20 to 40 percent may significantly inhibit revegetation.

Soil with $CaCO_3$ content between 15 and 40 percent has been viewed as a fair soil for soil reconstruction of drastically disturbed lands by the U.S. Forest Service. The Kuwaiti study suggests a possible upper limit of 15 to 20 percent CaCO3 for adequate revegetation.

However, several other plant productivity studies showed that increasing amounts of soil $CaCO_3$ can significantly decrease plant productivity even at relatively low soil $CaCO_3$ percentages. An Australian growth effects study showed a reduction in root and shoot growth, and nodule production by lupine (*Lupinus angustifolius*) with $CaCO_3$ levels as low as 1.5 percent (Jessop et al. 1990). This study mixed soil with high and low $CaCO_3$ percentages, so there was a range of soil $CaCO_3$ percentages from 0 to 6.6 percent used to grow the flowers. A linear response was evident, with decreasing relative dry weight of shoots and roots, correlating with increasing $CaCO_3$ concentration. Lupine shoot weight decreased from 5.86 grams (g) per pot with no $CaCO_3$ in the soil, to 0.32 g per pot for soil with 6.6 percent $CaCO_3$. George (1987) another Australian researcher also found that lupine growth was inhibited when $CaCO_3$ content in the soil was greater than 5 to 10 percent. A study conducted in Montana (Dollhopf and Mehlenbacher 2002) found a linear decrease in relative plant productivity for two grass species with increasing $CaCO_3$. This study showed that basin wildrye productivity decreased by 65 percent and redtop by 88 percent, as soil alkaline content increased from 0 to 12 percent.

Increased amounts of $CaCO_3$ in the soil system may also increase the potential for some plants to shift root-to-shoot (leaf) ratios, which could affect over all plant vigor. With increasing soil $CaCO_3$

percentage, Fuleky and Hussian (1991) found that with an increasing gradient of soil $CaCO_3$ percentage (0.5, 1, 3, 6, 8, 14, 25), wheat shifted its resources to roots at the expense of above ground parts while in sunflowers the trend was the opposite. The wheat results from the experiment showed, as the $CaCO_3$ content increased, the root dry weight increased by 0.50 grams per pot while the dry weight of shoots drastically decreased by 3 grams. This study points out not only is there a measurable affect with increasing $CaCO_3$, it also shows that various plant species have different responses to increased $CaCO_3$. Although neither sunflowers nor wheat are planned for use on most restoration sites, their sensitivity and clear response to carbonate levels below 10 percent is suggestive of important plant physiological processes that are negatively affected in that range of concentration.

In summary, it remains difficult to identify a specific percent of soil $CaCO_3$ content that would not significantly inhibit plant growth and would unequivocally establish a soil CaCO3 range suitable for revegetation of disturbed lands. Most of the above-cited studies were conducted on agricultural species and do not account for the likelihood that many native species may tolerate higher soil CaCO3 values, as well as decreased access to nutrients and water. Recommendations from the USDA Forest Service regard $CaCO_3$ of 15 percent or less a good soil for revegetation and between 15- 40 percent as a fair soil for revegetation. However, in light of studies of rangeland plants showing negative responses to concentrations even below 10 percent, the culmination of this review leads to a recommendation for a soil $CaCO_3$ content less than 10 percent in soils placed close to the surface (top 1.5 feet) for good to excellent restoration. The surface layers comprise the soil environment in which not only germination occurs but also passage of the fragile stages of plant establishment. A surface layered soil with a $CaCO_3$ content range of between 10 to15 percent would hopefully yield fair revegetation success. For any $CaCO_3$ amount higher than 15 percent, there is an increased chance that the revegetation would not establish well enough to control erosion, create suitable wildlife habitat, or have any aesthetic value. Subsurface concentrations of soil with $CaCO_3$ greater than 15 percent (depths greater than 1.5 feet) would be less deleterious, given that by the time the roots' downward growth reached the $CaCO_3$ rich soil, the rangeland plants would have achieved a more mature and quite possibly more "carbonate-tolerant" stage of growth.

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SOIL COMPACTION EFFECTS ON SLOPE STABILITY AND ROOT GROWTH: AN ARTICLE REVIEW

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ABSTRACT

Few standards exist for determining ideal design parameters for soil compaction when applying vegetation for stabilization and erosion control of slopes and banks. Geotechnical engineers regularly recommend the highest practical soil compaction based on data correlating soil density with increased mechanical strength. Agronomists, on the other hand, recommend minimal soil compaction because compacted soils are widely understood to impede the growth and development of crops, forests and native plant communities. Those who design treatments utilizing vegetation for structural performance, generally known as bioengineering, tend to borrow from various fields with a range of outcomes as a result. This presentation will review several research perspectives and will provide information that can help designers and natural resource managers make decisions regarding soil compaction so as to balance agronomic and mechanical considerations related to the installation and maintenance of bioengineered stabilization treatments.

REVIEW OF ARTICLES

Two articles by Wendi Goldsmith *et al*. (2001) and Donald Gray (2002) discuss issues involved in attempting revegetation on compacted substrates. The tendencies are said to be for geotechnical engineers to strive for the highest compaction levels while agronomists recommend minimal compaction. Practitioners in the field of bioengineering use plant materials for structural performance, and in doing so, they try to combine the engineering and biological aspects of the design process. To do so, the different components of compaction processes need to be understood.

A common type of compaction test was developed by R. R. Proctor in the 1930's. This test involves compacting three successive layers of soil in a 4 inch diameter mold with a 5.5 lb. hammer dropped from a one foot height for 25 successive times. This creates a standardized amount of compactive force that will provide different densities in substrates of different textures and at different moisture levels. By varying the moisture content, the lubricating effect of water results in different density levels at the same compactive effort. This creates compaction curves that gradually increase with greater moisture content and then sharply decline just before approaching the saturation line (tightest possible packing arrangement). More recently, a Modified Compaction Test is used to simulate more intensive compaction levels attained by current equipment. It provides about four times the compaction energy as the earlier Proctor test.

Root growth can be stopped or greatly reduced by excessive compaction. Growth limiting bulk densities have been calculated for different soil textures. Upper limits to compaction may be 1.4 g/cm^3 for clay soils and 1.7 g/cm^3 for sandy soils. While excessive compaction may actually exclude root growth, it may also occur that roots grow into the compacted volume, but then do not function well, either because of limited oxygen content or because the root structures for conducting water and nutrients do not function correctly. The growth-limiting bulk densities are reported to be between 82 and 91 percent of standard Proctor densities (Goldsmith et al. 2001).

Gray (2002) reports that there may be areas of compromise between the objectives of structural stability and plant growth, with regard to compaction activities. The first point is that compaction is not done to increase soil density, it is to change soil properties in a desirable direction, such as to increase soil strength, reduce compressibility or to reduce hydraulic conductivity. Density is only a target condition in specifications. So, density levels may be altered if the desired condition is still met. In one example, changes in substrate moisture content can be used during compaction to alter hydraulic conductivity at the same compactive effort.

The optimum moisture content of a substrate allows the particles to move over each other to attain a maximum packing density, with close particle-to-particle orientation. Excessive moisture can result in less strength in the substrate, as the extra moisture lubricates the particles and they smear across each other rather than developing a closer packing arrangement. Soils that are compacted at water contents that are lower than the optimum level tend to have higher hydraulic conductivity even at the required compaction level. The author cites conductivity increases of two or three orders of magnitude when substrates are compacted using drier than optimum conditions, compared to compaction when wetter than optimum. Whether these changes in hydraulic conductivity are functional for plant roots withdrawing water during daily or monthly weather cycles was not shown.

Both articles list literature and empirical examples of bulk density thresholds that limit root growth. In general, they suggest that a compaction level of 80 to 85 percent of Standard Proctor maximum dry density is still acceptable for root growth. An example is given of a soil in southern California that was compacted to 90 % of Standard Proctor, which was said to have successfully revegetated. Long term plant composition or field compaction data were not provided. Differences in compactability of non-cohesive soils versus cohesive soils, and the difference in pore sizes of sandy versus clay soils were reviewed, but plant response was not reported. In natural soils, much of the root growth occurs in fractures and planes between soil structures, so comparison of normal or adequate root growth between a heterogeneous soil and a homogeneous compacted substrate of a constructed fill is difficult.

Several other soil engineering strategies were suggested as improving revegetation in addition to optimization of compaction levels. These included surface modification such as tillage with disks, ripping, imprinting or trackwalking. These are said to roughen the surface of a smooth, low infiltration substrate into a condition more likely to hold and germinate seeds. Although surface tillage was suggested as being beneficial, the extent to which the compacted slopes required tillage to improve plant growth was not discussed. Soil blending methods were suggested that involve adding coarse textured (gravel) materials to a substrate, so that higher compaction can be attained but sufficient pore space remains to allow root growth. Specially formulated organic-based soil mixes, added to the surface of the compacted substrate were mentioned as reducing runoff and erosion, but the plant response and long-term performance were not discussed. Landform grading patterns was also suggested as being beneficial. This design method creates a non-uniform slope and utilizes planting grasses on drier sites with shrubs and trees in concave, wetter slope locations. Requirements for root growth volumes of these two plant types were not discussed and ability to root into the compacted soil was not documented.

CONCLUSION

In conclusion, the authors provide a useful explanation of the mechanics of compaction processes and measurement. Several possibilities are mentioned for optimizing compaction processes that could improve the chances for plant growth on treated materials, while providing the structural benefits of compaction activities. Examples of actual plant response to these treatments were not covered, however, especially for long term effects. A number of additional surface treatments were also discussed as being beneficial for plant growth, which suggests that while compaction activities may be modified to reduce negative effects on rooting, plant growth is still constrained to some degree by this treatment, and further surface treatments are required. Before wide-scale use on other sites, these suggestions should be evaluated on trial plots under specific local conditions.

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CREATION AND RESTORATION OF HIGH ALTITUDE
WETLANDS IN COLORADO

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ABSTRACT

Creation and restoration of wetlands in high altitude settings presents special challenges. Wetlands require a permanent water supply, and the ecological tolerances of some wetland plant species are relatively narrow. Some high altitude wetlands occur under unique circumstances that are difficult to replicate. These include fens, riparian wetlands along first order streams, hanging garden wetlands, and wetlands associated with solifluction terraces. Yet, high altitude wetlands provide unique ecological functions and they are increasingly threatened by recreational activities, energy extraction and land development. Relatively little work has been done on wetland restoration and creation in high altitude settings. Several projects completed by the author show that certain practices often beneficial for upland revegetation do not necessarily increase success for wetlands. Several factors have been important for success, including adequate characterization of hydrologic conditions, use of proper plant materials, and protection from predation. Long-term monitoring is needed to assess the sustainability of created and restored wetlands in high altitude settings.

INTRODUCTION

Wetlands in the montane, sub-alpine and alpine zones in Colorado, collectively referred to as high altitude wetlands in this paper, are under increased threat of disturbance from recreational uses, water diversions, land development and other impacts. It is estimated that over 53 percent of the 392 million acres of wetlands present in the United States at the time of settlement have been impacted (Dahl 1990). No estimates of the loss of high altitude wetlands in Colorado have been found, but dispersed impacts occur throughout the mountain portion of the state with areas of more concentrated impacts (author's experience).

High altitude wetlands occur in a variety of physiographic settings, some of which are unique to mountainous areas. As with most high altitude vegetation communities, high altitude wetlands exist in a relatively harsh environment characterized by undeveloped soils, a short growing season, intense solar radiation and limited precipitation. Revegetation of disturbed wetlands and restoration of their functions may take many years.

Wetlands are a unique ecotype since they are closely regulated at the federal level by the Clean Water Act (CWA). A federal permit (Section 404 permit) is required to impact a wetland. A similar permit system does not exist for other vegetation or ecotypes in the high altitude environment.

Most of the work on creation and restoration of wetlands in Colorado has been with lower elevation wetlands or with enhancing riparian vegetation at higher altitudes (Mutz et al. 1988). Much of this work has been done on a trial and error basis to create wetlands to compensate for impacts for Section 404 permits. Scientific research on wetlands creation and restoration is lacking. This is especially true for high altitude wetlands, which are typically more sensitive to disturbance and more difficult to restore or create.

This paper relates the author's practical experience with methods for successful creation and restoration of high altitude wetlands. This experience includes planning, design, and monitoring of over 50 acres of wetlands at a variety of locations above 8,500 feet.

BACKGROUND

Wetlands have been defined various ways, with a common theme being the dominance of the area by plant species adapted to prolonged inundated or saturated soil conditions (National Research Council 1995). The most widely used definition was developed for Section 404 of the CWA, which regulates wetlands at the federal level. According to the CWA, wetlands are areas that have the following three factors: 1) hydrophytic vegetation, 2) hydric soils, and 3) an abundant supply of water (U.S. Army Corps of Engineers 1987). Other definitions require the presence of at least one of the above factors (Cowardin et al 1980).

The occurrence of plants under various moisture regimes (i.e., hydrophytic species) has been rated by Reed 1988. Prolonged saturation or inundation of soil leads to anaerobic conditions from degradation of organic material in the soil. Excellent descriptions of changes in soil conditions due to saturation can be found in Vepraskas (1996) and U.S. Department of Agriculture (1996).

Wetlands in mountainous areas in Colorado occur in mountain valleys with primarily igneous or metamorphic geology in association with springs, seeps and areas of excess water. Wetlands also occur on the bottoms of large intermountain basins with sedimentary deposits. High altitude wetlands include more common types such as emergent wetlands, willow carrs, and fens. Less extensive types also occur such as wetlands formed by side-slope processes in nivation depressions and on solifluction terraces. A discussion of the occurrence and characteristics of high altitude wetlands in Colorado and the western United States can be found in Windell et al. (1986). Several studies have been completed to quantify the hydrology related functions of high altitude wetlands. Sundeen et al. (1989) found that the surface and groundwater systems of wetlands along Cross Creek in the Holy Cross Wilderness Area were independent of the streams bisecting the wetlands.

The creation of wetlands refers to the establishment of hydrologic and soil conditions suitable to sustain wetland plant growth (Hammer 1992). Wetland creation typically entails excavation, berming or other earthwork to provide an area with a saturated soils or shallow inundation. Wetlands are created in an area not presently wetland. Wetland restoration refers to the re-establishment of a wetland due to an impact or disturbance (Hammer 1992). For example, re-planting of disturbed vegetation or re-establishment of shallow groundwater to a wetland. Wetland restoration occurs in an area that is presently wetland.

PLANNING AND DESIGN OF WETLAND CREATION AND RESTORATION

Similar information is needed and a similar process can be followed for either creation or restoration of high altitude wetlands (Kentula et al. 1992). In both cases, the goals of the work need to be defined first. These may be dictated by a regulatory requirement if the wetlands are required to be created by a Section 404 permit. In this case, the goal is often to create wetlands of similar structure and function to the impacted wetlands. The goal of wetland restoration, which may also be dictated by a Section 404 permit, is often to revegetate an impacted wetland and restore its functions. A related step is to define the measures of success of the work. For example, that the wetland will be considered successful when there is at least 80 percent cover of wetland species.

Data need to be collected where wetlands will be created or restored or at a reference site, if applicable, to characterize the wetlands and support the design. This typically entails collection of data on the following:

Timing and amount of water sources,
Surface and sub-surface soil characteristics,
Existing plant species, and
Topography.

Other factors that may need to be considered are land ownership, flood hazard, access and water rights.

The design of wetland creation and restoration projects is typically multi-faceted and may require the expertise of a wetland scientist, plant ecologist, hydrologist, soil scientist, civil engineer and geotechnical engineer. More than a revegetation plan needs to be prepared. This is because of the fundamental need to create or restore hydrologic conditions to sustain wetland vegetation. In some cases, a significant amount of earthwork or construction of pipelines and water control structures may be needed.

Development of preliminary and final plans often requires preparation of the following (Garbisch 1990):

Earthwork plan (grading plan),
Planting plan,
Water control structures (e.g., weirs, pipelines, outlet structure),
Spoils disposal plan,
Revegetation plan for adjacent areas,
Erosion control plan, and
Monitoring plan.

Wetland Water Requirements

The amount of water needed to sustain wetlands (i.e. wetland evapotranspiration) is key to successful wetland creation or restoration. Studies have shown that wetland evapotranspiration (ET) varies widely depending on altitude, slope aspect, wetland size and water salinity (Christiansen 1970). Kruse and Haise (1974) determined ET to be 1.2 feet/year for wet meadow wetlands in a lysimeter study at an elevation of 9,100 feet. In a similar lysimeter study in the San Luis Valley at 7,600 feet, Young and Blaney (1942) found wetland ET to be 2.2 feet/year. Studies completed for the Homestake II water project estimated wetland ET at 1.5 feet/year for wetlands at 9,500 feet (ERO Resources 1987). Studies at lower elevations have found wetland ET to be considerable higher, exceeding 5 feet/year (Christiansen 1970).

Available studies indicate the amount of water needed to sustain wetlands:

1. Decreases with increased elevation,
2. Increases for small wetlands surrounded by uplands (“oasis effect”),
3. Does not vary significantly by plant species, and
4. Is approximated by Class A Pan evaporation (unadjusted).

The ET value used can be used to construct a water balance for the wetland. Detailed procedures for constructing a wetland water balance are provided in Pierce (1993). The water balance, and supply to the wetlands, needs to consider both the degree of saturation or inundation and its duration. It is often necessary to install and monitor a series of piezometers on the site to characterize groundwater conditions.

PRACTICAL LESSONS

The success of creating and restoring viable wetlands depends on establishing (or re-establishing, in the case of restoration) hydrologic conditions consistent with the ecological requirements of the wetland species present. Providing an adequate hydrologic regime is the single most important component of success. Challenges to accomplishing this at high altitudes include: the prevalence of steep slopes, undeveloped soils, relatively erosive soils and highly fluctuation hydrologic conditions. Following snowmelt and before the growing season, areas of saturated soils and ponding may be widespread. However, available water typically then diminishes rapidly after runoff as water drains from relatively shallow and coarse soils and precipitation is limited.

The following provides suggestions for site selection for successful wetland creation and restoration at high altitudes.

1) Look for Opportunities. The ability to select a site may be limited, but if possible, use a site with a higher chance of success. For wetland creation, more successful sites include sites next to existing wetlands, adjacent to a drainage and next to a perennial pond or lake. For restoration, select all or the portion of a site that is conducive towards restoration. This is often the area with the least amount of impact (e.g., fill, altered vegetation or other disturbance).

2) Select a Site with an Adequate Water Supply. Relatively flat sites either underlain by shallow groundwater or in proximity to a stream or lake are good candidates. Groundwater can be a more reliable source of water with less variation, but it is necessary to adequately characterize the groundwater table and this requires a site-specific study with piezometers. Creation of wetlands next to a lake with a relatively constant water surface provides a reliable water supply. Use of water in a stream or river is more problematic due to flood hazard and the need to construct a controlled diversion.

3) Use Sites that Require a Minimum Amount of Work and Maintenance. Using a site that does not require a large amount of earthwork can save costs and increase the chance for success. Likewise, it is advantageous to use a site where an adequate water supply can be provided with a minimum amount of work.

   Sites should be used where the design does not include structures requiring a high degree of maintenance. Use of pumps, diversions, and engineered water supply systems, for example, require routine operation and maintenance which may not be practical.

The following provides guidelines for successful design and construction:

1) Mimic the Conditions of Natural Wetlands in the Area or of the Impacted Wetland That is Being Restored. The same or similar setting should be employed. This includes establishment of wetlands with a similar water source, hydrologic conditions and plant communities.

2) Use Redundant Water Sources and Conservative Design Standards. If possible, use both surface water and groundwater. Plan to provide more water than needed and provide gravity discharge from the wetland to drain surplus water.

3) Create Several Planting Zones. Establish several planting zones each with a different hydrologic regime (e.g., an area with shallow flooding and an area with saturated soils but no flooding). Select several species adapted to each zone. The most important characteristic is the moisture regime required.

4) Use Containerized Nursery Stock. Nurseries exist in Colorado and the western United States that grow wetland plant species in small pots or tubes. Use of potted stock, while more costly,

reduces plant establishment time and increases vegetation success. Plants should be planted at a spacing of at least three-feet on center (4,840 plants/acre).

5) Avoid Creating Planting Zones with a Standing Water Greater Than Six Inches Deep. Species typical of wetter conditions in high altitude wetlands (e.g., beaked sedge, water sedge, tufted hairgrass) do not favor prolonged periods of deep, standing water. It will take longer to establish plants in deeper water.

6) Avoid Excessive Compaction. This is more likely with clayey soils. The upper at least three inches should be loose and friable at the finished grade.

7) Provide Diligent Construction Oversight. Final earthwork elevations and grades should be checked for compliance with the plans. Planting materials and methods should be verified.

8) Expected Some Plan Changes. It is common to encounter unexpected conditions during construction that require a change in the plans. This is often due to different soil or hydrologic conditions than anticipated.

9) Revegetate Disturbed Areas. Areas adjacent to the wetlands disturbed by access, construction, stockpiling and related activities need to be revegetated. This includes transition areas adjacent to the wetlands.

10) Use Erosion Controls. An erosion control plan needs to be prepared, and this may be a regulatory requirement. The measures used in the plan need to be installed and maintained throughout construction and into post-construction monitoring.

11) Protect the Site from Predation. Livestock, waterfowl and wildlife may damage a newly planted wetland. Re-planting could be required, which is expensive and will delay vegetation establishment. Fencing can be used to restrict livestock and wildlife. Restricting waterfowl may be more difficult and require netting or use of sound and visual deterrents.

12) Prepare a Weed Control Plan. Invasive weeds are less common at high altitudes but can still present a problem. Emphasis should be on methods for weed control besides herbicides.

13) Observe the Site Following Construction. Observing a site routinely during the first growing season after construction can be valuable to identify potential problems. Correction of problems early on will encourage success.

Based on the author's experience, things that do not necessarily increase the success of wetland projects and are typically used for upland revegetation work include:

1) Providing high quality topsoil,
2) Use of soil amendments and fertilizers, and
3) Mulching of wetland areas.

Other considerations with wetland creation and restoration projects are:

1) Obtain Necessary Regulatory Approvals. Creation and restoration of wetlands, while environmentally beneficial, may require federal, state and local governmental permits and approvals. A Section 404 permit will be needed if existing wetlands will be impacted, even temporarily. This is the case for both private and public property. The process for obtaining a

Section 404 permit is described on the U.S. Army Corps of Engineers website (www.usace.army.mil).

Other approvals that may be needed include: confirmation of no impacts to threatened and endangered species, a Special Use Permit (for federal land), National Environmental Policy Act compliance (e.g., an EA or EIS), Discharge Permit for Construction Activities and local permits and approvals. This latter category includes compliance with city and county wetland protection ordinances. It is a good idea to identify the permits required early in the process as it may take several months to obtain some of them.

2) Long-Term Responsibility. It may be necessary to designate an entity for the long-term maintenance and care of the wetland. If the wetland is being provided for a Section 404 permit or is required by other regulations, it may be required that the wetland be placed in a conservation easement or that deed restrictions be used to protect the site.

3) Water Rights. Water may have to be diverted into a wetland, and wetlands consume a relatively large amount of water. Diversion of water from a stream or exposure of groundwater through excavation may require a water right under state law.

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SOUTHWEST RIPARIAN RESTORATION: NEW STOCK TYPES AND RELEASES, PLANTING CONCERNS, AND SEEDING CONSIDERATIONS

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ABSTRACT

A broad overview of riparian restoration experience was presented by the Los Lunas Plant Materials Center at the Sixteenth High Altitude Revegetation Workshop in 2004. Since that time, new planting methodologies and plant material stock types have been tested to improve establishment and reduce restoration costs. In particular, longstem transplants of riparian understory shrubs have shown promising results in plantings on cottonwood floodplain sites. New releases of important riparian grasses are being developed or are in the process of being released. Recent revegetation experiences have highlighted a number of concerns that can hinder restoration activities including the proliferation of annual weeds following saltcedar control and the effects of inundation on new plantings. Following saltcedar control, many riparian sites in the Southwest have deep water tables and no flooding potential and present serious challenges to establishment by direct seeding. An overview of seedbed ecology is presented to elucidate the factors that control germination and establishment in arid regions. Techniques to maximize success with direct seeding on these sites include appropriate species and ecotype selections, seeding depth control, and mulch application.

INTRODUCTION

The Los Lunas Plant Materials Center (LLPMC) has been involved with the development of plant materials and planting technologies for the revegetation of riparian areas in the southwestern U.S. for over two decades. Although some of these activities have addressed restoration of montane riparian areas, the vast majority of our efforts have involved the cottonwood floodplain forests of the major rivers in the Southwest (Dreesen et al. 2002). During the development of riparian restoration techniques, the LLPMC has conducted numerous large-scale demonstration plantings to test plant materials and planting methods and has monitored the success of these plantings to determine how to improve survival percentages and reduce costs. New planting techniques to establish riparian vegetation with minimal or no irrigation have been developed, some of which can be recommended for broad application in the Southwest.

The 2004 Proceedings of the High Altitude Revegetation Workshop contains a paper by the LLPMC which addresses a broad overview of topics related to restoration of southwestern U.S. riparian ecosystems (Dreesen et al. 2004). The previous paper discussed the mechanisms of riparian disturbance, the selection of revegetation species based on site characteristics, riparian plant material stock types and their production, planting procedures for the various stock types, and case studies of several large plantings in the Middle Rio Grande Valley. The current paper will serve as an update on plant material development and new planting methods as well as address site limitations which have impeded revegetation efforts or have reduced the establishment of desirable vegetation. In addition, the establishment of herbaceous species through direct seeding is an issue gaining increased attention due to large areas requiring revegetation following control of invasive woody species such as saltcedar and Russian olive. The factors that make the establishment of herbaceous cover so difficult in arid regions will be reviewed as well as techniques to improve the success rate of direct seeding.

DEEP PLANTING OF LONGSTEM TRANSPLANTS

Many riparian sites requiring revegetation in the Southwest have relatively deep water tables because of altered hydrology of large rivers due to flood control structures and flow management. The cottonwood floodplain forests can no longer regenerate due to the lack of flooding. The establishment of phreatophytic woody plants (overstory and understory) requires either lengthy irrigation until the transplant's root system can extend into the permanent soil moisture (capillary fringe) above the water table, or planting techniques that allow immediate or rapid root extension into this water source by utilizing deep planting methods.

The LLPMC began investigating deep planting methods over two decades ago with studies to improve pole planting methods by determining the influence of ground water depth relative to pole placement, salinity, and stock attributes (Dreesen et al. 2002). Large-scale plantings based on these results have shown high success rates when site characteristics are not limiting. In the last decade, two other techniques have been tested on large scales: (1) the use of non-rooted dormant poles or large whips of understory species not in the Salicaceae (cottonwoods and willows) family and (2) the planting of rooted stock with very long root balls.

Plantings of non-rooted poles of understory species such as New Mexico olive (*Forestiera pubescens*), false indigo (*Amorpha fruticosa*), false willow (*Baccharis salicina*) have been problematic. The best success rates achieved have approached 50 % for certain species under particular circumstances, but poorer survival rates are more common as well as some complete failures. Factors that may influence establishment of these "understory poles" include the amount of time the pole is hydrated after cutting, the age of the cutting, and planting site characteristics. Although no comprehensive study has been made to ascertain the cause of failures, a number of factors may be important: hydration times after harvest should not be longer than a few days, cuttings from old stems are less likely to root, and planting in fine-textured sediments with poor aeration may retard rooting. Because these species are considerably slower growing than cottonwoods and willows, pole length materials (>6 feet) are by necessity older stems. Although the understory pole technique can work to a limited degree, we do not recommend this technique except when it is the only remaining planting option.

We have been producing riparian understory transplants in 30-inch deep pots (tallpots) for about 10 years. Success rates of 90 percent or more have been achieved in many situations when the bottom of the root balls have been placed in contact with the capillary fringe or when embedded watering tubes have been placed in the planting hole. Depending on the depth to the capillary fringe and soil moisture conditions, up to three irrigations per year using the watering tubes are applied for the first year or two which provides deep soil moisture which allows root extension through the soil above the capillary fringe.

In the last few years, we have encountered riparian planting sites with fairly deep water tables where the bottom of 30-inch root ball is still quite distant from the capillary fringe. Some initial trials with deep burial of tallpot stock in holes up to 6 feet deep have shown positive results using transplants with stem heights up to 6 feet (i.e., total plant height 8.5 ft.). This approach violates several basic horticultural tenets including the deep burial of the root crown and the use of transplants with high shoot-to-root ratios. After one or two growing seasons, samples of each of the species planted using this technique, New Mexico olive (*Forestiera pubescens*), false indigo (*Amorpha fruticosa*), false willow (*Baccharis salicina*), were excavated to ascertain the development of adventitious roots above the root ball. Impressive shoot growth and root observations indicate that extension of roots into the capillary fringe has occurred as well as the development of adventitious roots in shallow soil horizons. The main cause of mortality of longstem plantings has resulted from some sites undergoing prolonged (i.e., 6 week) inundation due to an extreme runoff event in the Middle Rio Grande Valley during the spring of 2005.

As soon as it became apparent that deep burial of longstem planting stock might hold promise for planting in sites with deeper water tables, we decided to test the same procedure with one gallon treepot (4" x 4" x 14") longstem stock. The expense and inconvenience of producing 30-inch tallpots makes treepots an attractive alternative stock type. Longstem treepot stock of the same three species previously mentioned was installed in later comparison plantings along with deep planted tallpot stock. Similar results with survival, growth, and adventitious root development were observed. Although the growing time to produce longstem one-gallon treepot stock may only be slightly less than tallpot stock, treepot production offers the advantages of an inexpensive container, the ease of transplanting seedlings into the container, the ease of watering and moving plants, and the simplicity of supporting and insulating treepots. These efficiencies result in a production cost of a one-gallon longstem treepot being only one-third to one-half of a tallpot. One approach to reduce production time is to plant large bareroot seedlings into treepots, if a source for these native riparian species can be identified. Other species of the cottonwood floodplain forests that might be amenable to longstem deep plantings include golden currant (*Ribes aureum*) and skunkbush sumac (*Rhus trilobata*). We have not yet tried this technique with tree species such as netleaf hackberry (*Celtis reticulata*) or boxelder (*Acer negundo*), but we may have an opportunity to test these species in the near future. Some understory riparian species are not amenable to this technique because of the difficulty in growing longstem material in containers; wolfberry (*Lycium torreyi*) is a prime example.

After the initial longstem deep burial trials were installed, we came across some restoration work from Australia that has taken a similar approach, which they call "longstem tubestock" (Hicks 2003a, Hicks 2003b, Hakewell and Hicks 2004). Their work acknowledges the longstem approach runs counter to conventional horticultural recommendations regarding deep burial and establishment of plants with long stems in small containers. Their approach uses smaller container sizes, 2" x 5" forestry tubes, and attempts to produce stock with stem heights of 3 to 4 feet. Much of their deep planting has been in riparian environments, but they have also used this stock type for arid region plantings in areas with high salinity in surface soils as well sand dune restoration.

New deep plantings are planned which will be monitored for the long-term survival and growth response. Additional riparian species of longstem stock will also be included in new trials to determine their response to deep planting. Shorter and less expensive longstem stock may also be grown in smaller containers for testing on sites where water table depths are not excessive.

PLANT MATERIALS PROGRAM CULTIVARS FOR RIPARIAN SITES

A number of native grass cultivars have been released by the LLPMC which are appropriate for revegetation of cottonwood floodplain riparian sites in the Southwest. Many of these releases are adapted to xeric sites no longer under the influence of periodic flooding. These species are listed in Table 1.

The LLPMC is in the process of releasing alkali muhly (*Muhlenbergia asperifolia*) Westwater Germplasm from San Juan Co., New Mexico. This species is an aggressive rhizomatous species principally adapted to moist soils along streambanks and ditches, but it is also found on dryer floodplain sites. A release of vine mesquite (*Panicum obtusum*) is also being planned and will contain a composite of Southwestern accessions which produce high seed yields; this species is a stoloniferous/rhizomatous grass of heavy soils in swales, playas, and low spots. The LLPMC has been working for a decade on a release of giant or big sacaton (*Sporobolus wrightii*) which has been selected for its large (8-10 feet tall) upright stature for use as an herbaceous windbreak. It also should be suitable for revegetation of xeric floodplain sites; sacaton meadows still exist on some undisturbed floodplains along secondary drainages in central and southern New Mexico. A release of sandhill muhly (*Muhlenbergia pungens*) is also contemplated from germplasm collected from xeric sandy areas in northwest New Mexico.

Table 1. Native grass cultivars released by the Los Lunas Plant Materials Center that are suitable for revegetation of southwestern U.S. riparian areas. Most of these species are adapted to xeric sites no longer undergoing periodic flooding.

Scientific Name	**Common Name**	**Cultivar**	**Origin**
Achnatherum hymenoides	Indian ricegrass	Paloma	Pueblo, CO
Andropogon hallii	sand bluestem	Elida	Elida, NM
Bothriochloa barbinodis	cane bluestem	Grant Germplasm	composite from AZ and NM
Bouteloua curtipendula	sideoats grama	Niner	Socorro Co., NM
Bouteloua curtipendula	sideoats grama	Vaughn	Guadalupe Co., NM
Bouteloua eriopoda	black grama	Nogal	Socorro Co., NM
Bouteloua gracilis	blue grama	Alma	composite
Bouteloua gracilis	blue grama	Hachita	Hachita Mtn., NM
Bouteloua gracilis	blue grama	Lovington	Lea Co., NM
Elymus elymoides	bottlebrush squirreltail	Tusas Germplasm	composite from NM
Pascopyrum smithii	western wheatgrass	Arriba	Kit Carson Co., CO
Pleuraphis jamesii	galleta grass	Viva	Newkirk, NM
Schizachyrium scoparium	little bluestem	Pastura	Pecos, NM
Sorghastrum nutans	Indiangrass	Llano	Elida, NM
Sporobolus airoides	alkali sacaton	Salado	Socorro Co., NM

SITE LIMITATIONS IMPEDING RESTORATION EFFORTS

The experience of implementing numerous riparian restoration demonstrations for the last two decades has yielded a list of concerns which have often hampered the installation or success of projects (Los Lunas Plant Materials Center 2005a, 2005b). In the past few years, problems with inundation of planting sites from extreme water releases and dense herbaceous weed stands following exotic woody species control have been large impediments to recent projects. Other site limitations that have often affected revegetation ease or success are described as well as some potential responses or solutions to these hindrances.

Flooding Resulting in Prolonged Inundation

A site consideration which has not received adequate attention in recent years is the impact of significant flood events and prolonged inundation. This inattention is reasonable considering the drought the Southwest has been experiencing for many years. In the late spring and early summer of 2005, a controlled release of massive quantities of snowmelt water stored in reservoirs was released in the Rio Grande. Within the confines of the levee system, many low lying areas were flooded, and many of these areas remained inundated from six to eight weeks. Several sites that had been planted in the spring of 2004 and 2005 were inundated. High mortality rates of pole and containerized stock (tallpot and longstem treepot) were observed for plants that had been planted several months prior to the flooding. A majority of the pole plantings that had been installed in 2004 survived the inundation while those planted in 2005 succumbed. If extreme snowmelt flood events are forecast, it would be advisable to delay plantings in low areas until later in the year or into the next year or make sure the inundation potential for the site is known in advance.

Effect of Weed Competition on Revegetation

Proliferation of annual weeds can drastically influence reseeding efforts to re-establish native grasses and forbs. After the control of invasive exotic woody species, it is paramount that land managers consider the herbaceous weeds that frequently invade such areas after clearing and the accompanying disturbances. For severe weed infestations on disturbed sites, herbicidal control of weeds for two or more years may be necessary to reduce the weed seedbank before direct seeding and to maximize revegetation potential. The survival and growth of small containerized stock will be severely diminished by competition with large dense weed stands which shade transplants and deplete soil moisture. In some extreme situations, the installation of weed barrier fabrics in V-ditches or basins can be used for planting woody species to reduce weed competition, to harvest runoff, and to reduce evaporation.

Extreme Depth to Ground Water and Severe Water Table Fluctuation

Measurement of depth to ground water using shallow monitoring wells will confirm the depth and seasonal fluctuation of the water table to help determine appropriate species and the most effective stock type (container depth or pole length) for revegetation. Extreme depths to groundwater may indicate the only practical restoration goal is revegetation with xeric shrubs and grasses rather than riparian species.

Revegetation Limitations Due to Soil Salinity and/or Soil Texture Extremes

Fine-textured soils or soils with restrictive layers can limit the selection of species and stock types for revegetation. Soils with high percentages of cobble can make augering impossible; whereas, augered holes in dry sands and gravels will often collapse before planting. Visual observation of soil samples from augered holes should be sufficient to determine if soil texture or restrictive soil layers will be limiting. Extreme salinity and sodicity of floodplain soils can profoundly influence species suitable for revegetation. Salinity problems (i.e., electroconductivity greater than 3 dS/m) can be especially persistent in clay soils where natural leaching of salts is limited. Augered soil samples can be analyzed for electroconductivity (EC) to determine if surface or subsurface salinity is a problem. Electromagnetic induction field instrumentation can also be used to rapidly estimate soil salinity for large acreages.

Loss of Planting Stock from the Scouring Action of Flood Flows

Dormant pole and whip cuttings planted to substantial depths can resist the extractive forces of flood flows compared with shallow planted containerized and cutting stock. Willow whips with their inherent flexibility are more appropriate for higher flow regimes and less stable channel systems. In lower elevation situations where scouring is severe, it is advantageous to plant containerized stock with deep root balls during the fall to provide some root development prior to spring runoff. Some riparian species in small containers but with long stems (i.e., longstem stock) can be buried in deep planting holes for anchorage. Many riparian species should be adapted to this planting method which is comparable to natural burial by sediment deposits.

Eradication of Woody Invasive Species and Removal of Resulting Biomass

A long-term commitment for spot spraying of sprouts must be part of any control program. The dead biomass resulting from herbicide treatment can be burned in slash piles for interspersed noxious woody plants or by crown fires in monoculture stands. The removal of large diameter biomass as firewood and burning of slash is another alternative. The mulching of dead biomass is expensive, but the benefits of mulch include limiting wind and water erosion, reducing soil moisture loss, and enhancing salt leaching by decreasing evaporation and increasing infiltration. A mulch layer will also retard the growth of weeds that commonly occurs after clearing operations.

Woody Riparian Plant Communities versus Wet Meadow Communities

Planting sites should be evaluated to determine whether they are a wet meadow environment and not appropriate for woody vegetation. Shallow depth to ground water and fine-textured organic-rich or anaerobic soils are some of the factors consistent with wet meadow environments. On low elevation floodplains, saltgrass (*Distichlis spicata*) meadows are inappropriate for revegetation with woody species because of shallow groundwater as well as generally high levels of soil salinity.

Planting Equipment Access

Large equipment requires site access which can be restricted by ditches, arroyos, levees, soft sand, or steep slopes. One unanticipated problem with equipment access, which has been identified with the recent upsurge in saltcedar clearing, is the ubiquitous presence of cut stumps which can easily puncture heavy duty truck and tractor tires.

Protection and Maintenance of Revegetated Sites

The continued spot spraying of sprouts of noxious woody species and any other invasive weeds will be required for an indefinite period. Protection from cattle will require adequate fencing and periodic monitoring of fence integrity. The presence of beaver necessitates poultry wire tree guards around individual pole plants as well as protection of unplanted poles and whips placed in streams or canals for hydration. Controlling defoliating insects is crucial for pole plantings during the initial growing seasons; cottonwood leaf beetle will occasionally require control.

ESTABLISHMENT OF HERBACEOUS SPECIES BY DIRECT SEEDING IN DISTURBED RIPARIAN AREAS

The revegetation of riparian sites disturbed during the eradication of invasive woody species and by wildfire has resulted in direct seeding being extensively used as a conservation practice in riparian areas. In the arid Southwest, such plantings frequently fail to accomplish the intended conservation objectives. After saltcedar control, the deep water tables, saline fine-textured soils, and scarce precipitation make many of these sites especially difficult for establishing herbaceous cover by seeding. The expense and effort expended on seeding provides motivation to thoroughly investigate all the factors which may help to maximize the likelihood of successful establishment. Successful establishment requires the coincidence of seed situated in favorable microenvironments, precipitation sufficient to stimulate germination, subsequent precipitation pulses to allow seedling establishment, and negligible competition from weeds and insignificant herbivory (Call and Roundy 1991). Technical resources detailing the numerous aspects of revegetation by seeding have been developed in recent years and serve as excellent sources of background information and practical advice (e.g., Monsen et al. 2004, Colorado Natural Areas Program 1998).

Precipitation is the Controlling Factor

Many Southwestern floodplain forests are situated in arid regions with less than 10 inches (254 mm) of annual precipitation. Many of these riparian sites no longer undergo flooding and have deep water tables; thus, these sites are truly arid ecosystems relying on infrequent, variable, and highly unpredictable precipitation (Noy-Meir 1973). At the Jornada site in the northern Chihuahuan desert of New Mexico, long-term climatic data shows an average annual precipitation of 247 mm with 54% falling in the July-September period and 50 rainy days per year but one third of these days have rainfall amounts less than 1 mm (Reynolds et al. 2004). Storm pulses (rainfall events on successive days) of less than 5 mm occurred an average 17 times per year; whereas, pulses between 5-10 mm, 10-15 mm, and greater than 15 mm occurred an average of 6, 3, and 3 times per year, respectively (Reynolds et al. 2004). These data are long-term averages which overstate the number of significant pulses in drought years.

An estimate of how large of a pulse is required for a significant recruitment of seeded species is complicated by a myriad of weather and site variables. For grass seedings, near surface soil moisture content must be sufficient to allow seed imbibition and germination, seminal root extension, coleoptile emergence, and sufficient seminal and adventitious root development for the seedlings to survive the succeeding dry inter-pulse. Soil water in the top inch of soil is depleted from optimal to inadequate levels in 1 to 4 days after a rainstorm in hot desert areas (Winkel 1991a). For a number of desert grass species, if seeds imbibe for two or more days and then experience a dehydration event, substantial mortality of germinating seed results (Emmerich and Hardegree 1996). Seed of Arizona cottontop (*Digitaria californica*) exposed to three successive days of water applied in total amounts of 3 mm, 6 mm, 10 mm, and 15 mm had germination percentages of 0%, 15-20 %, 50-70%, and 90-95%, respectively (Smith et al. 2000). Adventitious root initiation in grasses requires 2 to 4 days of optimal soil water conditions (Winkel 1991b). Two scenarios can be postulated regarding wet-dry sequence effects on seed and seedling survival: 1) a wet period sufficiently short that the seed does not germinate during the wet or subsequent dry period or 2) a wet period sufficiently long to produce a seedling with vigor and root development to survive the following dry period (Frasier et al. 1985).

The storage of moisture in the top 100 mm of soil is critical for germination and establishment (Noy-Meir 1973). The volumetric water storage capacity (i.e., the difference between soil at field capacity and dry soil) of sands range 3 to 6 %, and for clays from 15 to 25 % (Noy-Meir 1973). For a sandy soil, a rainfall pulse of 5 mm infiltrating the soil surface would wet approximately the top 100 mm of soil and could result in significant germination and root elongation. In a heavy soil, an infiltration pulse of 5 mm would only wet approximately the top 25 mm; this surface soil moisture could be depleted rapidly by evaporation. Based on storm pulse data, recruitment events for sandy soils during the growing season would be infrequent in average years but rare in drought years and very rare for heavy soils.

The preceding precipitation data indicates the low likelihood of precipitation pulses adequate for recruitment events. The unpredictability of precipitation pulses within decades, years, and seasons, makes it paramount to maximize the use of the precipitation that occurs by selecting the appropriate species and ecotypes, by burying the seed at optimal depths for establishment, by manipulating the seedbed to conserve near-surface soil moisture.

Species and Ecotype Selection

Appropriate native species should be given top priority when specifying seed mixes. Unique situations may require the use of introduced species, for instance, to better compete with invasive weed competition. Introduced species are often used when (1) appropriate native species are not available and (2) adapted introduced species can be identified which will not adversely affect the surrounding ecosystem. Often seed cost is used as the primary reason to justify the seeding of introduced species. This economic rationalization must be balanced against long-term ecological repercussions.

The surrounding plant community can be used as indicator of suitable species, especially if nearby sites with minimal disturbance can be identified. Descriptions of range cover types (e.g., Shiflet 1994), ecological sites (e.g., USDA-NRCS New Mexico 2006), and other plant community lists (e.g., Dick-Peddie 1993) can be useful to determine common or dominant species for various plant community types.

By selecting species suited to the soil texture and chemistry, the chances of successful establishment are greatly enhanced. Certain species perform best on well aerated (well drained) coarse (sandy) soils whereas others perform better on fine-textured (clay) soils. The salinity and sodicity of the soil will have profound effects on which species may be established. If the planting site contains a variety of soil textures, a seed mix could include species suitable for the range of textures and salinity. Conversely, separate mixes could be used if the site can be delineated into separate soil types and seeded accordingly.

Cultivars resulting from selection or breeding as well as source-identified germplasm have been developed by various Plant Materials Centers in the western U.S. and have been extensively tested in seeding trials. Many of these native plant materials are appropriate for riparian restoration seedings depending on the eco-region in question and other site characteristics. Use of cultivars or germplasm from the applicable eco-region is generally preferred. If such plant materials are not available, testing has shown that some cultivars have broad areas of adaptation.

Other seed characteristics of particular species which should be considered include the dormancy of the seed. In agronomic settings, seed dormancy is undesirable due to reduced germination rates or percentages. However, in wildland restoration, particularly in areas where seedbed conditions conducive for a recruitment event are rare, it is very desirable to have seed of some species persist in the seedbank. Non-dormant seed can imbibe water and initiate germination as a result of precipitation events insufficient to allow the establishment of the seedling. By initiating germination with inadequate soil moisture, this seed is lost from the seedbank for future adequate soil moisture events which could result in regeneration. Some grass species have seed coat-induced dormancy and/or embryo dormancy. These types of dormancy may be desirable attributes in order to retain viable seed in the soil seedbank for several years. Seed coats can be barriers to water or oxygen uptake, impediments to embryo expansion, or sources of germination inhibitors (Adkins et al. 2002). After-ripening is often referred to as the development of a mature embryo after seed harvest by storage under warm, dry conditions; moist chilling or stratification has also been classified as after-ripening during which the dormant seed is transitioning to a germinable state (Foley 2001).

Seedbed Ecology

Some of the dominant issues regarding the manipulation of the seedbed to improve the likelihood of establishment include control of annual weed competition, conservation or concentration of soil moisture, depth of seed burial, and optimizing seedbed environmental conditions. The rarity of optimum precipitation pulses for recruitment is justification to manipulate those factors which could maximize establishment with scarcely adequate soil moisture events.

As previously described, proliferation of annual weeds (e.g., *Kochia scoparia*) following the removal of invasive exotic woody species often occurs during the restoration of floodplain cottonwood forests. Soil disturbance such as made by heavy equipment traffic, extraction of root crowns, and skidding fallen trees often result in flushes of annual weed growth. Thick mulch layers resulting from shredding or chipping this biomass can suppress this weed growth. If this mulch layer is disrupted during seeding to achieve seed contact with mineral soil, annual weeds could proliferate. If annual weeds invade right after invasive species removal, it is of paramount importance to control these stands before they can release additional weed seed into the soil seedbank. To reduce the weed seedbank, herbicidal control may be required for several years. Alternatively, controlled burns of herbicide-killed annual weed stands might produce sufficient soil temperatures to reduce the weed seedbank. The ability of many of the common annual weeds to establish with minimal moisture inputs portends little or no survival of seeded species having to compete with such weed stands.

The depth of seed burial is a crucial factor in establishment of grasses and forbs. A number of factors influence optimum depths including intrinsic seed characteristics of the species as well as soil and site factors. The depth of seeding of grasses is influenced to a great degree by the length of coleoptile (the structure that forces through the soil while protecting the plumule bud). Some grasses (panicoid type) have an internode (sub-coleoptile) structure which allows the reach of the coleoptile to be the total length of the coleoptile plus the internode (Hyder et al. 1971). The presence of the internode in panicoid grasses results in adventitious roots developing well above the seed position (Hyder et al. 1971). Establishment of grass seedlings is dependent on the development of adventitious roots from the crown node (between the coleoptile and sub-coleoptile internode); elevation of the crown node by the elongation of the internode results in root development occurring in the moisture limited near-surface soil (Tischler and Vogt 1993).

The other grass type, festucoid, does not have an elongated sub-coleoptile internode resulting in the lowermost adventitious roots developing near the seed planting depth (Hyder et al 1971). Under sub-optimal soil moisture conditions, adequate emergence of seedlings from shallow seed burial depths must be balanced against the more favorable moisture environment at greater depths.

An ideal seedbed assures that the seed is surrounded by soil particles firmly packed around the seed to ensure conductivity of water from the soil to the seed (Winkel et al. 1991b). Very small seeded species can be sown on the soil surface where this intimate contact with soil particles is provided without any disturbance beyond rain drop impact (Winkel et al. 1991b). For broadcast or drilled seed, firming of the seedbed is recommended to assure adequate seed to soil contact. In areas where equipment traffic has compacted surface soils, ripping or other tillage methods may be required to provide seedbed tilth sufficient to allow optimal root elongation and resulting drought resilience.

Seedling recruitment depends on the number of seeds in favorable micro-sites (Call and Roundy 1991). The micro-topography of the seedbed surface can greatly influence seedbed temperature and moisture: cracks, depressions, rocks and gravel, and plant litter can all play a significant role in eventual germination and establishment (Call and Roundy 1991). Depressions retain surface moisture longer and have more favorable temperature regimes than smooth soil surfaces; these imprints also aid seed burial by trapping wind-blown particles and by soil sloughing off the sides of the depression (Call and Roundy 1991). Deep-furrow rangeland drills, rangeland imprinters, and contour furrowers have improved seedling recruitment under certain soil and site conditions (Call and Roundy 1991). Contour furrows improved recruitment by increasing moisture storage and leaching salts from the surface soil; furrow treatments were most effective for medium to fine-textured soils (Branson et al. 1966). Contour furrowing provided favorable microenvironments in the bottom of the furrow for seedling recruitment in salt desert habitats (Wein and West 1971). Soil cracks can also provide beneficial micro-environments for seedling establishment (Winkel et al. 1991b).

Surface mulches can provide substantially enhanced micro-site environments. Gravel and plant litter mulches provided 4 to 5 days longer favorable soil moisture than bare soil under situations of intermittent water pulses (Winkel 1991b). These mulches provided increased emergence under all watering scenarios (daily, intermittent, and single pulse) for surface-sown seed (Winkel 1991b). Thick mulch applications can be detrimental to seedling survival if the mulch layer hinders coleoptile emergence or causes increased elevation of coleoptile node in grasses and results in adventitious root development in more droughty surface soils. Thin straw mulch applications with gaps exposing the soil surface should provide some micro-site enhancement but not alter seedling root development (Hyder et al. 1971). Litter and by implication thin mulch layers modify seasonal and diurnal temperature patterns, moderate the diurnal range of relative humidity, and delay water depletion in the soil surface (Call and Roundy 1991). Mulch or litter layers need to be anchored to prevent redistribution by wind forces. Vertical crimping of straw is one of the most frequently employed methods of anchoring mulch.

In the arid Southwest, revegetation of xeric riparian sites by direct seeding always will be problematic, especially in times of drought. By proper selection of species and ecotypes, seeding methodology, and manipulation of the seedbed environment, the chances of successful establishment can be maximized. An alternative approach to the restoration of diverse plant communities involves the use of seed source islands or seed islands to provide a natural source for seed dispersal and eventual seedling recruitment (Reever Morghan et al. 2005). In arid regions, intensive cultural practices (e.g., irrigation, herbivore exclusion, mulch application) could be employed to establish these small islands of diverse plant communities. The continued dispersal of seed should provide soil seedbanks which over time will establish an expanding community around the periphery of the seed source islands. This approach would involve an alternative expenditure of resources compared with conventional seeding methods. Direct seeding represents a large-scale, non-intensive, immediate, high-risk venture compared with seed source islands which involve small-scale, intensive, enduring, low risk endeavors. However, the patience required for plant community expansion from the seed source islands is not an attribute of most restoration projects.

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COTTONWOOD ESTABLISHMENT ON BOULDER CREEK

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ABSTRACT

The City of Boulder and the City of Lafayette created a joint habitat mitigation site along Boulder Creek in the spring of 2003 as compensatory habitat mitigation for impacts to wetlands and riparian habitat on Boulder Creek and nearby wetlands in the City of Boulder Open Space. Compensatory habitat mitigation was associated with construction of the 75th Street diversion structure and pipeline and an outfall into Boulder Creek. The long-term goal of the habitat mitigation project (50 years or more) is to create a mature cottonwood grove that will provide a potential nesting substrate for great blue herons, great egrets, and black-crowned night herons. The habitat mitigation site is about 2.35 acres in size.

ERO Resources, the City of Boulder, and the City of Lafayette coordinated to create a sandy substrate and a temporary flood irrigation system to be used during cottonwood establishment. The site was prepared in the spring of 2003, prior to cottonwood seed dispersal and heron nesting, and was flood irrigated during the spring and early summer of the first growing season. Seed from the adjacent mature cottonwood stand was used as a seed source for the site. Seed was allowed to disperse naturally.

The habitat mitigation site has been very successful during the first three years of monitoring. The average density of cottonwoods calculated from transect data was5.56 cottonwoods/square meter during 2003, 6.83 cottonwoods/square meter during 2004 monitoring, and about 4.37 cottonwoods/square meter during 2005 monitoring. The habitat mitigation site currently is exceeding success criteria for cottonwood stem density and is expected to meet or exceed both cities goals for habitat mitigation.

INTRODUCTION

In the spring of 2003, Lafayette and Boulder created a habitat mitigation site at the Culver farm in Boulder County, Colorado. The habitat mitigation site is located near an existing heron rookery, and is an open area characterized by a small stand of immature cottonwoods located in the center. The City of Boulder Open Space and Mountain Parks (BOSMP) would like to see this site become a replacement stand of cottonwoods for the heron rookery. The long-term goal (50 years or more) is to create a mature cottonwood grove that will provide a potential nesting substrate for great blue herons, great egrets, and black-crowned night herons. ERO Resources coordinated with the BOSMP to create a 2.35-acre site suitable for the establishment of plains cottonwood.

CONSTRUCTION METHODS

Grading

The site was graded with a uniform slope with an eastern to northeastern aspect. Following grading, the subsoil was ripped with grooves running perpendicular to the fall line of the slope so that flood irrigation water will flow through the site more slowly.

Sand Placement

Sand was placed at a depth of 4 to 6 Inches throughout the habitat mitigation site to minimize competition from weeds and other vegetation. Over most of the site, pit run, composed of a mix of 80% sand, 10% silt, and less than 10% gravel, was applied. Masonry sand, with a much smaller particle size than the pit run was placed in one 200-square foot area in the northeastern portion of the site.

POST CONSTRUCTION ACTIVITIES

Irrigation

An open, flat ditch was graded at the upslope end of the habitat mitigation site to supply water to the site. Minor grading of the ditch and creation of small lateral feeder ditches was necessary to evenly disperse water throughout the site. During May and June 2003, the site was flood irrigated using water pumped from nearby lakes. In May and early June 2003, the habitat mitigation site was flood irrigated daily for two weeks when plains cottonwood in the area disperse seed. Irrigation was discontinued by mid-June 2003, and the site has not been irrigated since that time.

Seeding and Planting

During initial cottonwood establishment, the habitat mitigation site were not seeded or planted. Seed naturally that dispersed from neighboring cottonwoods germinated and established in the habitat mitigation site. After two years of cottonwood growth, the site was seeded in late 2005 to help establish native grasses.

Weed Control

During each growing season since 2003, weed control has been very important. Weeds such as tamarisk, Canada thistle, and yellow sweet clover have colonized the site. Weed control has included both herbicide application and hand pulling. Opportunities for herbicide application were limited because of the close proximity of wetlands and other water bodies, and because cottonwood seedlings were difficult to avoid. Backpack sprayers were used wherever possible to individually spray weeds, especially hard to control weeds such as Canada thistle. Other weeds were controlled by hand pulling.

MONITORING METHODS

Natural recruitment of cottonwoods was quantitatively assessed at the habitat mitigation site on September 19, 2003, September 14, 2004 and August 30, 2005. An average density per square meter was determined from a reference site by sampling the density from 5 meter x 5 meter test plots, located every 10 meters along a 50 meter transect through the center of the reference site.

To determine the average density of cottonwood recruitment in the habitat mitigation site, three permanent transects were established. The number of cottonwood seedlings were counted in 1 meter x 1 meter plots located every 5 meters along the transects. Data were collected from a total of 43 plots. The monitoring results were not analyzed statistically because this level of accuracy was not required for the project.

Success Criterion

A success criterion was established to assess the short-term success of the site using the density of the existing stand of cottonwoods (about 10 years old) located in the center of the habitat mitigation site. This stand was used as a reference to establish a cottonwood density success criterion against which annual monitoring of the habitat mitigation site was compared. The average density of cottonwoods in the reference stand of cottonwoods was estimated to be 1 cottonwood/square meter.

RESULTS

Three years of quantitative monitoring have been conducted. The 2003 data from the reference stand of cottonwoods revealed an average density of one cottonwood/square meter. Table 1 lists the results of 2003, 2004 and 2005 monitoring.

Table 1. Annual Monitoring Results.

Plot	Transect 1			Transect 2			Transect 3		
	2003	**2004**	**2005**	**2003**	**2004**	**2005**	**2003**	**2004**	**2005**
1	9	8	3	12	3	3	0	4	1
2	1	6	1	4	9	3	0	0	0
3	0	1	0	6	16	10	19	29	15
4	0	1	1	0	2	2	31	52	30
5	2	0	2	0	0	2	45	40	17
6	5	3	7	1	0	3	33	60	27
7	4	2	2	0	1	0	26	22	14
8	1	0	1	0	0	0	18	14	7
9	0	5	0	1	0	1	12	10	10
10	1	2	1	0	0	0	0	0	3
11				0	0	0	0	1	6
12				0	0	1	0	0	0
13				1	0	1	0	0	0
14							0	0	0
15							0	0	0
16							0	0	0
17							0	0	2
18							3	0	1
19							1	3	9
20							3	0	2
Mean	**2.30**	**2.80**	**1.80**	**1.92**	**2.38**	**2.00**	**9.55**	**11.75**	**7.20**
Standard Deviation	**2.91**	**2.70**	**2.04**	**3.55**	**4.81**	**2.68**	**14.10**	**18.90**	**9.14**

2003 Monitoring Results

For 2003, the average density of cottonwoods along Transect 1 was 2.3 saplings per square meter and ranged from 0 to 9 individuals. The average density of Transect 2 was 1.92 cottonwoods per square meter and ranged from 0 to 12 individuals. The average density of cottonwoods along Transect 3 was 9.5 saplings per square meter and ranged from 0 to 45 individuals. The average density of all three transects was 5.56 cottonwoods per square meter in 2003.

2004 Monitoring Results

At the time of the 2004 site visit, the average density of cottonwoods along Transect 1 was 2.8 saplings per square meter and ranged from 0 to 8 individuals. The average density of Transect 2 was 2.38 saplings per square meter and ranged from 0 to 16 individuals. The average density of Transect 3 was 11.75 saplings per square meter and ranged from 0 to 60 individuals. The average density of cottonwoods calculated from 2004 transect data was 6.83 cottonwoods/square meter, well above the success criterion of 1 cottonwood/square meter. In 2004, the highest density of cottonwoods occurred in the southwest corner of the site.

2005 Monitoring Results

At the time of the 2005 site visit, the average density of cottonwoods along Transect 1 was 1.80 saplings per square meter and ranged from 0 to 7 individuals. The average density of Transect 2 was 2.00 saplings per square meter and ranged from 0 to 10 individuals. The average density of Transect 3 was 7.2 saplings per square meter and ranged from 0 to 30 individuals. The average density of cottonwoods calculated from transect 2005 data was 4.37 cottonwoods/square meter, well above the success criteria of 1 cottonwood/square meter.

CONCLUSION

A suitable substrate and appropriate hydrologic conditions are necessary for cottonwood establishment. Cottonwood seed dispersal and germination coincide with spring runoff events, in which peak flows often deposit suitable substrates and raise the water table. The goal of the project was to provide a suitable substrate and create appropriate hydrologic conditions for the establishment of cottonwoods. After the first three growing seasons since construction, the Culver mitigation site appears to have successfully established a replacement cottonwood stand. After initial construction and irrigation, the site has required relatively little maintenance other than annual weed monitoring and control. The established success criterion has been met during the first three years of monitoring with many relatively dense areas of cottonwoods.

WEED CONTROL AND HABITAT RESTORATION AT THE ROCKY MOUNTAIN ARSENAL NATIONAL WILDLIFE REFUGE.

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ABSTRACT

Weed control is a major step in habitat restoration work at the Rocky Mountain Arsenal National Wildlife Refuge. Many of our sites have been planted to crested wheatgrass since the mid 1940s. We are trying to get at least two years of weed control on new restoration sites before they are planted and irrigated. We are working to deplete the seed bank in order to reduce competition for our restoration seeding.

CONTROLLING CHEATGRASS WITH HERBICIDES
TO ESTABLISH PERENNIAL SPECIES

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ABSTRACT

The Division of Oil, Gas and Mining and U. S. Magnesium performed revegetation work on a mined area heavily infested with cheatgrass (*Bromus tectorum* L.). After the area was graded and ripped, Plateau® herbicide was applied to part of the area at a rate of 3 oz. per acre in an attempt to control the cheatgrass and allow establishment of perennial species. We then seeded the entire area with a mixture of range grasses and shrubs. After two years, there is little difference in the amount of perennial vegetation cover in clayey swales, but in areas with sandy soils, there is significantly more perennial vegetation in the sprayed compared to the unsprayed areas. There is also less cheatgrass and more perennial vegetation in areas where surface soil was removed in the grading process.

INTRODUCTION

Establishing desirable perennial vegetation in areas dominated by cheatgrass (*Bromus tectorum* L.) can be very difficult (Monsen 1994). In a cooperative effort between five state and federal agencies, and as part of a friendly bond forfeiture agreement with U. S. Magnesium, the Utah Division of Oil, Gas and Mining seeded about 185 acres mostly heavily infested with cheatgrass. Before seeding, 76 acres were sprayed with Plateau®, an herbicide registered for use in controlling annual grasses.

METHODS AND MATERIALS

Site Description

U. S. Magnesium's Rowley plant is just west of the Great Salt Lake about 12 miles north of Interstate 80. It is on a flat, ancient lakebed, and the distance from the Great Salt Lake varies from about 100 yards to a few miles depending on the level of the lake.

In the 1980s and 1990s, Magnesium Corporation of America or MagCorp (now a bankrupt company whose assets were purchased by U. S. Magnesium) mined deposits of oolitic sands between about two and four miles north of the plant. The operator did not salvage topsoil and did not seed following mining. There was limited grading to level piles created during mining.

The oolitic sands areas are mostly flat with sandy soils, but there are numerous small playas with clay soils. We have not tested the soils, but because of the vegetation community, soil texture, and proximity to the Great Salt Lake, we assume the clayey soils have high salt concentrations. We did not take quantitative vegetation cover measurements prior to seeding, but dominant species in the playas consisted of greasewood (*Sarcobatus vermiculatus*(Hook.) Torr. in Emory) and bottlebrush squirreltail (*Elymus elymoides* (Raf.) Swezey). The areas from which oolitic sands were taken are about 2-4 feet higher than the playas, and vegetation cover was almost exclusively cheatgrass with scattered greasewood and bottlebrush squirreltail. The slope transition between the two areas was near angle of repose.

Annual precipitation at the nearest weather station in Grantsville, Utah, on the south end of the lake (about 24 miles away) averages 11.72 inches. Three years of below average precipitation preceded this revegetation effort, but monthly precipitation in 2004 and 2005 was consistently close to or above normal. Exceptions were March, July, and August of 2004, but there have been no months with less than 0.20 inches of precipitation (information from Western Regional Climate Center, Desert Research Institute, http://www.wrcc.dri.edu/.)

Grading and Revegetation

The agreement reached between the Division of Oil, Gas and Mining (the Division) and U. S. Magnesium was that U. S. Magnesium would use its dozers to rip the entire area about six inches deep and to grade the transition area between the clay playas and the sandy areas to about a 3h:1v slope so the area could be drill seeded. This work was done in September and October 2003. The Division would then take over the reclamation bond and do the rest of the revegetation work.

The Division contracted with a private pesticide applicator, Harward Farms, to apply three ounces of Plateau® herbicide and one quart of methylated seed oil per acre. This was mixed with 20 gallons of water and applied using a spray rig that had a GPS guidance system and controls that automatically adjusted the volume of spray being emitted as the velocity of the sprayer changed. This was completed on October 8, 2003, before there had been any significant rain or germination of the cheatgrass.

The herbicide was applied on land managed by the Utah School and Institutional Trust Lands Administration and on private land totaling about 76 acres. We were not able to obtain permission to spray land managed by the Bureau of Land Management, about 109 acres. The BLM was under a court injunction to not use Plateau® on any of its lands in Utah or Idaho.

The entire 185-acre area was drill seeded (except forage kochia—see below) on November 5-13, 2003. The seed mix and approximate rates of application were:

Species	Pounds Pure Live Seed/Acre
Forage Kochia	1
Shadscale	1
Russian Wild Rye	4
Rimrock Indian Ricegrass	1
Hycrest Crested Wheatgrass	3
Tall Wheatgrass	1
Bottlebrush Squirreltail	1

The rangeland drill used for this project had three banks of seed boxes. This allowed for separation of species that had special seeding requirements or that would be unlikely to compete well if planted in a mix with other species. Shadscale seed was put in a seed box that only fed the outside drops. The forage kochia seed was put in the alfalfa box, and the tubes were pulled from the drill openers allowing the seed to dribble on the surface.

Sampling

On September 29, 2005, Lynn Kunzler used the point-intercept method (10 points per transect, 20 transects in each area) to measure perennial vegetation cover in three areas: fee land on the south, state land on the north, and BLM land in the center. On all three parcels, measurements were taken in the clayey playas and in the slightly raised sandy areas. For vegetation cover analysis, data from comparable areas of fee and state lands were combined. On the BLM property, an additional set of samples was taken on the regraded slopes between the playas and the raised areas. The dozers had scraped the surface soil from much of this area. The established vegetation appeared to have less cheatgrass and a significant component of seeded species compared to adjacent unsprayed areas. Mr. Kunzler estimated cover from cheatgrass but did not make precise measurements.

A Student's t test was used to compare perennial vegetation cover in sprayed and unsprayed raised and playa areas.

RESULTS

Vegetation Cover

Table 1 shows results of vegetation cover sampling. There was significantly more perennial vegetation in raised areas where Plateau® was applied than in comparable areas that were not sprayed, and it appeared cheatgrass was suppressed in the sprayed, raised areas. There was no difference in perennial cover in the clayey, playa areas where there is little cheatgrass cover.

Table 1. Vegetation cover percentages in raised, playa, and slope, sprayed and unsprayed areas. Cheatgrass cover values are estimates. Cover was not measured on sprayed slope areas, and cheatgrass cover was not estimated on unsprayed playa areas. Within columns, means with different letters are significantly different with 95 percent confidence.

	Perennial Cover (measured cover)			Cheatgrass (estimated cover)	
	Raised	Playa	Slope	Raised	Playa
Sprayed	20.75a	6.75a	X	20-30	<2
Unsprayed	1.50b	5.50a	6.50	90	X

It appeared scraping soil from the slopes contributed to an increase in perennial cover in these areas, but the statistical comparison was not made.

We noticed, but did not quantify, a decrease in the number of other annual species in sprayed compared to unsprayed areas.

We have noticed few shrubs in the seeded areas which could be due to any of several factors, including lack of tolerance to Plateau®. The label lists tolerance to application of 8-12 oz./acre. Of those species seeded, bottlebrush squirreltail and Russian wild rye are considered tolerant, crested wheatgrass tolerance varies depending on certain conditions, and the tolerances of tall wheatgrass, Indian ricegrass, shadscale, and forage kochia are not listed.

Costs

<u>Sprayed areas</u> (Costs for seed, the tractor, the vehicle, and personnel were proportioned based on the number of acres)

Plateau® and MSO	$ 941.41
Herbicide Application	490.00
Seed	2,159.75
Tractor Rental/Repair/Fuel	800.81
Vehicle Rental (Motor Pool) includes fuel for all inspection trips	399.91
Personnel including all benefits and wages for all inspection trips	4453.55
Total	$9,245.43

Cost per acre: $121.65

<u>Unsprayed areas</u>

Seed	3,097.54
Tractor Rental/Repair/Fuel	1,147.81
Vehicle Rental (Motor Pool) includes fuel for all inspection trips	573.55
Personnel including all benefits and wages for all inspection trips	6,387.33
Total	$11,206.23

Cost per acre: $102.81

Costs not included:
Seed Drill and Trailer (Use donated by USDA Forest Service Great Basin Experiment Station and Utah Division of Wildlife Resources)
Trailer (Use donated by Utah Division of Parks and Recreation)

As can be seen from these figures, the difference in cost between the sprayed and unsprayed areas was $18.84.

DISCUSSION

Using Plateau® herbicide allowed increased germination and initial establishment of perennial species, but it did not eliminate cheatgrass completely. Information from the manufacturer, BASF, indicates Plateau® has residual effect for about two to three years, but we do not know if the suppression of cheatgrass about two years after application is due to the residual effects, seed bank depletion, competition with now-established perennials, some other factor, or a combination of these.

The rate of application of Plateau® was decided upon in consultation with a BASF representative. We wanted to use a lower rate both to reduce costs and to lower the risk of damaging germinating perennial species. Application rates need to be higher when there are larger amounts of litter, but since much of the litter was eliminated when the areas were graded and ripped, we were able to apply it at a reduced rate.

Plateau can be used as either a pre- or post-emergent herbicide, and we elected to spray before cheatgrass had germinated. It is recommended that seeded grasses reach the five-leaf stage before spraying Plateau as a post-emergent, and since cheatgrass competition might prevent most of the seeded grasses from ever reaching this stage, we felt it would be better to spray in the fall.

While the initial results are promising, stresses from soil conditions and herbivory in addition to competition from cheatgrass create a very difficult environment. This can be compounded by additional stresses from drought and potential wildfires. We intend to continue monitoring the site to see whether cover from perennial species can be maintained or increased or if the results we now see are only temporary.

CONCLUSION

Reclamation of areas dominated by cheatgrass is difficult at best. We did not eliminate cheatgrass from the vegetation community but controlled it sufficiently for perennial vegetation to germinate and establish. Using Plateau® was found to be an economical method to establish perennial vegetation in this situation.

ACKNOWLEDGMENTS

With the limited funds available, this project could not have been completed without the assistance in equipment and manpower of the USDA Great Basin Experiment Station, the Utah Division of Wildlife Resources, the Utah Division of Parks and Recreation, the Utah Department of Transportation, the Utah School and Institutional Trust Lands Administration, U. S. Magnesium, and BASF. Harward Farms and Pacific Tri-Star provided reduced rates to apply the herbicide and for the tractor rental. We greatly appreciate the cooperation these agencies and companies provided.

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INVASIVE SPECIES MANAGEMENT IN THE HAYMAN FIRE AREA

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ABSTRACT

In the Hayman Fire of 2002, over 137,000 acres of private, State, and National Forest system lands were burned. During such large scale events, the risk of exposure to invasive species increases. The U.S. Forest Service took several steps to prevent the introduction of invasive species and reduce their spread during fire suppression activities. After the fire was out, we also attempted to lower the risk of introducing noxious weeds during the BAER (Burned Area Emergency Rehabilitation) treatments. I will discuss the lessons learned in both situations. Also, I will present the noxious weed treatment strategy we have been following since 2003, discussing high priority species and the results to date in their control.

INTRODUCTION

This is a brief account of the lessons learned during and after the Hayman fire concerning our management of invasive species. I'll address our prevention efforts during the fire suppression activities and post-fire BAER activities. I'll also discuss our treatments to date, our joint efforts with partners, and the current status of several noxious weed species in the fire area.

Prevention Efforts During the Fire

During the Hayman fire, over 2500 firefighters and support staff and over 150 engines were deployed to the fire area (Graham 2003). Fire crews came from many states and the vehicles they brought had the potential to be carriers of noxious weed seed. In order to lower our risk of introducing invasive species, we focused our prevention efforts on the equipment that would be used in the fire area. We added noxious weed-free contract specifications for all contracted equipment and took direct action at the fire camps to establish vehicle wash stations for the fire crews. We also worked with the aerial operations staff to reduce our risk of inadvertently transferring whirling disease parasites from positive waters into clean waters. Lastly, we established a whirling disease de-tox station for fire crews to clean their water-holding tanks, hoses and fittings to ensure that whirling disease-positive water was not carried back to their home units.

Prevention Efforts During BAER

Burned Area Emergency Rehabilitation efforts began quickly after the fire was out. Most treatments included either aerial or ground seeding and many required a mulch product placed over the seed. In addition to continuing to ensure that the equipment used for these treatments did not carry noxious weeds, we added preventative measures for the material. Although we used other products, I will only discuss the seed and straw mulch materials that were used on the fire.

We used a mix of 70% barley (*Hordeum vulgare*) and 30% winter triticale (x *triticosecale* Rimpaul) and purchased almost 2 million pounds of seed. The seeds of these non-persistent cereal grains are large, free-flowing and relatively easy to clean. We selected these seeds to enhance our ability to obtain a noxious weed-free product. We used strict contract language that addressed both prohibited and restricted noxious weed seed, and had representative samples from each seed lot tested at the CSU lab for the presence of noxious weed seed. This process worked fairly well, although the rapid time schedule presents a challenge for seed testing.

In 2002 and 2003, we purchased over 24 million pounds of straw to be used during aerial straw application over the fire area. This treatment creates a protective layer on the soil surface and keeps soil particles from detaching and moving during rain events. Although the contract required certified weed-free straw, an inspector found several clumps of *Bromus tectorum* in a recently treated area. Unfortunately, at that time we realized that *Bromus tectorum* was on the Colorado state list of noxious weed species, but was not on the regional weed-free forage list shared by western states. We were able to isolate the straw shipments that were suspect and did not use this material. The contract was ultimately stopped for government convenience, re-written, and issued again in 2003 with specifications requiring certification that the straw had been inspected for *Bromus tectorum* and several other similar-looking species. It was helpful to contact the weed-free forage program managers in the western states to alert them to our added certification requirements.

Noxious Weed Management Actions

In addition to these preventative measures, we also conducted a rapid inventory of noxious weeds in the fire area in 2002. In 2003-2004, we conducted a more thorough survey in areas with a high risk for expansion, especially along roads, trails and drainages in the fire area. Large portions of the fire area have not yet been surveyed.

We also began herbicide treatments in 2002 and plan to continue these treatments into the future. We attempt to treat approximately 1500 acres annually, although this is funding dependent. Table 1 lists the noxious weed species selected for treatment in the fire area.

Table 1. Estimated acres of treated noxious weed species in the Hayman Fire area

SPECIES	**ACRES**
Orange Hawkweed *(Hieracium aurantiacum)*	6
Spotted Knapweed *(Centaurea stoebe* L. ssp.*mMicranthos)*	<1
Scentless Chamomile *(Tripleurospermum perforata)*	<1
Houndstongue (Cynoglossum officinale)	<1
Dalmation Toadflax *(Linaria dalmatica)*	7
Leafy Spurge *(Euphorbia esula)*	100
Diffuse Knapweed *(Centaurea diffusa* Lam.)	40
Scotch Thistle *(Onopordum acanthium)*	200
Bull Thistle *(Cirsium vulgare* (Savi) Ten.)	100
Russian Thistle *(Salsola tragus)*	25
Cheat grass *(Bromus tectorum)*	75
Yellow Toadflax (*Linaria vulgaris* P. Mill.)	2000
Canada Thistle (*Cirsium arvense* (L.) Scop.)	500

We have begun to see overall reductions in stem densities and plant vigor, and in some sites, plants have been eradicated. Orange hawkweed has been reduced to <1 acre, spotted knapweed to <50 stems, and leafy spurge to approximately 30 acres. Yellow toadflax remains elusive with some sites exhibiting reduced stem densities and plant vigor while other sites seem relatively unaffected.

We began to release biological control agents in 2002 although the initial releases were small. In 2005, we worked with the Colorado State University staff to release *Mecinus janthinus* on 18 sites with dalmation or yellow toadflax (Hardin and Norton 2005). We will monitor these sites in 2006 to determine if the insects have established.

We also released 350,000 *Apthona* beetles on leafy spurge both in and outside of the fire area. We have had success with these insects in the past on other sites outside of the fire area. We also released insects for musk thistle (*Rhinocyllus conicus*) on a few sites to re-establish populations in the fire area. More releases of both insects are planned for 2006.

In the fall of 2002, we met with other land owners and managers in the fire area who were interested in controlling noxious weeds and developed the Hayman Fire Noxious Weed Treatment Strategy (USDA Forest Service 2002). This organization continues to meet annually to coordinate our control methods, increase educational efforts and improve efficiencies. The members include weed managers from the four counties in the fire area (Jefferson, Douglas, Park and Teller), Denver Water, Colorado Division of Wildlife, Colorado Department of Transportation, Natural Resources Conservation Service, interested private citizens, and the Coalition for the Upper South Platte (CUSP). CUSP has been instrumental in working with private landowners in the fire area to encourage weed control on private lands. The continued efforts of this group will be critical to our future management of noxious weeds in the Hayman fire area.

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RULE CHANGES PERTAINING TO THE ADMINISTRATION
AND ENVORCEMENT OF THE COLORADO NOXIOUS WEED ACT

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ABSTRACT

In 2003, the Colorado legislature revised the Colorado Noxious Weed Act (C.R.S. 35-5.5) to restructure the state noxious weed list. These revisions provided a framework for implementing a more coordinated effort to stop the spread of noxious weeds in Colorado. The Colorado Department of Agriculture is utilizing this framework to develop and implement statewide noxious weed management plans for a variety of noxious weeds. Through the rule-making process, the Department has classified noxious weeds into one of the three lists: Lists A, B, and C. List A species are designated for statewide eradication in order to prevent them from establishing permanent and significant populations in Colorado. The goal for List B species is to stop their continued spread in Colorado. List C species are species for which no statewide management plan will be prepared due to their widespread nature but will become the focus for increased research and biological control efforts. This annual rule-making process provides an opportunity for public and private interests to participate in the development of species-specific management plans that guide local and statewide efforts to manage targeted species in a coordinated manner.

INTRODUCTION

In 2003, the Colorado Department of Agriculture led a diverse coalition of agricultural, environmental, and governmental interests organized to press for significant changes to the Colorado Noxious Weed Act (C.R.S. 35-5.5) that would help implement Colorado's strategic plan to stop the spread of noxious weeds. While the statute already provided a basic framework that enabled and required local governments to manage noxious weeds (e.g., adoption of a management plan/ordinance, appointment of a local advisory board, enforcement powers), it did not facilitate more coordinated, cooperative wed management efforts across the broad, multijurisdictional landscape of the state. The changes advanced in 2003 and subsequently passed by the Colorado General Assembly attempt to rectify this shortcoming.

STATE WEED LIST

Among the most significant change was the restructuring of the state noxious weed list. The prior state noxious weed list had been primarily a means to educate citizens, local, state, and federal agencies, and concerned public interest groups about the plant species that are non-indigenous to Colorado and cause significant impacts to agriculture and the environment. This list did not direct management or even identify the kinds of management that would be appropriate for specific species where they are located in Colorado. The purpose of restructuring the list was to provide the Colorado Department of Agriculture with the means to implement a more coordinated and cost-effective statewide effort to stop the spread of noxious weeds. Implementation of the new list and the act's amended provisions provides a regulatory framework to help all jurisdictions work towards a common solution for targeted species.

The revised state noxious weed list is now comprised of three separate categories:

1. List A noxious weed species (Table 1) are designated for statewide eradication in order to prevent the establishment of permanent and significant populations in Colorado that would facilitate the

spread of the species throughout the state. These species are so rare that statewide eradication is feasible and advisable given their extensive spread and negative impacts in other western states.

2. List B species (Table 2) are more common than List A species and, consequently, statewide eradication is no longer possible. Instead, the goal for all List B species is to stop their continued spread through the development and implementation of a statewide noxious weed management plans. Such plans will designate specific areas of Colorado for eradication, containment, or suppression to achieve this goal and promote a coordinated and strategic management effort across jurisdictional boundaries.

3. List C species (Table 3) are those noxious weeds that are so widespread that it may no longer be feasible to stop their spread. Consequently, the State's objective is not to prepare a statewide plan to coordinate management activities but rather to assist communities through research, technical assistance, and biological control in the management of local populations and the mitigation of their impacts.

This restructured state noxious weed list has provided the Department with the regulatory framework to develop and implement statewide noxious weed management plans for a variety of noxious weeds. Since 2004, the Department has amended the state weed list and adopted management plans for a number of species. The following chronology summarizes these actions:

CHANGES

<u>2004 – Rule Changes</u>

The state noxious weed list was evaluated and 74 species were redesignated while 11 species including kochia, Russian thistle, and blue mustard were removed. All 74 species were then classified in Lists A, B, and C:

List A: 18 species, 9 of which are known to be present in Colorado.
List B: 41 species
List C: 14 species

Management plans were adopted for all List A species requiring eradication statewide wherever the species are detected.

<u>2005 – Rule Changes</u>

Statewide noxious weed management plans were developed and adopted for several List B species: absinth wormwood, Chinese clematis, plumeless thistle, and spotted knapweed. In addition, several watersheds (upper Colorado River, North Platte River, Rio Grande River, and upper San Miguel River) were designated as eradication zones for tamarisk.

<u>2006 – Rule Changes</u>

Statewide noxious weed management plans were developed and adopted for several List B species: black henbane, diffuse knapweed, oxeye daisy, and yellow toadflax.

In 2007, the Department will focus on the development and adoption of noxious weed management plans for Canada thistle, houndstongue, perennial pepperweed, spurred anoda, Venice mallow, and yellow nutsedge. While the plans for houndstongue and perennial pepperweed will be statewide in nature, the management plans for the other species will focus primarily on agronomic areas where their potential and realized impacts are most severe. There will be many opportunities to participate in the development of these plans. Here is an outline of the steps the Department expects to follow in this process to develop and adopt plans for these species:

Late August: Issue preliminary survey to counties
September: Assemble data and issue quarterquad survey soliciting additional data and preliminary containment lines
October: Assemble data and preliminary plans
Early November: Meet with advisory committee to discuss
November 15: Post draft plans on web
Nov/Dec: Disseminate revised QQ maps and draft plans to agencies and counties for input
Early January: Meet with advisory committee to discuss
Mid-January: File proposed rule with Secretary of State and post on web
March: Hold public hearing
June 1: New rules will become effective

The public is welcome to provide information and comment at any stage of this process in order to help the Department develop and adopt noxious weed management plans that will facilitate more coordinated efforts to stop the spread of these noxious weeds.

Table 1. List A Colorado Noxious Weed List.

List A	
Common Name	**Scientific Name**
African rue	*Peganum harmala*
Camelthorn	*Alhagi pseudalhagi*
Common crupina	*Crupina vulgaris*
Cypress spurge	*Euphorbia cyparissias*
Dyer's woad	*Isatis tinctoria*
Giant salvinia	*Salvinia molesta*
Hydrilla	*Hydrilla verticillata*
Meadow knapweed	*Centaurea pratensis*
Mediterranean sage	*Salvia aethiopis*
Medusahead	*Taeniatherum caput-medusae*
Myrtle spurge	*Euphorbia mysinites*
Orange hawkweed	*Hieracium aurantiacum*
Purple loosestrife	*Lythrum salicaria*
Rush skeletonweed	*Chondrilla juncea*
Sericea lespedeza	*Lespedeza cuneata*
Squarrose knapweed	*Centaurea virgata*
Tansy ragwort	*Senecio jacobaea*
Yellow starthistle	*Centaurea solstitialis*

Table 2. List B Colorado Noxious Weeds.

Common Name	Scientific Name
List B	
Absinth wormwood	*Artemisia absinthium*
Black henbane	*Hyoscyamus niger*
Bouncingbet	*Saponaria officinalis*
Bull thistle	*Cirsium vulgare*
Canada thistle	*Cirsium arvense*
Chinese clematis	*Clematis orientalis*
Common tansy	*Tanacetum vulgare*
Common teasel	*Dipsacus fullonum*
Corn chamomile	*Anthemis arvensis*
Cutleaf teasel	*Dipsacus laciniatus*
Dalmation toadflax, broad-leaved	*Linaria dalmatica*
Dalmation toadflax, narrow-leaved	*Linaria genistifolia*
Dame's rocket	*Hesperis matronalis*
Diffuse knapweed	*Centaurea diffusa*
Eurasian watermilfoil	*Myriophylium spicatum*
Hoary cress	*Cardaria draba*
Houndstongue	*Cynoglossum officinale*
Leafy spurge	*Euphorbia esula*
Mayweed chamomile	*Anthemis cotula*
Moth mullein	*Verbascum blattaria*
Musk thistle	*Carduus nutans*
Oxeye daisy	*Chrysanthemum leucanthemum*
Perennial pepperweed	*Lepidium latifolium*
Plumeless thistle	*Carduus acanthoides*
Quackgrass	*Elytrigia repens*
Redstem filaree	*Erodium cicutarium*
Russian knapweed	*Acroptilon repens*
Russian-olive	*Elaeagnus angustifolia*
Salt cedar	*Tamarix chinensis, T. parviflora, and T. ramosissima*
Scentless chamomile	*Matricaria perforata*
Scotch thistle	*Onopordum acanthium*
Scotch thistle	*Onopordum tauricum*
Spotted knapweed	*Centaurea maculosa*
Spurred anoda	*Anoda cristata*
Sulfur cinquefoil	*Potentilla recta*
Venice mallow	*Hibiscus trionum*
Wild caraway	*Carum carvi*
Yellow nutsedge	*Cyperus esculentus*
Yellow toadflax	*Linaria vulgaris*

Table 3. List C Colorado Noxious Weed List.

List C	
Common Name	**Scientific Name**
Chicory	*Cichorium intybus*
Common burdock	*Arctium minus*
Common mullein	*Verbascum thapsus*
Common St. Johnswort	*Hypericum perforatum*
Downy brome	*Bromus tectorum*
Field bindweed	*Convolvulus arvensis*
Halogeton	*Halogeton glomeratus*
Johnsongrass	*Sorghum halepense*
Jointed goatgrass	*Aegilops cylindrica*
Perennial sowthistle	*Sonchus arvensis*
Poison hemlock	*Conium maculatum*
Puncturevine	*Tribulus terrestris*
Velvetleaf	*Abutilon theophrasti*
Wild proso millet	*Panicum miliaceum*

VEGETATION MONITORING OF AUGMENTATION AREAS

ALONG BIG DRY CREEK AT HIGHLANDS RANCH, COLORADO

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ABSTRACT

Undisturbed prairie swales are generally well protected from erosion by dense plant cover. Even with sizable open space set backs, swales in new developments along the Front Range corridor, still tend to degrade quickly when utilized for storm water conveyance. This presentation describes the installation and first few years of monitoring results for an innovative vegetation augmentation trial in Highlands Ranch. This method shows potential for reducing erosion, costly drainage repairs, and associated ecosystem degradation.

INTRODUCTION

In the spring of 2003, a proactive channel protection treatment was installed on the Upper Big Dry Creek drainage in Highlands Ranch, utilizing vegetation augmentation. This concept was developed by the Restoration Group, Inc. in conjunction with Muller Engineering Co., Inc. The following report describes the trial augmentation project and presents sampling results for the third year of vegetation monitoring. Because of the difficulty in tracing unpublished reports, much of the information presented in the earlier reports is included.

BACKGROUND

Undisturbed prairie swales are well protected by the native vegetation. Swales are generally well vegetated by rhizomatous perennial species, typical of upland grasslands. These species are capable of providing adequate above and below ground erosion protection, as long as other sources of disturbance have not degraded the plant cover. During a heavy precipitation event, above ground vegetation may be washed flat, creating a thatch of protection for the soil surface. Below the surface, dense fibrous root systems provide a second line of defense, gripping the soil particles and resisting erosion. Periods of soil saturation (anaerobic conditions) are generally short term. The vegetation can recover as the storm water is spread across the swale floor, leaving soils moist, but still aerobic.

The natural resistance to erosion, in prairie swales, may be destabilized by disturbance. Historic disturbances, such as fires and grazing, remove herbaceous plant cover, exposing the soil surface. Coupled with associated damage to root systems, these disturbances can leave the effected areas vulnerable to degradation. Heavy precipitation events which occur while the plant cover is limited, can lead to erosion of deep channels. These V-channels leave lasting evidence of historically disturbed landscapes.

New housing developments in the Front Range corridor often rely upon native prairie swales for storm conveyance. These swales are vegetated by upland species. Unlike wetland vegetation, the native swale vegetation requires aerobic soil conditions to maintain health. Extensive grading coupled with construction of new roofs and pavement leads to increased run off, extended periods of soil saturation and elevated silt deposition. While these changes may not be overly detrimental to certain wetland communities, they can be very serious to the historic native upland species still present in the swales.

Native upland vegetation may become stressed when soil saturation is extended over several weeks. The above ground leafy portions of the plants stop growing. Below the ground, the suffocating root systems lack vitality to recover when silt deposits bury the weakened plant, leaving the soil vulnerable to erosion. In a few years, V-channel development can progress dramatically even where generous development setbacks have been required. Sandy soils or those derived from wind blown loess may be particularly susceptible to erosion.

Areas of bare soil from erosion or silt deposition are vulnerable to weed invasion. Weed seeds are washed in with run off or transported by the wind. In the Front Range communities of Colorado, these disturbed areas maybe quickly colonized by rapidly moving noxious weeds such as Canada thistle, Scotch thistle, musk thistle, bull thistle, common mullein, and diffuse knapweed. Weed roots are generally less resistant to erosion and may do little to slow channel degradation. Storm events further undercut steep banks, causing them to fail; washing away any recently established side slope vegetation and further degrading channels.

Even before erosion expands the channel very far, the downcutting process lowers the local hydrology, which leads directly to disturbance of the remaining native swale floor vegetation. The V-channel transports storm water away rapidly, robbing the remaining valley floor of the time necessary to recharge soil moisture. The flanking swale soils clinging to the banks above the channel are now high and dry. Hydrologic isolation, leads to a broadening zone of ecosystem disturbance, called desertification. The plant communities on the swale flanks degrade, as some of the species dependant upon periodically higher soil moisture, weaken and die out. Upland weeds with shallow root systems begin to colonize the degrading plant community. Cheatgrass, common mullein, and diffuse knapweed are weedy colonizers. Within a few years of the new housing development's arrival, the formerly well vegetated prairie swale may have lost much of its historic vegetation. In such channels, a single precipitation event can lead to significant down cutting. Once a V-channel has begun, its steep banks offer little resistance to erosion. A period of rapid degradation can follow.

When site conditions have changed permanently as a result of the disturbance, a different set of species may be required than those initially present on the site. The process of natural succession will gradually provide better adapted native species to fill in a disturbed area. The natural agents for distribution of native species are wind, water, and wildlife. It may take years for natural processes to provide better adapted native species to the wet swale areas. By the time these species arrive, irreversible erosion damage may have already occurred.

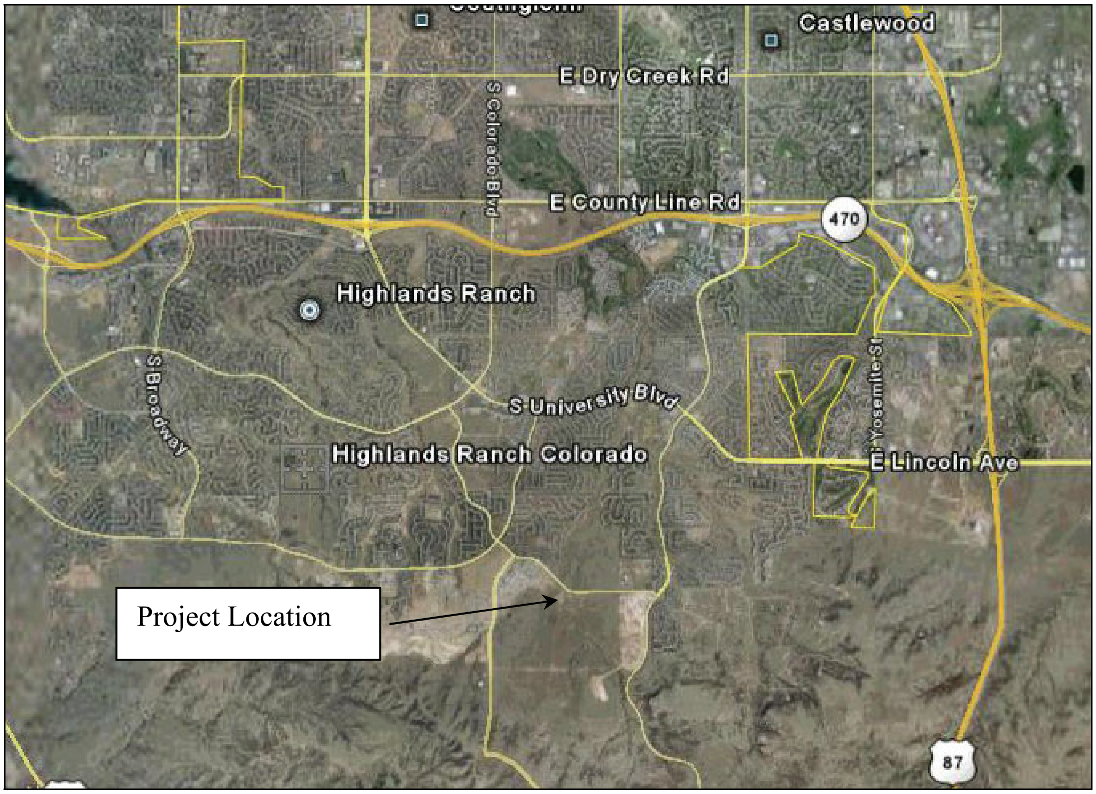
Early in the process of development, as storm waters are first introduced, proactive vegetation augmentation with better adapted native hydric species, can help preserve the channel configuration. By not permitting the loss of soils, hydrology, and the integrity of the historic native plant communities, it may be possible to preserve the quality of native open space areas and prevent erosion requiring costly repairs. If properly planned, installed, and managed, vegetation augmentation offers the potential to reduce erosion, diminish ecosystem degradation, and maintain a more aesthetic open space system (Photo 1). Grade control and drop structure installations in deeply incised channels may cost from $150 to $400 a linear foot to repair. Vegetation augmentation offers an alternative treatment at a considerable savings.

VEGETATION AUGMENTATION PROCEDURES

During spring 2003, two areas of the upper west tributary on Upper Big Dry Creek just east of Briar Glen Lane and Briarglen Circle in Highlands Ranch (Maps 1 and 2) were selected for the trial of the augmentation channel improvement. The area was divided into a lower reach and an upper reach. Each selected reach is fed by a newly installed storm drain at the upper end, entering from the west side of the valley (Photo 2).

Photo 1. Upper Big Dry Creek prior to development and augmentation installation. Note first houses on upper right. Feb 2003.

Photo 2. Augmentation monitoring area, in the lower reach, just after installation, April 5, 2003.



Map 1. Regional air photo showing project location.

Map 2. Project area along Upper Big Dry Creek, southeast of Briarglen Lane.

In early April, 2003, augmentation species were planted (Photo 3). Installation areas were selected based on the current drainage and pattern of soil saturation in the swale. The plants were installed in areas where the historic vegetation was stressed due to soil saturation and silt deposition. Augmentation species that were planted included:

1100	Prairie cordgrass (*Spartina pectinata*)
550	Nebraska sedge (*Carex nebrascensis*),
550	Woolly sedge (*Carex lanuginosa*)
200	Hardstem bulrush (*Schoenoplectus lacustris* ssp. *acutus*)

Photo 3. Volunteers and Highlands Ranch staff installing herbaceous plants in the augmentation monitoring area.

Plants were distributed within the two reaches on 18 to 24 inch centers. Two thirds of the plant materials were installed in the monitoring area, in the lower reach, due to its less shrubby and moister condition. Bulrushes were planted closest to the storm drain outlets. Just downstream, woolly sedge and Nebraska sedge were installed. Below this, and at the margins of the channel, Prairie cordgrass plants (Photo 4) were planted. (A recent view of the same vegetation is provided in Photo 5 for comparison.)

The plants were robust, well filled 10 cubic inch (10 T) wetland plugs. Planting holes were pried open using a planting bar (a heavy narrow 12 inch steel wedge mounted on a 3 foot handle). The planting bar worked better than trenching shovels due to the difficulty digging in heavy, well vegetated clay soil. The operation was completed by supervised volunteers. It required approximately four hours to install plants in the monitoring area.

Dormant bareroot two year old plants and cuttings of native riparian shrubs were installed as thickets scattered along the flanks of the channel.
Shrub species included:

25 - Red-osier dogwood (*Swida sericea* syn: *Cornus stolonifera*) (also live cuttings)
150 - Wild plum (*Prunus americana*)
100 - Chokecherry (*Padus virginiana* syn: *Prunus virginiana)*

The shrubs were planted on three to five-foot centers in single species thickets. Other than visual observations for survival, the woody species have not been included in the monitoring study.

Photo 4. View of Prairie cordgrass (broad-bladed grass in center) during first growing season, in the original native Buffalograss sod (shorter green sod), an upland species in the augmentation monitoring area, August 2003. (Lighter colored grasses are weedy annual grasses.)

Photo 5. Three year old Prairie cordgrass plants in the central portion of the augmentation area. August 2005.

METHODS

In order to evaluate potential vegetation changes along the Big Dry Creek floodplain, permanent sampling transects were installed. Three transects were located within the augmentation installation area, near the wetter central channel. Two transects were located along the flanks of the historic swale floor, away from the central channel, in areas less likely to be effected by increased soil saturation. In all, five transects were installed.

Vegetation data were collected along the transect using a point intercept approach. This method utilizes an ocular sighting device in conjunction with transects. Sampling consists of evaluating what is "hit" by the crosshairs in the viewing field of the sighting device. The optical sighting device is a precision instrument that has been designed to reduce parallax, provide a clear bright viewing field, and to utilize very fine cross hairs so that the evaluation point is nearly dimensionless. The design consisted of establishing a single transect 25 meters in length at each sampling location. For sampling, the sighting device is attached to a tripod which is leveled at each sampling point. Observations were made on each side of the 25 meter transect at 1.0 meter intervals along the transect for a total of 50 point observations per transect. Each of the sampling locations was permanently marked using concrete reinforcing rods. Latitude and longitude data for each transect were obtained using a hand held global positioning system receiver. GPS location data are accurate to approximately 15 feet. Each transect was photographed with a digital camera.

DESCRIPTIONS OF PERMANENT TRANSECTS

The vegetation along each of the transects is described in the following section.

Transect A-A’

Transect A-A’ was established in the native vegetation on the unaffected floodplain of Big Dry Creek. The vegetation is dominated by upland prairie species and currently is not included within the vegetation augmentation areas. If the overall hydrologic regime of the site were to change dramatically, vegetation changes might occur at this transect location. The transect was located at this site as a means of monitoring natural vegetation fluctuations along Big Dry Creek. Future data from this transect will also provide a means of evaluating the extent to which weedy species may be invading the site.

In 2005, the major species along this transect included western wheatgrass *(Agropyron smithii)*, blue grama *(Bouteloua gracilis),* green needlegrass *(Stipa viridula)* and skunkbush sumac *(Rhus trilobata)*. These four species accounted for 93 percent of the cover along the transect. An additional 19 species were observed within one meter of the transect line (Table 1). Total vegetation cover was 82 percent. None of the augmentation species were encountered along this transect (Figure 1). In all, 23 species were observed either along the transect or within one meter of the transect line.

The vegetation at this site was similar to what was observed in 2003 and 2004. The transect is dominated by native upland perennial grasses. Some differences in species composition and abundance were noted, however these are likely related to the time of sampling rather than to actual changes in the vegetation. The moisture regime in this part of the floodplain appears to be comparable to what it was in 2003 and 2004. No effects of stormwater discharge were noted.

Transect B-B’

Transect B-B’ was established along the channel of Big Dry Creek. The upstream part of the transect is within the augmentation area. The vegetation along the transect consists primarily of a mixture of species characteristic of the dry floodplain along Big Dry Creek. Planted augmentation species occurred to only a limited extent along the transect in 2003 but were more prevalent in 2004.

Major perennial grass species along this transect included western wheatgrass *(Agropyron smithii)*, and prairie cordgrass *(Spartina pectinata)*. These two species accounted for approximately 34 percent of the cover by all species. In 2003, Nebraska sedge *(Carex nebrascensis)* was noted along the transect. In 2004, cover for this augmentation species had increased to 18 percent (Figure 2), which was 30 percent of the total cover along the transect. In 2005, cover by augmentation species was 8 percent. Cover by weedy species increased from 6 percent in 2004 to 40 percent in 2005. The tall weedy species were much larger than the understory perennial grasses. The remainder of the cover was distributed among four other species (Table 1). Total vegetation cover was 76 percent. In all, 23 species were observed either along the transect or within one meter of the transect line.

Overall, moisture conditions were comparable to 2004 with moist to wet surface soils along the length of the transect. The abundance of the weedy species tended to reduce the abundance of the augmentation species. Overall, augmentation species accounted for 10 percent of the cover along the transect.

Transect C-C’

Transect C-C’ was established at the edge of the Big Dry Creek floodplain near Transect B-B’. The transect is not within the augmentation area, but is close enough to the stream channel to potentially be impacted by increased amounts of surface water flow. The vegetation along the transect is similar to that which occurs along Transect A-A’ and consists primarily of species characteristic of the dry floodplain along Upper Big Dry Creek. No augmentation species were planted or yet occur along the transect. Future data from this transect will provide a means of evaluating vegetation changes and also to evaluate the extent to which the augmentation species are increasing.

While this transect is close to Transect B-B’, there was no evidence of surface flow at this site.

Major species along this transect included western wheatgrass *(Agropyron smithii)*, blue grama *(Bouteloua gracilis)*, three-leaved sumac *(Rhus trilobata)*, and scurfpea *(Psoralea tenuiflora)*. These four species accounted for approximately 69 percent of the cover by all species. The remaining 31 percent of the cover was distributed among eight other species (Table 1). Total vegetation cover was 78 percent. Native perennial grasses accounted for most of the cover (Figure 3). In all, 31 species were observed either along the transect or within one meter of the transect line.

The most notable change along this transect was the increase in weedy species. Time of sampling was an influence in differences in species abundance and total vegetation cover.

Transect D-D’

Transect D-D’ was established along the channel of Big Dry Creek. The transect follows the section of the channel where much of the augmentation planting was conducted. Hydrologic conditions in this section of the channel have already changed. While there was no flow at the time of vegetation sampling, the soil was saturated and some standing water was present. The source of the water responsible for keeping the soils wet was from the stormwater drain near the upstream end of the transect. The vegetation along the transect is dominated by augmentation species, species of wetland plants that have naturally colonized the area and several weedy species.

Major species along this transect included woolly sedge *(Carex lanuginosa)* and Nebraska sedge *(Carex nebrascensis)*. These two species accounted for approximately 41 percent of the cover by all species. Other wetland species that were not included in the augmentation planting have become established along the transect. Common cattail *(Typha latifolia)* and willowherb *(Epilobium ciliatum)* accounted for 48 percent of the total vegetation cover. The remaining 11 percent of the cover was distributed among three other species (Table 1). Total vegetation cover was 92 percent. Augmentation species and native forbs accounted for most of the cover (Figure 4). In all, 13 species were observed either along the transect or within one meter of the transect line.

Native perennial grasses (other than augmentation species) were not encountered in the cover sampling. This reduction in abundance may be related to the changes in moisture conditions.

Transect E-E’

Transect E-E’ was established parallel to Transect D-D’ approximately 5 meters southeast of Transect D-D’. The first 10 meters of the transect are located in the augmentation area and the rest of the transect crosses the lower portion of the Upper Big Dry Creek floodplain. There was evidence of surface flow within the first 10 meters of the transect. The vegetation along the transect is dominated by native perennial grasses and forbs. Future data from this transect will provide a means of evaluating vegetation changes and also to evaluate the extent to which the augmentation species may be increasing.

In 2005, major species along this transect included willowherb *(Epilobium ciliatum)*, green needlegrass *(Stipa viridula)*, western wheatgrass *(Agropyron smithii),* prairie cordgrass *(Spartina pectinata)* and Japanese brome *(Bromus japonicus)*. These five species accounted for approximately 65 percent of the cover by all species. The remaining 35 percent of the cover was distributed among ten other species (Table 1). Total vegetation cover was 86 percent. Augmentation species *(*Prairie cordgrass - *Spartina pectinata)* had a cover value of 8 percent and accounted for 9.3 percent of the total cover (Figure 5). In all, 26 species were observed either along the transect or within one meter of the transect line.

Field observations suggest that the amount of prairie cordgrass is increasing in the augmentation area along this transect. The cover value increased from two percent in 2003 to four percent in 2004 and to 8 percent in 2005. Many new shoots of cordgrass were noted in the moist portion of the transect.

Table 1. Cover summaries (%) for established transects. Data collected on August 5, 2003, October 15, 2004 and August 7, 2005.

	TRANSECT NAME														
Species	**A-A'**			**B-B'**			**C-C'**			**D-D'**			**E-E'**		
	2003	**2004**	**2005**	**2003**	**2004**	**2005**	**2003**	**2004**	**2005**	**2003**	**2004**	**2005**	**2003**	**2004**	**2005**
Total Vegetation Cover	64	68	82	92	60	76	72	48	78	92	64	92	78	50	86
Litter and Rock Combined	36	32	18	8	40	24	28	52	22	0	32	8	22	40	14
Bare Soil	0	0	0	0	0	0	0	0	0	8	4	0	0	10	0
Total Cover	100	100	100	100	100	100	100	100	100	92	96	100	100	90	100
Litter	36	32	18	8	40	24	28	52	22	0	32	8	22	40	14
Augmentation Species															
Carex lanuginosa										28	28	34			
Carex nebrascensis				<1	18	<1				10	8	4			
Spartina pectinata				<1	<1	8						2	2	4	8
Sub-Total	0	0	0		18	8	0	0	0	38	36	40	2	4	8
Native Perennial Grasses															
Agropyron smithii	24	24	64	26	16	18	8	4	20	30			14	8	8
Agropyron trachycaulum											<1				
Aristida longiseta							<1								
Bouteloua gracilis	10	38	4	10	10		12	28	18				10	2	<1
Buchloe dactyloides	6			4	<1		14								
Carex heliophila							4	<1	2						
Hordeum jubatum					<1										
Koeleria macrantha			<1	<1		<1	<1	<1					<1		<1
Poa secunda	<1														
Sporobolus cryptandrus		<1					<1						<1	<1	
Stipa comata	4			2		<1	2		<1				2		<1
Stipa viridula	<1	2	4	<1	<1		<1	2	2	<1			6	4	10
Sub-Total	44	64	72	42	26	18	40	34	42	30	0	0	32	14	18
Introduced Perennial Grasses															
Bromus inermis	<1														
Poa pratensis				2					<1	<1			<1	2	
Sub-Total				2	0	0	0	0	<1					2	0
Native Forbs															
Artemisia ludoviciana			<1	2		<1				<1	<1	<1	2	2	<1
Aster falcatus						2			<1	<1	<1	2	2	8	4
Bahia oppositifolia	2	<1	<1	<1	<1	2	<1	<1	<1						
Cirsium undulatum							<1			<1					
Comandra umbellata							<1								
Dalea candida				<1											
Epilobium ciliatum											8	36		8	20
Erigeron strigosus					<1				<1				<1		<1
Gaura coccinea	<1			<1	<1		<1	<1	2				<1	<1	2

Table 1. (Continued) Cover summaries (%) for established transects. Data collected on August 5, 2003, October 15, 2004 and August 7, 2005.

	TRANSECT NAME														
Species	**A-A'**			**B-B'**			**C-C'**			**D-D'**			**E-E'**		
	2003	**2004**	**2005**	**2003**	**2004**	**2005**	**2003**	**2004**	**2005**	**2003**	**2004**	**2005**	**2003**	**2004**	**2005**
Native Forbs (cont'd)															
Heterotheca villosa			<1	<1									<1		
Kuhnia eupatorioides		<1													
Laythrus sp.										<1					
Liatris punctata			<1												
Lupinus parviflorus		<1		2	8				<1				2		2
Mertensia lanceolata													<1		
Oenothera villosa					<1									<1	
Onosmodium molle			<1												
Penstemon secundiflorus	<1														
Psoralea tenuiflora	2	<1	<1	<1			8	2	6	<1			10	<1	4
Ratibida columnifera			<1			<1			<1	<1			<1		
Rumex venosus							<1								
Solidago rigidus	<1	<1	<1	<1											
Sphaeralcea coccinea	6	2	<1	<1	<1	<1	6	<1	<1				<1		
Typha latifolia											4	8		<1	
Veronica anagallis-aquatica											6	<1		<1	
Vicia americana	<1	<1	<1	4		<1	<1	<1	2	<1			<1	<1	<1
Sub-Total	10	2	<1	8	8	4	14	2	10		18	46	16	18	32
Shrubs/Semi-Shrubs															
Artemisia frigida				<1	<1	<1							<1	<1	<1
Gutierrezia sarothrae	4	2	<1	<1	2	<1	2	<1	2				<1	2	<1
Rhus trilobata			4				12	10	10	<1			2	2	4
Rosa arkansana										<1	4	<1	<1	2	2
Sub-Total	4	2	4		2	<1	14	10	12		4	<1	2	6	6
Cacti and Succulents															
Opuntia polyacantha	<1	<1					<1	<1	<1						
Opuntia compressa					<1	<1			<1						
Yucca glauca											<1				
Sub-Total				0		<1			<1	0		0	0		
Undesirable Weedy Species															
Alyssum desertorum									<1						
Amaranthus retroflexus				<1					<1	<1					
Bromus japonicus			2	20	4	16	2		6	8		<1	12	4	10
Cardaria draba		<1			<1										
Carduus nutans								<1							
Centaurea diffusa		<1	2	18	2	24	<1	2	<1		<1	<1	6	2	<1
Cirsium arvense					<1						6	6		<1	2
Convolvulus arvensis	6	<1	<1	2		<1				<1			2	<1	6
Conyza canadensis			<1		<1	<1			<1		<1		<1	<1	<1
Echinochloa crus-galli										10	<1			<1	
Erodium cicutarium					<1										
Gaura parviflora				<1		2			<1					<1	2

Table 1. (Continued) Cover summaries (%) for established transects. Data collected on August 5, 2003, October 15, 2004 and August 7, 2005.

	TRANSECT NAME														
Species	**A-A'**			**B-B'**			**C-C'**			**D-D'**			**E-E'**		
	2003	**2004**	**2005**	**2003**	**2004**	**2005**	**2003**	**2004**	**2005**	**2003**	**2004**	**2005**	**2003**	**2004**	**2005**
Undesirable Weedy Species (cont'd)															
Helianthus annuus										<1					
Kochia scoparia				<1						<1					
Lactuca serriola			4	<1	<1	4	<1	<1	6	2		<1	6		2
Lepidium perfoliatum														<1	
Medicago lupulina				<1		<1				<1			<1		
Melilotus officinalis			<1	<1	<1	<1		<1	2	<1			<1	<1	
Panicum capillare				<1	<1					4					
Podospermum laciniatum							<1			<1			<1		
Polygonum aviculare					<1										
Rumex crispus															<1
Sonchus arvensis									<1						
Taraxacum officinale				<1	<1	<1	2						<1	<1	
Thlaspi arvense									<1						
Tragopogon dubius			<1	<1		<1	<1		<1	<1			<1		
Trifolium fragiferum		<1													
Verbena bracteata				<1						<1	<1				
Sub-Total	6	<1	8	40	6	46	4	2	14	24	6	6	26	6	22

CONCLUSIONS AND COMMENTS

The 2005 monitoring of Upper Big Dry Creek took place on August 7th, four years after installation of the storm drains and three years following introduction of augmentation species and initiation of the monitoring program. Since then, the total vegetation cover along all transects has been increasing. This is partly due to recovery from grazing, which ended in 2002, and also related to improving soil moisture conditions in the valley. With the recent years of adequate precipitation and the addition of storm water drainage to the valley, the near surface and surface soil moisture conditions have been elevated for the central drainage transects (B-B', D-D' and E-E'), leading to far longer periods of saturated soil conditions than historically occurred, and extended periods of anaerobic soil conditions. Even the drier transects (A-A' and C-C') which are farther from the central drainage area, seem to show the influence of increasing sub-surface soil moisture, although still maintaining aerobic soil conditions. On all transects the increasing soil moisture seemed to be an important contributing factor to the overall increase in vegetation cover.

For the native prairie grasses in the swale, the increased storm water is a mixed blessing. Continuing the trend noted in earlier reports, the pre-development native perennial grasses along the increasingly anaerobic transects in the central swale (B-B', D-D', and E-E') have almost entirely disappeared; due to silt and extended periods of soil saturation brought on by development run off. Along these wetter transects, the cover by the planted augmentation species has increased, except for transect B-B' (discussed later). In contrast, cover by historic native perennial grasses along the two drier transects (A-A' and C-C') has increased. The slightly elevated soil moisture in these areas, farther from the central drainage, has not seemed to be detrimental to the health of historic native prairie species. With slightly elevated soil moisture, the native midgrass species, western wheatgrass (*Agropyron smithii*), has grown better than the native shortgass species, blue grama (*Bouteloua gracilis*). Western wheatgrass favors moister sites. In drier years, blue grama, may be more prevalent along these two transects.

Cover values for the augmentation species increased along transects D-D' and E-E', while the native perennial grasses have shown decreases in cover. The expansion of these species indicates the ongoing successful process of establishment of the augmentation species in the more frequently saturated portions of the valley (Photos 6, 7, and 8.).

Cover by noxious weeds also increased along all transects except D-D'. (Native weedy species increased on transect D-D', see comments below.) Rapidly invading weedy species compete with other vegetation along transects and lend some confusion to the cover data. An increase in taller weedy species, along transect B-B' , obscured the shorter vegetation, leading to an apparent reduction in cover by augmentation species on this transect. The augmentation species are still present along this transect. However, the taller weeds are encountered first in the sampling procedure. In order to enhance the desirable vegetation, a weed control program was implemented in 2005. With ongoing weed control measures in place, future monitoring should provide a clearer understanding of the progress of the augmentation species in this area.

Along transect D-D' there has been an apparent decrease in cover by (augmentation species), Nebraska sedge, each year (10 percent in 2003; 8 percent in 2004; 4 percent in 2005). However, the cover by the weedy native forb, willowherb, has been increasing on this transect over the same period (0 percent, 8percent, 36 percent). As mentioned above, the taller vegetation is first encountered by the sampling method. In a few years, this weedy perennial may decline in abundance as Nebraska sedge begins to increase in abundance along this transect in a few years.


Photo 6.   Prairie cordgrass installation area April 2003.


Photo 7. Prairie cordgrass installation area in October 2004.


Photo 8. Prairie cordgrass installation area in August 2005.

The other two augmentation species in the monitoring area showed increases in cover values this year. Prairie cordgrass (*Spartina pectinata*) cover increased on transects B-B', D-D', and E-E'. Woolly sedge increased along transect D-D', the only transect where it currently is found. In other areas within the trial planting, all three of the augmentation species appear to be doing well.

The cover values for weedy species appear to be influenced by the time of sampling. In 2003 and 2005, sampling was conducted in early August. In 2004, transects were sampled in October. Weed species were encountered much more frequently and the cover values were higher for the years when the sampling was done in late summer, when the vegetation was in better condition. This was especially true for Transects B-B' and E-E'. The only transect that showed a consistent reduction in weeds was Transect D-D', where the cover by introduced weeds has decreased as the cover by augmentation species and native wetland forbs (including willowherb) has increased (discussed above). In 2005 there may also have been fewer heavy storm flows prior to sampling, which would also tend to account for generally better condition of the plant cover.

A weed management program was initiated in late summer 2005, due to increased abundance of noxious weeds. Weedy areas were mowed in preparation for herbicide application in 2006. A preliminary fall herbicide treatment of noxious weed basal rosettes was conducted mostly along the upper flanks of the valley, outside of the augmentation area. During this coming year, spring, summer and fall herbicide applications are planned for treatment of diffuse knapweed, Canada thistle, Scotch thistle, musk thistle, bull thistle and curly dock, wherever they occur throughout the valley. Special care will be taken in the actual monitoring area to assure that shrubs will not be damaged by herbicide application.

Where soil moisture is adequate but not excessive, the native shrubs (wild plum, chokecherry, and red osier dogwood) are surviving in spite of some browsing by the deer population. Wild plum appears to be the favored browse species. Improving soil moisture in the valley could support installation of native tree species, such as plains cottonwood and peach-leaved willows, along the edges of the valley floor. For best survival of establishing trees with little additional watering, deep planting methods should be used, which install the tree's root ball into the moist soil zone just above saturated soils.

SUMMARY

No downcutting has been noted in the augmentation area in spite of four years of elevated soil moisture and siltation. This year's monitoring occurred in August, and showed some greater relationship to the 2003 data, also collected in August. Results for this sampling period showed a trend toward increasing cover by augmentation species along the moister transects, correlated with decreasing cover by the historic perennial grasses. Transects in drier locations, farther from the drainage channel, exhibited an increase in native perennial grass cover. In transect D-D', which follows the center of the drainage channel closely, the perennial grasses decreased from 30 percent cover, in 2003, to 0 percent cover in 2005 while the augmentation species covered increased slightly. The total vegetation cover also increased in all transects. This may be due to more favorable moisture conditions at all sampling locations coupled with continued recovery from intensive grazing (prior to 2003).

Cover by weedy species increased along all transects, when both weedy natives and introduced weeds are considered (Table 1). This is due to maturation of these weedy species. It takes a few years after disturbances occur for fully grown weeds to develop and exert full influence. While weedy species have become more abundant this year, the transition to dominance by the more hydric augmentation species appears to be continuing. Weed control of the introduced noxious weeds and the natural attrition of the native weeds are both expected to favor of the gradual expansion of the augmentation species.

Since the 2004 sampling, some of the permanent markers had been removed. The impacted transects were relocated using a combination of GPS readings and photographs. T-posts were installed to better mark all transects and hopefully, discourage further vandalism.

CONTINUED MONITORING PROGRAM

Future monitoring is planned to continue to track the gradual changes, in the study area, for at least ten years. Regular data collection will allow for evaluation of the augmentation program and document further changes in the vegetation, during this period of relatively rapid succession. On-going monitoring data will provide a means of evaluating and refining this proactive channel protection treatment.

RESTORING FLORISTIC INTEGRITY:
MEASURING SUCCESS USING A FLORISTIC QUALITY ASSESSMENT

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ABSTRACT

A challenged faced by many natural resource managers is to understand if management activities result in degradation, enhancement, or restoration of ecological integrity of their resource. Since plants are reflective of both spatial and temporal processes, they are often used as indicators of change resulting from management or restoration activities. However, many floristic measures often used do not reflect a restoration target resembling natural communities. For example, dominance of native species does not necessarily equate to floristic integrity. Floristic integrity can be defined as the characteristic variety, abundance, and functional types of plant species which evolved with the specific set of physical, chemical, and biological interrelationships unique to various ecological systems. If we intend to restore our natural ecological systems then we need to restore the characteristic composition and structure of those assemblages. By describing high-quality examples of these ecosystems, one can delineate specific floristic attributes which define this condition and thus provide a target for restoration success. An effective and efficient way of accomplishing this is to measure the presence of conservative species which occur at a site.

High quality natural ecological systems contain both generalist and conservative species. The former are able to adapt to human-induced disturbances whereas the latter are more sensitive to habitat degradation and typically disappear from a site when the ecological processes to which they have evolved have been disrupted. Thus, the proportion of conservative plants in a plant community is strong indicator of the complexity of the flora and underlying ecological processes at a particular site and is a useful measure of floristic or ecological integrity.

The Floristic Quality Assessment (FQA) is a vegetative community index designed to assess the degree of "naturalness" of an area based on the presence of conservative species. Coefficients of conservatism (C-values) are assigned to each species in a flora and then used to indicate “naturalness”. The C-value indicates a plant’s fidelity to habitat integrity and can be thought of as the relative probability that a particular plant is indicative of ecological conditions free of human-induced disturbances. Application of the FQA entails compiling a species list for a particular study site and averaging the C-values. The site’s mean C-value is plugged into various indices which incorporate the effect of area, species richness, and non-native species on floristic quality. The resulting indices scores can be used to monitor and assess the performance of restoration, management, and protection projects and used to identify and prioritize conservation targets. Declining scores suggest that restoration or management is resulting in a negative change in floristic quality whereas increasing scores indicate a positive response.

The Colorado Natural Heritage Program has convened a panel of our region’s botanical experts to assign C-values to each species in Colorado’s flora. To date, 80 percent of the flora has been assigned a C-value. Preliminary analysis suggests that these values are able to discriminate between sites of varying degrees of human-induced disturbance and that the FQA may be an effective monitoring tool for measuring successful restoration of floristic integrity.

ALPINE ECOSYSTEM RESTORATION OF RECREATIONAL DISTURBANCES IN COLORADO'S WILDERNESS AREAS

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ABSTRACT

Colorado's 54 peaks over 14,000 ft. in elevation (14ers) are climbed by over 500,000 people annually. Unmanaged recreation on 14ers has caused trail braiding, erosion gullies, disturbance to rare alpine plants, and wide corridors of denuded vegetation. Given the slow recovery rate of alpine tundra, restoration is necessary to hasten the natural recovery process and maintain landscape esthetics. Wilderness regulations and site logistics provide unique restoration challenges. The Colorado Fourteeners Initiative (CFI), in partnership with the USFS, has been implementing alpine ecosystem restoration projects in Wilderness for over a decade. Recent efforts to address restoration needs of these disturbed areas include the use of plank walls and research to address seed viability and germinability of select alpine plants. Plank wall systems are cost efficient and an effective means of providing stabilization of fill material for recontouring efforts. Seed viability and germinability data indicate that local ecotypes are appropriate for achieving adequate vegetation cover in disturbed alpine sites. Results from germinability tests for nine alpine species from five 14ers indicate that 86% of the populations sampled had germinability over 60%.

INTRODUCTION

"In wildness is the preservation of the world." – Henry David Thoreau

If wilderness is to be preserved, society must, at a minimum, manage such lands in a manner that allows recovery, from human-induced disturbances, at a rate faster than the rate at which disturbances are being created. In the United States, the wilderness preservation system (Wilderness Act 1964) has been effective at protecting undeveloped public lands from logging, mining, roads, and other large-scale anthropogenic disturbances that are common in non-designated wilderness lands. However, wilderness lands are not free from less obvious disturbances such as fire suppression, nitrogen pollution, grazing, weed introductions, and recreation impacts. Even though ecosystem damage from 14er recreation often occurs on a relatively small scale, restoration measures are necessary to recreate original drainage patterns, halt erosion, and restore a native cover of vegetation that will persist over the long term in the harsh alpine environment.

Wilderness managers must weigh the cost of taking action versus not taking action when considering restoration plans (Cole 2000). Though large-scale restoration in wilderness areas has been interpreted as a paradox, restoration of local disturbances (e.g., recreation-related disturbances such as social trails or camp sites) can be accomplished without compromising wilderness values (Cole 2000). Of Colorado's 54 Fourteeners, 32 reside in federally designated wilderness areas. While the basic principles of ecological restoration are as appropriate on 14ers as they are on other sites, the specific treatments that are employed reflect a need to operate under several unique constraints: remote locations, lack of water available for irrigation, strenuous work environment, prohibition of mechanized equipment, and restricted use of off-site materials.

While recreational disturbances in general have been described as less severe than other types of disturbances (Chambers 1989), this is not the case on 14ers. Due to the fact that most recreation disturbances on 14ers involve fall-line social trails (i.e., trails created by users) on slopes ranging from 60-80%, these conditions require active restoration in order to provide conditions necessary for alpine ecosystem recovery to succeed. Alpine restoration efforts are also challenged by low number of suitable revegetation species and highly variable environmental conditions (Chambers 1997). Ebersole (2002) found that even after three decades, alpine plants were slow to recolonize disturbed alpine areas on Niwot Ridge, Colorado. Urbanska and Chambers (2002) suggest that some alpine disturbances may require more than 100 years to fully recover. Such challenges require creative solutions to conventional restoration problems. This paper describes several methods that have proven effective in restoring recreation-related disturbances on wilderness 14ers and presents results from recent seed germinability research for select alpine species. A summary of previous findings on 14er restoration methods is provided, and emphasis is given to aspects of 14er restoration not previously reported in the literature.

SUMMARY OF STABILIZATION TREATMENTS ON 14ERS

Check Dams

On steep (e.g., greater than 20 percent) denuded slopes, revegetation efforts are unlikely to succeed without adequate slope stabilization. Rock or log check dams are appropriate methods for achieving short-term stabilization for approximately 80 percent of closed social trails on 14ers. A check dam is a physical structure placed perpendicular to the flow of surface water in order to reduce water velocity, halt erosion, and encourage water to infiltrate into the ground rather than flow down the trail surface. Check dams are most effective for stabilizing social trails that have not eroded more than 12 inches deep. In deeper gullies, structures that provide greater structural stability (e.g., retaining walls, monowalls, or log terraces) are necessary to provide the necessary stabilization. It is beyond the scope of this paper to include descriptions and installation methods for retaining walls, monowalls, or log terraces. Following is a brief description of installation methods for rock check dams on 14ers.

Check Dam Spacing

Engineering specifications require that check dams be installed at a frequency such that the highest point on the lower check dam is equal in elevation to the base of the check dam immediately above it (Hesselbarth and Vachowski 2004). On the typical slopes encountered on social trails of 14ers, this would require installing check dams every 18 to 24 inches in many places. While this would provide excellent stabilization as a stand-alone treatment, the incorporation of seeding or plug transplants between check dams allows for installation of check dams at greater distances from one another.

Besides the importance of slope and revegetation treatments used, soil type, microtopography, and trail orientation affect check dam spacing. Soils that experience a low infiltration rate—compacted soils, soils with high clay content, and soils with low organic matter—often require check dams to be installed at a greater frequency. Microtopography influences slope angles at a small scale, determines where late-melting snow fields and other natural drainages may affect drainage along a social trail, and influences slope aspect along the social trail. Though water may not be present during project planning or implementation, late melting snowfields and other natural drainage areas can be determined by vegetation type, microtopography, and relative abundance of lichen on rocks (i.e., lichen is typically more abundant on rocks that experience a longer snow-free period during the growing season). In seasonally wet areas, check dam frequency should be increased. Where an eroded trail is oriented across the slope, rather than along the fall line, interception of subsurface water can significantly increase the amount of water received by the trail. Under such conditions, check dams should be installed at a greater frequency.

Check Dam Construction

The term “check dam” should not be confused with “check step”. Check dams, for use in restoring closed trails, are constructed differently than check steps, which are used in trail construction. Because hikers will not be stepping on restoration check dams, they do not need to be as structurally sound as check steps. Unlike check steps, check dams should not be backfilled with mineral soil. Rock check dams are constructed so that the surface of the rocks are equal to the level of the tundra, or slightly below it, with a small drainage gap in the center of the structure. This construction technique allows for maximum natural backfill behind the check dam—as the check dam collects sediment naturally over time—and prevents sediment from spilling out around the sides of the check dam. Excessive erosion around the sides of the check dam may undermine the structure and deposit damaging sediment on low-growing tundra plants. To ensure adequate stability, rocks should be installed so that at least 1/3 of the rock is buried below the surface of the trail, and the side rocks are anchored at least 4 inches into the banks of the trail. Other construction details, such as material dimensions and check dam diagrams, are provided by Hesselbarth and Vachowski (2004).

Other Considerations

Rock check dams are simple to construct and quick to install. However, they should be used only in conjunction with revegetation treatments to ensure long-term stabilization of steep slopes. Structures often fail due to shallow anchoring of rocks, lack of anchoring into the trail banks, or a combination of the two. When rock materials are not available, logs, excelsior wattles, and sand bags have been used to achieve short-term stabilization on 14ers. When conditions allow, willow wattles may also be used to provide short-term and long-term stabilization as a living structure.

Willow Wattles

Wattle Construction

Willow wattles can be described as a living check dam. Spacing of wattles follows the same guidelines as spacing of check dams. However, construction and installation methods for willow wattles require elaboration. Wattles are created by bundling and weaving thin willow stems (less than ½ inch thick) into a cylinder that measures at least 8 inches in diameter and 8 inches wider than the trail to be stabilized. Apical buds and the majority of lateral shoots should be removed prior to bundling. Leaving some long lateral shoots near the upper 1/3 of the stem will help create a stronger weave. Wattles are further secured by binding the cylinder with twine.

Willow Wattle Installation

Wattles are installed in a constructed trench (1/3 the depth of the diameter of the wattle) and anchored in place using willow stakes. In a trail setting, willow wattles should be installed so that the ends of the wattle are anchored at least 4 inches into each side of the trail bank. Installed wattles should be packed with soil, between the bundled stems, to create a mound and help retain moisture around the stems. Willow stakes, as described below, should be inserted every 8-10 inches through the center of the bundle, to a depth that ensures at least 6 inches of the stake is in contact with seasonally moist soil. Other construction details, such as design criteria and detailed diagrams, are provided by Lake and Dickerson (2003).

Considerations

This time consuming method should be limited to wet areas that will support willow growth, and sites that lack adequate rock or logs for constructing check dams. Selection of healthy willow stems is essential to success, and efforts should be made to avoid harvesting more than 50 percent of the stems from any single willow plant. Survival rates of willow wattles on 14ers have been low due to inadequate soil moisture, trampling from continued use of the closed trails, and incorrect installation. Even where adequate soil moisture exists, incorrect installation will most likely result in failure to establish living willows. Existing willow wattles on 14ers have been constructed from stems of *Salix brachycarpa* (short-fruit willow) and *S. planifolia* [planeleaf willow; plant nomenclature, with the exception of common names, follows USDA, NRCS (2006)]. Future studies are needed to determine if species selection is an important variable in determining willow wattle survival rates.

SUMMARY OF REVEGETATION TREATMENTS ON 14ERS

Willow Stakes

Willow stakes are useful for establishing willow plants vegetatively across a large area. Once established, willows can provide effective erosion control as well as provide a physical barrier to hikers who may otherwise be tempted to follow closed social trails. The following procedures have proven successful at elevations above 11,000 feet on 14ers.

Creating Willow Stakes

As with willow wattles, selecting healthy willow stems is key to success. Cutting stakes from the base of stems containing many adventitious buds also improves success. Stakes should be at least 1 inch thick and at least 12 inches long, but long enough so that the bottom 6 inches of the stake is in contact with seasonally moist soils and the top 4 to 6 inches of the stake remains above ground. The bottom of the stake should be cut at an angle to indicate the rooting end of the stake and to facilitate installation. All lateral shoots should be removed and the top of the stake should be cut flat to facilitate pounding. Stakes should be soaked in water for 3-7 days, in the shade, prior to installation to encourage swelling of root primordia.

Installation

Willow stakes should be used only in areas with high seasonal soil moisture to ensure successful establishment. In hard or rocky soils—frequently encountered on 14ers—a pilot hole should be created to facilitate installation. This can be accomplished by pounding rebar into the ground or using a pick mattock. If the top of the stake becomes split or mashed from pounding, it should be cut off clean to prevent excessive drying of the stem. Willow stakes should be spaced 8-12 inches apart on steep slopes, were erosion control is needed. When erosion control is not a concern, willow stakes can be spaced farther apart to meet other restoration goals.

Other Considerations

Preliminary evidence on Mt. Massive suggest a survival rate of 80 percent for *Salix drummondiana* (blue willow) after two growing seasons at 11,200 feet elevation in seasonally wet, gravelly soils (Giordanengo 2006). However, proper installation, site selection, and species selection affect survival rates. On marginally dry soils at the same site, *S. drummondiana* experienced a survival rate below 10 percent. Additional studies should be conducted to determine survival rates of willow stakes harvested from different subalpine species.

Turf Transplants

Background

Alpine turf transplants have a long history of use in Colorado. Buckner and Marr (1988) investigated the success of machine-harvested turf transplants used to restore pipeline disturbances on Rollins Pass in the late 1960's. Their findings indicated that turf transplants were a highly successful restoration treatment for alpine ecosystems. More recently, turf transplants were used to restore roadside disturbances along the Guanella Pass National Scenic Byway (Collinson 2006). In this project, the depth of turf ranged from 6 to 9 inches and stored material was watered daily to prevent desiccation of roots. Turf transplants have also been used to restore recreation-related disturbances on 14ers (Hesse 2000; Conlin and Ebersole 2001). However, 3 years after transplanting turf on Mt. Belford, results showed a sharp decline in forb species (28 percent), while grass cover increased by 25 percent (Ebersole et al. 2004). Though 38 of the 49 species transplanted showed no difference in absolute cover between control and treatment plots, *Geum rossii* (alpine avens)—a dominant forb in many alpine plant communities—decreased significantly. These results agree with results from Marr et al. (1974) who reported that *G. rossii* decreased significantly in turf transplants.

When possible, turf transplants are placed edge to edge to obtain immediate complete cover in the disturbed area. When this is not possible, turf transplants can be spaced apart from each other, creating safe sites for establishment of seedlings. The importance of safe sites in restoration is well addressed by Urbanska (1997). The concept of safe sites was originally described in the early 1960's by Harper et al. (1961) and Harper et al. (1965). These researchers describe a safe site as an environment that is favorable to seed germination and establishment. In the alpine, we hypothesize that safe sites experience less wind, increased soil moisture, and greater accumulation of surface organic matter. On 14ers, safe sites between turf transplants and other transplants have shown greater establishment of seedlings from the seed bank or seed rain from adjacent undisturbed tundra. On Mt. Belford, Colorado, Ebersole et al. (2002) documented seedling establishment at 0, 2, and 4 inches from the edge of turf transplants. Seedlings were significantly more abundant near turf transplants than in turf interspaces. They recommended spacing turf transplants 8 to 12 inches apart to create effective safe sites.

Installation

In trail restoration, this treatment involves salvaging blocks of tundra during new trail construction and installing them directly on disturbed social trails. Where immediate visual closure is not the goal, but revegetation is necessary, turf transplants are spaced 8 to 12 inches apart, with all roots being buried in the disturbed trail and any exposed edges covered with native topsoil. Rocks and boulders may also be used to fill voids and support transplants on steep slopes. Conlin and Ebersole (2001) reported high success with turf transplants on Humboldt Peak, where turf was excavated to a depth of 6 inches during new trail construction. Results after 1 yr showed that 72 percent of species present in turf transplants did not decrease in absolute cover.

Other Considerations

Since artificial irrigation is not available on project sites, turf transplants are not stored longer than 2 days before being transplanted. When stored longer than one day, blocks of turf are placed edge to edge and covered with tarps to prevent desiccation. Where steep side-slopes are encountered during new trail construction it is possible to salvage turf transplants 6-8 inches deep. On less steep slopes (e.g., 20 percent or less), standards for construction of trail tread do not allow harvesting of turf transplants deeper than 5 inches. In such cases, turf transplants can be expected to experience high mortality rates and lower species richness in surviving transplants. A viable option in this case is to harvest plug transplants (described below) sporadically within the new trail corridor to the desired depth. The holes left in the trail can be filled with rock and soil, thus helping to define the new trail.

Plug Transplants

Background

When turf transplants are not available, plug transplants have been shown to be a viable option in the alpine. While turf transplants may show lower survival of forbs than grasses, preliminary evidence by Ebersole et al. (2002) on Mt. Harvard—a moist site—showed that alpine forbs and grasses can survive very well in plug transplants after one year. On Niwot Ridge, May et al. (1982) also reported high transplant survival for *Deschampsia cespitosa* (98 percent), *Kobresia myosuroides* (83 percent), and *Geum rossii* (79 percent). In that study however, transplants were watered for several weeks. Payson (2002) also showed high transplant success for *D. cespitosa* (tufted hairgrass) transplants (100 percent), but low transplant success for *G. rossii* (25 percent) on the Beartooth Plateau, Wyoming, USA. In that study, however, container plants were grown from native plant seed collected on site.

Installation Considerations

Installation of plug transplants is very similar to transplanting container stock in similar conditions, with a few important differences:

1. Harvesting native plants on site usually results in a plug with an incomplete root system.
2. A single plug often contains several species.
3. Plant species in the plugs are adapted to local soils and environmental conditions.
4. Nursery stock may be root bound and have a low root to shoot ratio.

For these reasons, and given the lack of water usually available on 14er restoration sites, there are several considerations for using alpine plug transplants. Plugs should be installed only in areas that contain naturally high soil moisture (e.g., moist alpine meadows) or early in the growing season when soil moisture is adequate to ensure plug survival. Even though the alpine zone of Colorado receives higher precipitation than most other zones of Colorado, most of this precipitation falls as snow, which is redistributed across the landscape, creating a mosaic of xeric and mesic conditions. During the summer months, rainfall is sporadic, spatially variable, and often inadequate to replenish soil moisture in disturbed sites. This lack of predictable precipitation, coupled with high evaporation potential, results in conditions that can result in low plug survival rates in dry exposed sites.

Other Considerations

While individual plugs may experience high survival rates after 1 year, they do not appear to spread vegetatively into adjacent disturbed areas on Mt. Harvard. On Quandary Pk., plugs have not appeared to spread into disturbed sites three years after transplanting (Giordanengo 2006). It is currently unknown what factors encourage or inhibit vegetative spread of alpine plugs in disturbed sites on 14ers. Efforts to harvest plugs containing rhizomatous species [e.g., *Achillea millefolium* (western yarrow), *Senecio atratus* (black-tipped groundsel), *Elymus trachycaulus* (slender wheatgrass), and some *Carex* sp. (sedge)] may help to increase the rate of vegetative colonization into disturbed areas from the transplants, though this hypothesis has not been tested.

Long-term studies to document alpine plug transplant survival over time are lacking. Based on the varying results from May et al. (1982), Ebersole et al. (2002), and Payson (2002), survival of alpine plug transplants is likely influenced by cultural practices (e.g., irrigation and soil amendments), transplant source (e.g., container stock or plug), plant species, and site conditions. While some alpine grasses (e.g., *D. cespitosa*) have been shown to survive transplanting better than alpine forbs, high survival rates for many alpine forbs should encourage practitioners to use a variety of forb and grass transplants in restoration projects.

Native Mulch

Background

Mulch is widely recognized as a component of successful seeding projects and alpine restoration on 14ers is no exception. Results reported by Ebersole et al. (2002) show a dramatic difference among mulch treatments on seedling establishment on Mt. Harvard. Abundance of monocot seedlings was 6 times greater, and dicot seedlings 5 times greater, in the “seed+erosion matting” treatment than in the “seed alone” treatment. By helping to reduce soil erosion, slow soil evaporation, and by sheltering small seedlings from harsh winds, intense UV radiation, and heat extremes, mulch improves the conditions that are necessary for germination of many alpine plant species. Brink (1964) also showed that needle ice, which can uproot small seedlings, is less likely to form in areas protected by mulch. However, the logistics associated with hauling off-site materials to 14er work sites often prohibits use of commercially available erosion matting. Using native mulch is one option that has proven successful on 14er seeding projects.

General Guidelines and Considerations

Native mulch involves hand-clipping of stems and leaves of tundra plants in the fall when vegetation is dormant and some seeds remain on tundra plants. Clipped vegetation is bagged and spread over seeded areas to an unpacked depth of 1.5 to 2 inches. The mulch is secured with photo-degradable plastic mesh (stapled every 6-8 inches in the center and every 4 inches around the border). This method is efficient only where drainages, late-melting snow fields, or other moist meadows—areas where significant above-ground biomass often occurs—are readily accessible. Due to the compaction of loose native mulch over the winter, and decomposition of leaves and stems, final mulch depth may be reduced by 50-60 percent in the first year. Although using native mulch saves time associated with hauling off-site materials to the work site, and saves materials costs, native mulch requires more labor during application due to the time required to harvest mulch. While anecdotal evidence show this method to be highly effective on Mt. Massive (Giordanengo 2006), further studies are needed to determine the range of conditions under which this method is effective on 14ers.

SEED GERMINABILITY ON FOURTEENERS

Background

In remote wilderness areas, using machinery and non-native species for seeding are not options since management goals require use of local native ecotypes and prohibit the use of motorized equipment. Due to these limitations, seeding is often a high cost method of revegetation. However, by involving volunteers in the process, and restricting this method to only the most critical areas, seeding with native ecotypes can be a successful means of revegetating small recreation disturbances on high elevation sites (>11,500 feet in elevation).

Because most revegetation on 14er social trails is accomplished using plug transplants—using mostly grass species and a limited number of forbs—seeding of native ecotypes can increase plant diversity in restored areas. In addition to the ecological benefits of high diversity, establishing a diverse plant community also meets wilderness goals to maintain landscape esthetics. By establishing a diversity of native species the restored site is more likely to visually match the reference plant community, thereby masking the linear feature of trails.

Since native species are often not available commercially, and those available may not be specifically adapted to the microclimate being restored (Chambers 1989), on-site collection is the preferred method of generating seed stock. Urbanska (2000) reported that viable seeds collected near the restoration area from a comparable elevation and soil type are often the most successful in restoration. Since there is a wide range of alpine species that have been shown to colonize severe alpine disturbances from seed in Colorado (Harrington 1946; Ebersole 2002; Ebersole et al. 2004), efforts should be made to collect seed from those species that are common in the adjacent undisturbed areas near social trails. However, knowledge of seed viability and germinability of these plants is important to determine adequate seeding rates, or if establishment by seeding is practical for selected alpine species.

Chambers (1989) observed a high level of variation among both species and years in the viability of alpine seeds. She hypothesized that this variability could be due to many factors including: length of growing season, inconsistent precipitation, low temperatures, and varying life histories of the species studied. Chambers (1989) also reported that variation in viability among species could also be affected by the fact that all species were collected on the same day, even though different species' seeds mature at different times. Her findings agree with Young (1986), who reported that seed maturity is an important factor affecting seed viability. He recognizes that collection of immature seeds often results in lower seed viability.

Knowledge of spatial variation of alpine seed viability along the Rocky Mountains of Colorado is lacking. Increased understanding of the relationships between environmental conditions and seed viability may help practitioners select species and locations more likely to produce viable seeds. For the purposes of this paper, seed germinability is used to describe the percent of viable seeds that germinate under laboratory conditions (described below). This section of the paper highlights preliminary results of a seed germinability study conducted on a broad spatial scale in the Colorado alpine.

Methods

In 2005, staff and trained volunteers of the Colorado Fourteeners Initiative collected seeds from several plant species from various alpine habitats on six 14ers across four mountain ranges in Colorado (Table 1). Seeds were dried, stored for one month, and tested for germinability (Auchincloss 2006). Tests for immediate germinability (i.e., without stratification) were performed with 50 surface-sterilized seeds of each species. Samples were randomly divided into two groups of 25 and placed evenly over a 0.5 percent agarose gel in two Petri dishes sealed with parafilm. The two dishes for each collection were placed into separate growth chambers. The chambers received 12 hours dark and 12 hours light, incandescent and fluorescent, and temperatures of 10^{o} C (dark period) and 15^{o} C (light period). Germination, defined as the presence of a radical visible to the naked eye, was recorded for 3 months.

A test of equality of proportions was performed to test the null hypothesis that the groups tested do not show different proportions germinability. If variation existed among groups, follow-up tests of all possible pairs were done.

Table 1. Collection dates and locations (Auchincloss 2006).

Mountain	Collection Date	Elevation	Slope	Aspect	GPS Coordinates
Quandary Pk.	9/3/2005	3,444 m	25 to 30º	180 to 185º	not available
	9/4/2005	3,566 m	0 to 20º	175 to 185º	not available
Mt. Tabegauche	9/10/2005	3,414 m	0 to 25º	200 to 240º	38 º36.900 N; 106 º18.02 W
Mt Evans	9/17/2005	3932 m	0 to 10º	180 to 225º	36 º36.016 N; 105 º38.382 W
	9/18/2005	3901 m	25 to 30º	160 to 190º	39 º37.117 N; 105 º37.711 W
Wetterhorn Pk.	9/24/2005	3,608 m	10 to 16º	238 to 260º	38 º02.897 N; 107 º29.591 W
Sunlight Pk, Mt. Eolus, and Windom Pk. (SEW)	9/12/2005	3,444 m	33 to 39º	136º	37 º 36.730 N; 107 º 36.651 º W
	9/12/2005	3,730 m	19 to 37º	112 to 172º	37 º37.185 N; 107 º36.617 W
Mt. Massive	10/1/2005	3,414 m	10 to 25º	200 to 228º	39 º10.081 N; 106 º28.720 W
	10/1/2005	3,700 m	25º	175 to 190º	39 º10.393 N; 106 º28.538 W

Results

Seed germinability varied significantly among populations of most species. No mountain produced consistently higher seed germinability than any other mountain. Germinability, above 60 percent for 86 percent of the populations tested (Figure 1), was relatively high for native ecotypes. Only two species, *Trisetum spicatum* (spike trisetum) and *E. trachycaulus* (Figure 1), failed to show statistical differences in germinability among collection sites. This may indicate lack of genetic variability among populations of these species or that seeds from these populations were collected at a similar stage of development on each site. Germinability of *D. cespitosa* (Figure 1d) and *Phleum alpinum* (alpine timothy, Figure 1e) was much lower for seeds collected in an earlier stage of development (soft dough and milky-soft dough, respectively). This result agrees with Young (1986), who reported that collection of immature seeds often results in low viability. Significant differences also existed for *Heterotheca pumila* (dwarf golden aster, Figure 1g) and *Festuca thurberi* (Thurber fescue, Figure 1h) between Tabeguache Pk., Quandary Pk., and SEW (Sunlight Pk., Mt. Eolus, and Windom Pk.). The decreased germinability on Tabeguache peak, for both species, may be due to the apparent low nutrient availability resulting from a severe fire event in the 1980's followed by continual erosion to the present day.

In addition to the germination data in Figure 1, seed viability was tested for *Phacelia sericea* (purple fringe). *P. sericea* has been reported as a common colonizer of disturbed areas (Harrington, 1946) in the alpine. Results form two collections on SEW in 2005 revealed seed viability between 90 and 93 percent. When seeded on Mt. Massive restoration sites, evidence suggested that *P. sericea* comprised approximately 60 percent absolute cover and 80% relative cover in the first growing season after being seeded (Giordanengo 2006).

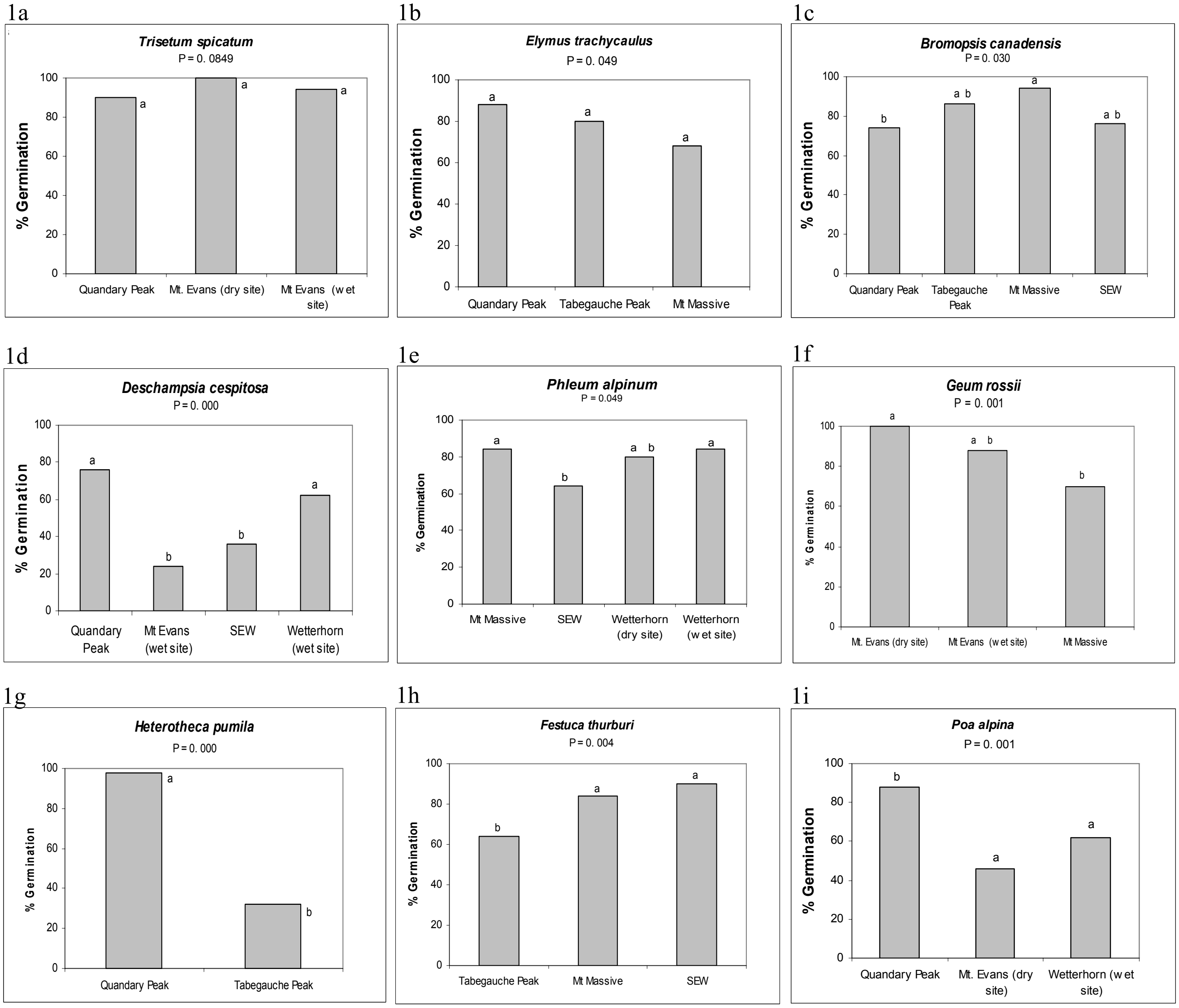

Figure 1. Percent germination (of 50 seeds) of alpine species collected on six 14,000 ft mountains across four mountain ranges in Colorado. The significance of a comparison of multiple proportions is given as *P*. Lower case letters indicate homogenous subsets. SEW = Sunlight Pk., Mt. Eolus, and Windom Pk. Graphs from Auchincloss (2006).

Discussion

Given the relatively high seed germinability of select alpine species from several mountain ranges, seeding with native ecotypes appears to be a viable option for future alpine restoration efforts. While native seed viability is often reported as highly variable on a temporal and spatial scale, viability may be more predictable for certain species than others, and can be more successful when seeds are collected in the most developed stage (e.g., hard dough). For alpine species particularly, precipitation—in previous and current years—can play a significant role in seed viability (Chambers 1989). Snow pack for the 2004-2005 winter season was above average in the mountain ranges of this study (National Water and Climate Center, 2005), though the tundra on these ranges may still be recovering from a severe drought that ended in 2004. Accurate climate data is not available for any of the sites in this study, making it impossible to determine to what degree growing season precipitation affected these results.

Differences in genetics, and the amount or resources available within the plant to support the fruit (Fenner 1985), are also important factors in determining seed viability. Though none of these factors were directly measured in this study, current year precipitation, precipitation patterns over the previous three years, genetic differences among plant populations, and available resources likely influenced germinability results. In addition, for some species, stratification of seeds may be necessary to achieve adequate germination rates. Germination tests after stratification are currently in progress, though these results indicate high germination rates for tested species can be achieved without stratification. These results are also supported by Bliss (1971) who concluded that seed dormancy characterizes only some tundra species.

This study provides a snapshot for one year of seed germinability across a large spatial scale for several species of interest to restoration ecologists. High germinability for most species tested suggest that inherent germinability may be adequate, and is not a barrier for practitioners interested in using local native ecotypes for restoration. When possible, seeds should be collected during times when there is a high probability of producing viable seeds. While *D. cespitosa* is recognized as a common colonizer of disturbed areas, high variability in germinability among mountains suggest that seed collection efforts for this species may not be consistently productive. Extreme alterations of soil fertility, such as was the case on Tabeguache Pk., also affects viability. Avoiding seed collection efforts in such areas is advisable based on preliminary results for *H. pumila* and *F. thurberi*.

Little focus has been given to the use of *P. sericea* in alpine restoration efforts on 14ers or elsewhere. Based on our limited results, and results from Harrington (1946), this species may be a desirable forb species for restoration. High germinability and quick establishment on disturbed sites may allow this species to coexist with grasses and increase the structural and biological diversity in restored sites. However, there are several forb species that have been shown to colonize disturbed areas by seed rain, including *G. rossii*. Based on our preliminary results, *G. rossii* appears to have a consistently high germinability (70-100 percent) among sites. This is important, since *G. rossii* is such an important component of many alpine plant communities of Colorado (May and Webbe, 1982) and should be re-established on disturbed sites where it is often the dominant plant. While the reference community should ultimately guide the seed mix used in revegetation efforts, the steep slopes often encountered on 14er restoration sites may warrant a greater proportion of grasses to more effectively control erosion. While many grasses are widely reported as providing better erosion control than forbs, previous research has not evaluated the erosion control potential of native grasses and forbs in the Colorado alpine. By using a diversity of species with different growth forms, adequate erosion control may be possible while developing a plant community that more closely resembles the reference plant community.

RECONTOURING WITH PLANK WALLS

Recontouring the landscape is required for many restoration projects (e.g., mine land restoration) and is often a goal of land managers. The most obvious challenges to recontouring erosion gullies on 14ers include the difficult work conditions and lack of materials available on site for backfilling gullies. The use of a plank wall system (Leisy, 2004) helps to address some of the constraints by reducing construction times and effort to build retaining structures necessary to support backfill. Typically, rock retaining walls and log terraces are used to support fill material and constructed trail tread on 14er trail and restoration projects. In areas where sufficient rock material is lacking, plank walls are a viable option for recontouring erosion gullies.

Figure 2. Front view of plank walls used to stabilize fill material for recontouring gullies. Prior to backfilling and revegetation on Mt. Massive, 2004.

Plank walls were pioneered by Leisy (2004) in Washington State to recontour erosion gullies associated with social trails. They were first used in Colorado by The Colorado Fourteeners Initiative in 2004 on Mt. Massive. Materials include 1x6 inch cedar planks and 2x2 inch cedar posts. Posts are sharpened on one end and are installed at 18 inch intervals. Posts are pounded into the ground to a depth that, at a minimum, equals the depth of the gully. The appropriate depth will also vary depending on the soil structure (i.e., in loose soils, posts should be inserted deeper). Planks are installed edge to edge on the uphill side of the stakes (Figure 2), and extend at least 6 inches into the banks of the gully. The bottom plank is buried half way in the ground and the top plank is installed at or just below the original contour. Planks are nailed to the stakes to provide short-term structure for the fill material.

At the foundation of each system of plank walls lies a rock retaining wall, which is connected to the first plank wall using rock rubble or a junk wall (not shown in Figure 2). This provides additional stability for the plank wall system. On a 30% slope, plank walls are spaced at a distance of 5 feet from one another, with only 3 feet between the retaining wall and the first plank wall. On slopes greater than 60%, plank walls are spaced no greater than 3 feet apart in order to facilitate backfilling the slope at the appropriate angle to regain the original contour. On slopes greater than 60 percent, a system of up to four plank walls was used in conjunction with each retaining wall foundation on Mt. Massive.

Once constructed, the plank walls are backfilled with rocks and mineral soil, allowing 8 inches of space at the surface for native topsoil. The recontoured gully is revegetated using plug transplants, turf transplants, or locally collected seed and native mulch. Once backfilled and revegetated, the plank walls should be completely buried and not visible to hikers. On Mt. Massive, this method has been highly effective at recontouring erosion gullies up to two feet deep and 12 feet long on slopes between 60 and 80%. The plank walls are not designed to provide stabilization of fill material beyond 10 years, after which time adequate vegetation must be established to provide long-term stabilization. On Mt. Massive, willow stakes measuring up to three feet long were incorporated into the rubble wall, and in the backfill, to create a bioengineering structure to provide better long-term stabilization of this moist site.

Because of the use of off-site materials, the efficiency of the plank wall system is reduced the farther the work site is from the trailhead. Where long distances (greater than 1.5 miles) are encountered between the trailhead and worksite, hauling times for materials make this an unattractive option for gully restoration. Land manager approval of lumber is a prerequisite to using plank walls in wilderness areas. Plank wall materials are currently packed into work sites by staff and volunteers using frame packs. Preliminary results indicate that this method is more time efficient than constructing rock or log retaining walls for recontouring slopes. In areas that lack the necessary native constructional materials on site, plank walls may be the only option available for stabilizing fill material. While the specifications above appear to work on dry to moist slopes, constructing plank walls in wet areas may require drilling one inch drainage holes every 6 inches in planks to allow for natural drainage patterns to prevail. Research is needed to develop more detailed specifications and applications for the plank wall system and to further evaluate the success of installations on Mt. Massive (completed in 2005) and Pyramid Pk (completed in 2006).

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MECHANICAL REVEGETATION IN WILDERNESS: CONFLICT OR SUPPORT OF DESIGNATED WILDERNESS?

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ABSTRACT

There are many disturbed lands in designated Wilderness as the result of mining, recreation, grazing, and other activities. The Wilderness Act requires that Wilderness areas remain undeveloped, offer solitude, and be untrammeled or free from modern human control and manipulation. It also requires that the area remain natural and free from the effects of modern civilization. Do these requirements prohibit or constrain mechanical revegetation of disturbed sites? What is the Minimum Requirements Decision Process and can it be used to allow mechanical activities and direct manipulation of the environment? Does "natural" mean leaving the area alone to recover on its own, or rehabilitating it so it recovers back to natural conditions sooner?

INTRODUCTION

Restoration of disturbed lands in designated Wilderness seems on its face value to be a positive action for speeding the healing of land to more natural conditions. However, the restoration activities themselves can disturb wilderness character. Accordingly, the question must be addressed as to whether mechanical restoration and revegetation activities conflict with or support Wilderness character? This quickly becomes a philosophical, legal, and bureaucratic issue that you need to be fully aware of when planning such restoration activities in Wilderness.

Some individuals philosophically advocate a hands-off approach to Wilderness management. In other words, leave the lands alone so natural process can function as they will in a wild state. However, there are many disturbed lands in Wilderness from mining, recreation, grazing, and other human caused activities. In some cases the effects of these disturbances are so long-term that restoration through natural processes will only be realized on geologic time scales. Restoring such lands to speed their recovery may bring them back to natural conditions much more rapidly then leaving them alone.

The guidance in the Wilderness Act is complicated. *Section 2 (c) states:* "A wilderness, in contrast with those areas where man and his own works dominate the landscape, is hereby recognized as an area where the earth and its **community of life are untrammeled by man** …. An area of wilderness is further defined to mean …land **retaining its primeval character and influence**, without permanent improvements…, which is protected and managed so as to preserve its natural conditions and which …generally **appears to have been affected primarily by the forces of nature**, **with the imprint of man's work substantially unnoticeable**..."

To summarize, the Wilderness Act calls for its lands and waters to be managed to preserve or allow:

1. **Natural Conditions -** "land retaining its primeval character and influence" and "protected and managed so as to preserve its natural conditions"
2. **Undeveloped Quality** – "without permanent improvements or human habitation", "with the imprint of man's work substantially unnoticeable," and "where man himself is a visitor who does not remain."

3. **Solitude/Primitive Recreation** – "outstanding opportunities for solitude or a primitive and unconfined type of recreation"
4. **Untrammeled by Human Actions** – "an area where the earth and its community of life are untrammeled by man" and "generally appears to have been affected primarily by the forces of nature."
5. **Wilderness Character –** "Except as otherwise provided in this Act… to preserve its wilderness character."
6. **Public Purposes** – "Except as otherwise provided in this Act… devoted to the public purposes of recreational, scenic, scientific, educational, conservation, and historical use."

Section 4 (c) adds additional twists: "Except as specifically provided for in this Act,... there shall be no permanent road within any wilderness area designated by this Act and, **except as necessary to meet minimum requirements for the administration of the area for the purpose of this Act**, ...there shall be no temporary road, no use of motor vehicles, motorized equipment or motorboats, no landing of aircraft, no other form of mechanical transport, and no structure or installation within any such area."

It is Section 4 (c) that has led the federal agencies who manage Wilderness to develop the Minimum Requirements Decision Process.

> Minimum Requirements is a documented process the agencies that manage Wilderness will use for the determination of the appropriateness of all actions affecting wilderness that involve any of the Wilderness Act 4(c) prohibitions. The minimum requirement concept is to be applied as a two-step process that documents:
>
> (1) A determination as to whether or not a proposed management action is necessary to meet minimum requirements for administration of the area for the purpose of the Wilderness Act and if the benefits of the proposed action should outweigh the potential impacts to wilderness resources, character and purposes in a manner so as to leave the area unimpaired for future use and enjoyment as wilderness.
>
> (2) If the action is appropriate and necessary in wilderness, the selection of the management activity (method(s) or tool(s)) that causes the least amount of impact to the social, biological, and physical resources of Wilderness.
>
> Minimum Activity (often referred to as "minimum tool") means a use, method or tool, determined to be necessary to accomplish an essential task, which makes use of the least intrusive tool, equipment, device, force, regulation, or practice that will achieve the wilderness management objective. This is not the same as the terms "primitive" or "traditional" tools, which refer to the actual equipment or methods that make use of the simplest available technology (i.e., hand tools).

DISCUSSION

In the wilderness community recently some have contrasted "natural" with "wild." "Wild" precludes intentional human intervention. These authors emphasize leaving nature alone to manage itself. Others argue that the pervasive and insidious magnitude of human activity has largely rendered the distinction between "wild" and "natural" moot. This is particularly true in many of the small, eastern lands Congress has set aside as designated Wilderness. There is, for example, very little wild about Cumberland Island Wilderness on Cumberland Island National Seashore, which includes roads, motor vehicles, residences, utility corridors, many introduced species, and several key species extirpated. They may require urgent intervention and long-term maintenance simply to preserve what remains of their original native biodiversity, and sometimes what remains is quite irreplaceable. To put it another way, a case can be made that their value as managed reserves of biodiversity exceeds their value as "wilderness." (National Wilderness Steering Committee 2004).

Some human-induced impacts on Wilderness that seem to be affecting the naturalness of the areas will require societal action to correct. The effects of global warming are rapidly becoming evident, even in our most remote Wildernesses. The glaciers in Glacier National Park are expected to disappear in the next 25 years. A variety of plant and animal species are migrating up in altitude as warming affects Yosemite National Park. Air pollution at Rocky Mountain National Park from urban and agricultural areas on the plains to the east are contributing sulfur and nitrogen compounds that are altering soil and water acidity and acting as fertilizers. Studies are continuing at all three parks to monitor the effects of these human-induced changes to the naturalness in these Wilderness areas.

Then there are the human-induced changes in Wilderness that could be feasibly addressed by restoration and revegetation activities within the Wilderness. The Minimum Requirements Decision Process then comes into play. Social trails can be blocked with down trees and limbs, or cactus plantings. These actions rarely require anything more than hand tools and hard work.

However, soils may be compacted to the degree that tilling is needed. If it is determined that the action is needed to manage the area as Wilderness, the second question becomes critical of HOW to accomplish the action with minimum impacts on Wilderness character, values, and resources. Can the tilling of compacted soils be accomplished with hand tools or would motorized rototillers have less impact?

Miles of trails at Rocky Mountain National Park had eroded so severely that huge quantities of fill were required to restore them. Teams of pack stock could be used to haul in the fill, or helicopters could do the work. Non-motorized pack stock trains could work all summer, with their hoof impacts degrading the miles of trail they would have to traverse just to reach the trails where they would dump the fill. Additionally, they would directly affect thousands of visitors along the way and with their horse manure left on the trails, and the manure would affect water quality. The alternative would be to use helicopters for a few days to haul the fill material in the early fall when visitation was relatively low. This is a clear example to show that such decisions are rarely as simple as just traditional, non-motorized versus motorized, and that all impacts to Wilderness character, values, and resources must be considered. In this case the decision was made to use helicopters.

Larger scale land restoration is needed at Bandelier National Monument. From decades of intense grazing prior to the area's establishment, the soils over tens of thousands of acres were compacted, the surface organic soil horizons eroded away, and the vegetation composition altered. This condition persists over vast areas of the western United States. At Bandelier the situation was so bad that natural recovery was not happening, or was so slow as to be taking place on a geologic time scale. Much of the area was intensively used by the ancestral Puebloan peoples who occupied the area. In some areas there are thousands of pieces per acre of pottery shards, chips from flint knapping, and other cultural material and resources. Due to the lack of surface organic material and vegetation, the rate of soil and cultural artifact erosion has accelerated (Allen 2004).

U.S.G.S. scientists Craig Allen has begun experimenting with practical methods to restore the natural vegetation, slow soil erosion, build an organic layer on the soil surface, and preserve the cultural material in the soil. While a variety of methods are being tried, the most successful involve sawing smaller pinyon and juniper and spreading their cut up debris around on the soil surface. This mulch material begins stabilizing the soil surface and slowing erosion, and providing organic material to catch seeds and hold moisture. Amazing results with this simple method have been achieved.

But central to preservation of Wilderness character, is the Minimum Requirements Decision Process. In the first step of the process, the activity has been deemed "necessary to meet minimum requirements for administration of the area for the purpose of the Wilderness Act" and have benefits that "outweigh the potential impacts to wilderness resources, character and purposes in a manner so as to leave the area unimpaired for future use and enjoyment as wilderness."

The second step of the process is a bit more difficult. Chainsaws area bit faster and more efficient. But clearly, the small trees can be cut up with hand saws and large-handled pruners. In the most remote locations work crews will have to camp and will have to hike in by foot or be flown in by helicopter, and be supplied by pack stock trains or helicopter. If hand tools are used the work may take a little longer and crews will have to spend more time in the Wilderness with greater impacts. How different are the impacts of these alternative methods of accomplishing the same work? These decisions have yet to be made, but hopefully they illustrate the concern over HOW the work is accomplished while having minimum impact on Wilderness character, values, and purposes so as to leave the area unimpaired as Wilderness.

The outcomes of conservation activities can be considered to offer varying degrees of benefit to wilderness ecosystems, while the activities themselves impose varying magnitudes and longevities of compromise to Wilderness character. (National Wilderness Steering Committee 2004).

When there are short-term wilderness disturbances and long-term wilderness character enhancement the decisions are fairly easy to make. This class of activity entails one-time reversals of anthropogenic changes that, once accomplished, are self-sustaining. Examples include revegetation of disturbed sites – mined lands, grazing, or development.

When there are long-duration or recurring entry requirements and mixed benefits and costs to wilderness character, the decisions become more difficult. Many ecosystems that include wildernesses suffer anthropogenic disturbances for which we lack the knowledge, the legal authority, or the financial resources to correct permanently at the present time. For example, introduced weedy plants often invade natural areas from adjacent lands, and require regular removal and frequent monitoring. Periodic liming of some eastern streams mitigates acid precipitation and permits continued survival of native fish and amphibians which otherwise would be entirely eliminated from the ecosystem—at least until the source pollution is eliminated. Pyrophytic ecosystems that lie adjacent to developed lands may no longer receive sufficient natural fire ignitions, or those ignitions are no longer socially acceptable. However, periodic managed ignitions may accomplish most of the objectives of maintaining the natural structure and composition of the native biological community. The managing agency of the Wilderness must ultimately weigh the restoration benefits to the ecosystem against the impacts to other aspects of wilderness character.

When actions support laws or agency policies, but don't directly enhance wilderness character, the decisions become even more difficult. These activities represent substantial impacts on wilderness character. They clearly violate the intent of the Wilderness Act. Some of these, such as control of pests, reflect the incapacity of some landscapes designated as Wilderness to function as such either ecologically or politically. On the other hand, some severe interventions, such as the removal of native organisms for restoration elsewhere, illuminate the fundamental and unavoidable connections between many wildernesses and their surrounding, more modified landscapes. Examples include habitat modification for endangered species, control of native pests or dangerous species to protect life or property outside wilderness, and removal of native organisms in support of restoration elsewhere. Ultimately, decisions in this category may require a public review for their resolution.

None of the activities described above is necessarily precluded by statute, regulation, or policy. However, all require evaluation in the Minimum Requirements Decision Process. You must carefully weigh the benefits against the significant impacts on Wilderness character, and consider whether the proposed restoration activity is sufficiently beneficial to outweigh those impacts.

CONCLUSION

Restoration and revegetation activities are appropriate and necessary in Wilderness. How those activities are achieved is critical. Advanced consideration of the method and tools is critical to accomplish the minimum requirements. The realistic alternatives must be evaluated in a Minimum Requirements Decision Process developed by the responsible managing agency. Plan accordingly in advance and work with the agency Wilderness management staff, and the process will yield results that are good for the restoration and revegetation activities needed to restore natural conditions to the Wilderness, and the Wilderness character will be preserved at the same time.

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IN OUR OWN HANDS: THE STORY OF WILDLANDS RESTORATION VOLUNTEERS

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ABSTRACT

Wildlands Restoration Volunteers (WRV, www.wlrv.org) restores ecologically damaged areas by building community, fostering agency and business relationships, and teaching restoration skills. Working with land management agencies, WRV accomplishes restoration work that would not be possible within budget constraints. To assure quality, WRV conducts a variety of skill trainings. WRV agency partners include U.S. Forest Service, U.S. Fish and Wildlife Service, Rocky Mountain National Park, and local open space departments. WRV also seeks participation from diverse stakeholder and user groups. Financial support comes from agency funding, grants, and private donations of money, food, equipment, and professional services. Besides restoration projects, volunteers participate in fund raising, project design, education, and cooking. Projects include: wetland and streambank restoration, weed pulls, road obliteration, tundra plantings, and post-wildfire erosion control. Project duration ranges from an evening of seed gathering to a camping weekend. WRV's volunteer restoration works because projects are fun and highly social, use a variety of skills, are located in beautiful settings, and build empowerment to care for the land. By the end of our 6th season in October 2006, volunteers will have completed 100 projects and donated 75,000 hours with a labor value of $1,000,000 to improve ecosystems as WRV seeks to put restoration “In Our Own Hands."

Volunteers complete a homemade “biolog” to stabilize the banks of Left Hand Creek.

INTRODUCTION

Wildlands Restoration Volunteers (WRV) of Boulder, Colorado, USA restores ecologically damaged areas on public land with volunteer labor. We see that numerous local forests, grasslands, wetlands, and streams have been damaged, mainly by human activities. Although more and more people live in urban areas blocked off from most connection to the natural world, many people want to give back to the community and care for the land. Knowing this, our mission is to cultivate a community spirit of shared responsibility for stewardship and restoration of public lands.

Now in our eighth year of restoration work, we complete over 20 volunteer projects annually in northern Colorado. Ed Self founded the group in 1999 with a shoreline willow planting project at the Boulder County Pella Ponds Open Space. Since 2002 WRV has been a non-profit organization with 501(c)(3) tax exempt status from the IRS and now operates with two paid staff to direct the organization and oversee projects.

Growing from a handful of volunteers, average project attendance is now 60 volunteers. Over nine hundred people have worked on at least one project, and 350 have returned for two or more projects. Many have attended more than 10, 20, or more projects. While volunteer recruitment has mainly been through word of mouth, we also send our newsletter "Gaining Ground" to 1500 WRV members and list upcoming projects on a website (www.wlrv.org). Additionally, we place project advertisements in local newspapers and give interviews on local radio stations.

Restoration project types include replanting a wetland at a former livery stable in Rocky Mountain National Park, post wildfire mulching and seeding of erosion prone foothills gulches, planting of a native shrub barrier to limit prairie dog migration from Boulder County property, removal of Russian-olive from Longmont's newly restored Left Hand Creek Greenway, working with the James Creek Watershed Initiative to reduce sediment input to the local water supply, and wetland creation at gravel mining ponds of Saint Vrain State Park. Project duration ranges from an evening of native seed gathering, to a day of willow planting along a plains stream, to a weekend of closing social trails in a high mountain wilderness.

At WRV we accomplish our mission by fostering relationships with agencies, businesses, and conservation organizations; teaching a wide range of restoration skills; and building community. Financial support comes from grants and agency funding and from private donations of money, food, equipment, and professional services.

AGENCY RELATIONSHIPS

WRV works with local land management agencies to connect people and projects. Government agency partners include US Forest Service, US Fish and Wildlife Service, US Geological Survey, Rocky Mountain National Park, Arapahoe-Roosevelt National Forest, Arapahoe National Wildlife Refuge, Saint Vrain and Eldorado State Parks, and Boulder City and County Open Space Departments. The volunteer labor accomplishes important restoration and stewardship work not otherwise possible within limited agency budgets. Typically, the land management agency recommends the restoration site and provides funding and materials. From our pool of volunteers, WRV provides the work force and expertise in design and implementation. Volunteer service includes a Project Leader to coordinate agency and volunteer roles, technical advisors who design projects and provide onsite quality assurance, and trained leadership of WRV crews on project day.

WRV specializes in organizing projects and leading our volunteers to assure quality work. Prior to the project day, project leaders and technical advisors usually hold a project orientation meeting for all crew leaders and issue detailed technical notes on all aspects of project construction including assembly of erosion control logs, installation of erosion matting, planting of ball and burlap trees, and other aspects of project construction.

Although project completion and volunteer enjoyment are important goals, on project day volunteer safety is the first priority. Early in project planning, project leaders conduct a basic risk assessment for each site and plan an emergency communication network. On project day crew leaders deliver a safety talk about site conditions such as fast-moving cold streams and ticks as well as prudent use of tools, especially pick axes and rock bars. Crew leaders maintain radio contact with the project leader throughout the work day, and most participate in a Red Cross Wilderness First Aid training and. WRV carries liability and accident insurance for volunteers.

From Eurasian milfoil control to revegetating social trails at local climbing areas, we are now offered more projects than we can fit into the April through October project season. Each winter the Project Selection Committee reviews project possibilities for meeting our goals of offering ecologically significant work in a variety of project types, work settings, and agency partnerships.

BUSINESS PARTNERSHIPS

Involvement of the business community has been essential in success of WRV projects. Local companies donate money and materials ranging from bagels and coffee at project check in, to a storage locker for tools, to a laptop computer for the office. Other businesses offer at a discount such services as tree thinning for fire prevention or site preparation with a backhoe. Corporate groups participate on weekday projects.

COOPERATION WITH CONSERVATION ORGANIZATIONS

Conservation organizations including Society for Ecological Restoration, Central Rockies Chapter and Colorado Riparian Association have provided essential cooperation in recruiting volunteers, including restoration professionals with key skills. Additionally the Indian Peaks Wilderness Alliance helps WRV identify restoration sites and Volunteers for Outdoor Colorado provides discounted training and collaborates with WRV on local stewardship projects.

TEACHING RESTORATION SKILLS

Although we now count many knowledgeable volunteers among our group, new people can come to a project with no previous restoration experience. WRV teaches volunteers a variety of techniques necessary for a successful project including how to install erosion control matting, transplant sedge plugs, tie a willow bundle, and construct check dams and water diversion structures.

During off season trainings, specialists from the professional community and experienced volunteers share their knowledge in both classroom and outdoor programs to provide essential skills for project leaders, technical advisors, project support crews (cooks and tool managers), and crew leaders. For example, with the assistance of the Colorado Outdoor Training Initiative, WRV produced a “Guide to Crew Leadership for Ecological Restoration” which details how to promote good relationships among crew members as well as step by step illustrated examples of commonly used restoration methods. Additionally, we provide mentoring for volunteers as they step into new roles.

BUILDING COMMUNITY

The sense of belonging to a restoration community, participating in beneficial projects, and the associated friendships and good memories are WRV core values. To promote these values, we bring diverse groups together to cooperate on common goals and take good care of volunteers.

Whenever possible, project planning and recruitment include local outdoor clubs such as the Colorado Mountain Club and the Sierra Club as well as other recreational users of public lands. For example, Trail Ridge Runners 4WD Club not only obtained the Colorado State Off Highway Vehicle Grant that paid for our 2004 restoration of an extreme erosion channels in a Left Hand Canyon meadow, but also worked closely with WRV to mobilize tools and materials to the site during the restoration process.

We encourage community-based involvement by recruiting volunteers who live near project sites such as the Jamestown residents who gave a spring Saturday to plant willows along an upper reach of James Creek. Through partnership with the Growing Gardens' "¡Cultiva!" program, we bring urban youth to wilderness locations for important habitat restoration work.

A major component of building community is keeping hardworking volunteers happy. To further volunteer well-being and enjoyment, WRV welcomes newcomers, and crew leaders work hard to find an appropriate task for every crew member from planting tiny trees for the youngest family member to bundling willows for the oldest. We locate projects in beautiful settings such as Fall River Pass in Rocky Mountain National Park and Mammoth Basin near the Continental Divide. A much appreciated project benefit is the hot, tasty meals prepared on-site by trained and well equipped cooks. On camping weekends, we wind down the work day with music around the campfire.

Social get-togethers continue in the November through March off season. Following completion of the last project day, a specialized group of volunteers meet at the donated storage unit to clean and repair tools and refurbish supplies. Volunteers skilled in writing and graphic design help produce the newsletter. At our winter banquet we reconnect with project friends, acknowledge the hard work of the previous season, and recognize exceptionally high levels of participation. We hold a group meeting in late winter to introduce upcoming projects and begin the process of signing up participants to fill all leadership positions.

Our years of asking for feedback on project success have helped us understand that people want to do useful ecological work that produces tangible results and to know why their work is useful to the project area. We pause at the end of the work day to celebrate and photograph our grove of newly planted cottonwood trees or former road now covered with seed, erosion matting, and a scattering of cobbles and pine needles. Computer savvy volunteers post these photographs on the website so that after the project volunteers can track revegetation progress. During the lunch break, we hold informative discussions on such topics as prairie dog ecology or the importance of riparian protection.

WRV volunteers tend to find project motivation not in t-shirt and water bottle mementos, but in working with like-minded people to "give something back and help restore the land." Through a formal survey we learned that appreciation expressed by the WRV project staff and partner agency is sufficient recognition of volunteer work. As one respondent said, "Saying 'thank you' is fine enough for me." Another replied, "Doing this project is thanks enough."

SUMMARY

Volunteer restoration works because the projects are fun and social; volunteers learn new skills and practice existing skills in doing satisfying work; projects are located in beautiful areas; and partnerships with individuals, agencies, business, and the community are strong. Most importantly, volunteer restoration works because WRV empowers the community to care for the land.

Those of us in the restoration community see that what humans are doing to the earth is unsustainable on many levels. From Ethiopia to Chile, from Antarctica to Canada, we see the same story everywhere on earth: loss and fragmentation of habitat, invasion of non-native species, global warming. The earth's ecological loss and destruction are driven by our large and still increasing human population, our greater and greater ability to conduct rapid and massive land changes, and the incremental small-scale ecological changes driven by the several billion people on the edge of survival. Underneath all these problems is an international groundswell of people wanting and working for ecological restoration.

As evidenced in both local and international conferences, people now possess a substantial body of technical knowledge as well as the broad thinking necessary to creatively connect ecology with the issues of society. Ecologists and other scientists no longer see our disciplines as isolated from society and are now thinking globally by planning ahead for global warming and associated changes, working for more sustainable government policies, and forming partnerships to solve problems from a base of community support.

As our project waiting lists become longer, the calls for how to duplicate WRV expertise become more frequent, and the necessity to engage more of society in ecological restoration becomes greater, WRV is considering broadening our network to facilitate community-based restoration in other geographic locations.

For now, WRV is looking ahead to the upcoming project season. By the end of October 2006, Wildlands Restoration Volunteers will have completed over 100 projects, donating 75,000 hours with a monetary value of $1,000,000 to improve local ecosystems, as we seek to put restoration "In Our Own Hands."

WILDLANDS RESTORATION VOLUNTEERS

ACKNOWLEDGEMENTS

With deep gratitude, we acknowledge all the hardworking and enthusiastic volunteers who have given their time, energy, creative ideas, and financial support to further ecological restoration in Colorado and give special thanks to Kimberly Kosmenko for her constant and cheerful management and Gregg Campbell for insightful comments. Additional thanks are due to the thoughtful attendees at many conferences including High Altitude Revegetation, Western Wetlands, Scientific Committee on Antarctic Research – Antarctica and the Southern Ocean in the Global System, International Society for Ecological Restoration, and Central Rockies Chapter of Society for Ecological Restoration.

WHAT MAKES REVEGETATION SUCCESSFUL AFTER TAMARISK (*TAMARIX* SPP.) REMOVAL IN SOUTHWESTERN RIPARIAN ECOSYSTEMS?

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ABSTRACT

Infestation by the non-native tree tamarisk (*Tamarix* spp., aka saltcedar) has made habitat restoration projects necessary to maintain the ecological integrity of many riparian communities in the Southwest. These restoration projects may include tamarisk removal, manipulation of hydrographs, and active revegetation of natives. There is no single strategy for achieving success in these projects; rather success will vary by site based on specific site characteristics and methods used. Revegetation success, plant species diversity, and vegetative cover were evaluated at 28 sites in New Mexico, Arizona, and Nevada where active revegetation was completed after tamarisk removal. These data were incorporated into regression tree models with independent variables that included years since removal and multiple management, climate, soils, and hydrologic variables to look at success of native plant communities, control efforts, and revegetation success. Results suggest that time, drainage and water are important factors for increased native plant community diversity and cover. Additionally, tamarisk and other noxious weeds are most persistent in dry, basic soils where competition from native species is limited. These quantitative models are intended to assist researchers and land mangers in the future to design more effective riparian restoration efforts in this critical arid lands ecosystem.

INTRODUCTION

Tamarisk (also Salt Cedar, Tamarix spp.) invasion is currently considered one of the most critical threats to southwestern river ecosystems. Tamarisk is present in every major watershed in the Southwest in a variety of native communities and is the dominant woody species in many riparian areas (Zouhar 2003). Tamarisk has been linked to changes in surface and groundwater quality and quantity, plant and animal biodiversity, wildlife habitat, soil conditions, and fire regimes (Busch & Smith 1995). However, it is unclear whether tamarisk is the cause (Busch & Smith 1993, Shafroth et al. 1995) or the effect (Anderson 1996, Sher et al. 2002) of these changes. What is clear is these changes have created situations where the native plant and animal species are not as well adapted to their environment as they used to be (Briggs and Cornelius 1998; Cohan et al. 1979) and are now having to compete with non-native invasive weeds such as tamarisk for resources.

Tamarisk, like many weedy species, is well adapted to disturbance. Stress from fire, drought, cutting, and herbicides have been shown to actually increase seed production in mature plants (Zouhar 2003). However, past studies have also shown that tamarisk seedlings are not strong competitors (Sher et al. 2000, Sher et al. 2002, Sher and Marshall 2003), suggesting that aggressive revegetation projects can prevent the reestablishment of tamarisk.

Revegetation of tamarisk infested sites, like most revegetation projects, presents a variety of challenges, many of which are site specific (National Invasive Species Council 2001). Because of the ecological changes that are associated with tamarisk, these revegetation projects often include components beyond simple tamarisk removal and revegetation with native species. Manipulation of flooding patterns, soil analyses and amendments, assessment of animal habitats, and long-term maintenance may also be required to achieve success. Every site will have different characteristics, but it is likely that some of these characteristics have much greater influence on the success of revegetation efforts than others.

Past studies have looked at the effectiveness of tamarisk eradication techniques (Stevens and Walker 1998, Dudley et al. 2000, Bryan et al. 2001, Sprenger et al. 2001) as well as success of revegetation projects (Pinkney 1992, Briggs and Cornelius 1998, Taylor 1998, Taylor et al. 2003, Taylor 2004). However, what do those successful sites have in common that the unsuccessful ones lack? Are there management, soils, climate, or hydrological characteristics that play a larger role in revegetation success than others?

To investigate these questions we evaluated three different aspects of revegetation success. First, we evaluated success in terms of tamarisk control, including both a low percentage of total cover and density that is tamarisk. Second, we looked at the success of the revegetation effort in terms of planted species cover, density, and survival success. Finally success of revegetation projects was measured in terms of the desired replacement community: as having a high native plant cover and diversity, a low noxious plant cover, and a high total plant cover of which a large percentage is native and a small percentage is noxious. ,.

METHODS

Site Characteristics

We surveyed 28 fields sites (Figure 1) in summer 2005. All sites selected had mechanical removal of *Tamarix* (burning, clearing, cutting, and/or root plowing) and/or herbicide treatment, active revegetation with native species (seeding or pole planting) between 1 and 18 year(s) ago, no subsequent mechanical *Tamarix* control after the initial removal, and no supplemental irrigation. The majority of sites (20) were dense *Tamarix* stands prior to removal; however, eight sites were mixed stands with less than 50% *Tamarix* cover. None of the 20 sites with controlled flooding regimes have flooded since tamarisk removal; however, all eight uncontrolled sites have flooded. Table 1 includes all other variables used in analyses.

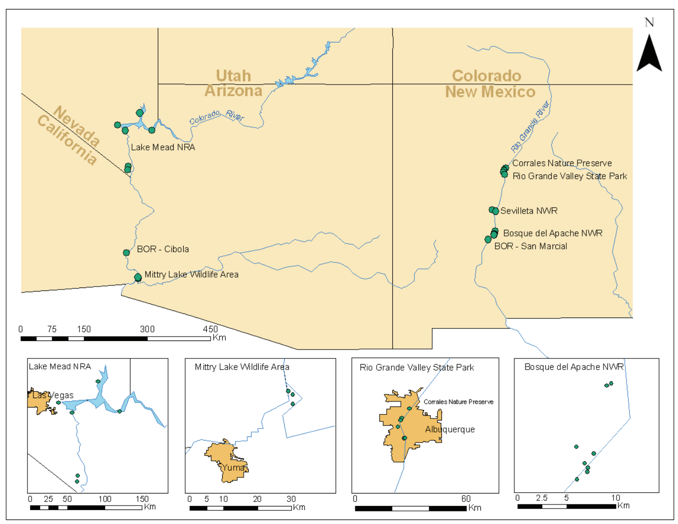

Figure 1. Field sites (n=28) were selected along the Colorado and Rio Grande Rivers in Arizona, Nevada, and New Mexico.

Vegetation & Soil Sampling

We sampled plant species composition and cover using a modified Whittaker sampling method (Stohlgren et al. 1995). At each site, three 15 m x 40 m plots were randomly selected and subplots were established within the larger area (10 - 1 m^2, 2 - 6 m^2, 1 - 60 m^2). We collected 10 surface soil samples (10 cm deep x 2 cm dia.) in each of the 1 m^2 subplots and created a composite sample for each plot for analysis of pH, texture, and salinity (Table 3). Using PCA analysis on sand, silt, and clay percentages, we produced the following vector which explained 98.4% of the variation in the data:

PCA1 = (-0.078765 * %Sand) + (0.57964 * %Silt) + (0.20887 * %Clay)

Using PCA1 as a measure of soil texture for each plot, we can most clearly separate out sand from silt and clay; such that more negative numbers have a higher percentage of sand.

Table 1. Site characteristics (mean ± SE) by river system (Colorado and Rio Grande) and revegetation method (Pole planted and Seeded), including elevation (m), and area (ha), growing seasons since *Tamarix* removal, distance to running water (m), and average annual depth to water table (DWT), annual and growing season (GS) precipitation, years with greater GS precipitation than the historic average (GS>mean), annual and growing season maximum and minimum temperatures since *Tamarix* removal for each site, average soil pH, salinity (mmhos/cm), percent gravel, and the resulting value of the principle components analysis to combine sand, silt and clay into a single variable (PCA1).

	Colorado	**Rio Grande**	**Pole Planted**	**Seeded**
Sites (Plots)	10 (27)	18 (50)	18 (50)	10 (27)
Elev (m)	283 ± 45	1432 ± 9	998 ± 82	1087 ± 108
Area (ha)	10.7 ± 4.6	10.8 ± 1.8	10.7 ± 1.8	11 ± 4.6
Yrs since Removal	4.3 ± 0.5	7.6 ± 0.7	6.9 ± 0.7	5.6 ± 0.7
Water				
Distance (m)	57.4 ± 22.3	162.4 ± 32.1	97.5 ± 29.9	177.5 ± 32.9
DWT (m)	1.0 ± 0.2	2.2 ± 0.1	1.4 ± 0.1	2.5 ± 0.1
Precipitation				
Annual	12.3 ± 0.5	20.9 ± 0.8	18 ± 0.7	18.2 ± 0.9
GS	8.4 ± 0.2	11.4 ± 0.5	10.3 ± 0.4	11.1 ± 0.4
GS > mean	3.3 ± 0.3	3.3 ± 0.7	2.9 ± 0.4	2.6 ± 0.4
Temperature				
Annual Max	29.7 ± 0.8	25.6 ± 0.5	26.2 ± 0.4	28.2 ± 0.5
Annual Min	14.7 ± 0.7	4.0 ± 0.2	8 ± 0.8	7.1 ± 1
GS Max	20.1 ± 0.8	33.3 ± 0.4	27.2 ± 0.9	31.7 ± 1
GS Min	6.8 ± 0.7	11.3 ± 0.3	9.1 ± 0.4	11.2 ± 0.4
Soils				
pH	8.3 ± 0.1	8.0 ± 0	8.2 ± 0.1	8.1 ± 0.1
Salinity	17 ± 3	9.1 ± 1.9	9.7 ± 1.8	15.9 ± 3.2
% Gravel	18.3 ± 3.2	1.0 ± 0.3	9.9 ± 2.1	1.7 ± 0.6
PCA 1	-38.8 ± 7	-13.9 ± 4.4	-27.2 ± 4.6	-14.3 ± 7.3

Data Analysis

A Pearson produce-moment pairwise correlation matrix was created for all predictor variables, and we removed all those that were highly correlated (R≥0.8) with other variables (Table 4) from our analyses. All response variables were analyzed for distribution normality and some were log or arctangent transformed prior to regression tree analyses using Systat11 statistical software. Regression tree analysis is a non-parametric alternative to multiple linear regression analysis for developing predictive and descriptive models of a single response variable with multiple predictor variables (Quinn & Keough 2002). For each predictor in the model, all possible binary splits of the response are considered, and the resulting branch is based on the predictor which results in the smallest within-group sum-of-squares for the response. This process was repeated for each resulting branch until no splits could be made that would result in either branches of greater than 5 observations or an increased reduction of error of at least 5%. The final predicted values are a mean of all the observations within the terminal group ($n \geq 5$), in contrast to the linear model which would produce predicted values for each observation. We ran multiple analyses using no more than eight predictors at a time to minimize multicollinearity until the optimal proportional reduction in error (PRE) was reached for the response.

Regression tree analysis does not allow analysis of variation within each site. Thus, using data for each site averaged from 2 to 3 plots per site may be over simplifying in the case of most of the dependent variables and some of the independent variables. However, using individual data from each plot could be considered pseudo-replication for those variables that are not specific to each plot. For example, although precipitation would be constant within a site, salinity and soil texture often varied highly between plots within a site. Since neither level is ideal for my dataset, I conducted all analyses at both levels and compared and contrasted results when they differed.

RESULTS

We identified a total of 173 unique species across all sites (CR = 92, RG = 98). Seventeen were present in both regions, 129 were native, and 33 were introduced, with six of the introduced species classified as noxious on federal or state lists (USDA, 2006).

Tamarisk Control

The most dominant noxious species was tamarisk, which was present in 83 percent of RG sites and 70 percent of CR sites. Russian olive (*Elaeagnus angustifolia*) and Siberian elm (*Ulmus pumila*) were each present in 28% of RG sites and no CR sites. Tamarisk relative cover was 0% with more than 9 cm growing season precipitation and more than 23 cm annual precipitation. When growing season precipitation was greater, relative cover increased to 32.1 + 12.4% (PRE = 0.64, Figure 2). When seeded sites were analyzed separately, relative cover of tamarisk was 0% in sites more than 4 yrs old (PRE = 0.55), and seeded plots ranged from 5.3 + 3.0% in sites that were root plowed to 11.6 + 12.4% in others (PRE = 0.60). Relative density was lowest (4.8 + 4.3%) within 15 m of permanent water, and highest (98.0 + 2.0%) farther from water with more than 2.7 mmhos/cm soil salinity in seeded sites (PRE = 0.67). Relative density was also 0% in seeded plots that receive aerial herbicide treatment and had sandy soils (PRE = 0.65).

Revegetation Success

Planted species relative cover was generally greater in older sites (35.7 + 11.2% > 8 yrs since removal), this was especially true for pole planted sites. However, the greatest relative cover of planted species (41.7 + 15.7%) was in younger, sandy (>38%), seeded sites (PRE = 0.61, Figure 3). These trends were the same for planted species absolute cover, however not quite as strong.

The percentage of pole planted individuals that were dead in our plot is likely an underestimate especially for older sites. There may have been dead individuals from years past that were no longer present or identifiable. However, the relative density of dead pole was greatest in sites with soil salinity ≥ 13.4 mmhos/cm (48.4 + 14.0%, PRE = 0.58) and in plots with soil pH ≥ 8.7 (74.2 + 14.0%, PRE = 0.53).

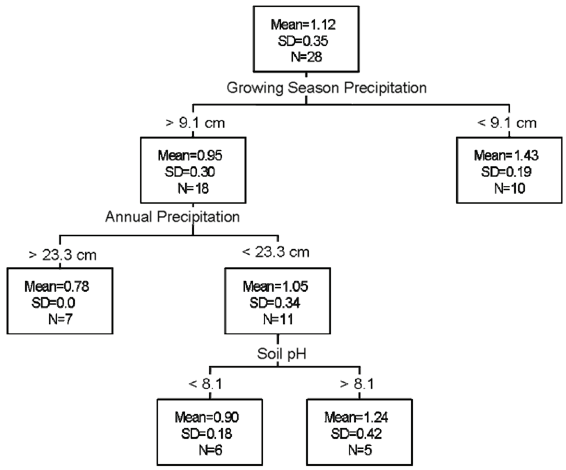

Figure 2. Regression tree for tamarisk relative cover within sites (arctangent transformed, PRE = 0.64).

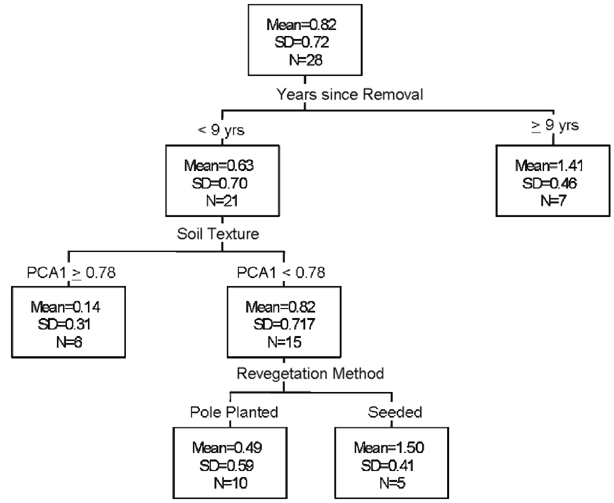

Figure 3. Regression tree for planted species relative cover within sites (log transformed, PRE = 0.61). A value for PCA1 of 0.78 is a loamy soil, sites with < 0.78 will generally have > 38% sand.

Native Plant Community

Native richness increased from 0.3 ± 0.1 (mean ± 1SE) in seeded sites to 1.5 ± 0.2 in pole planted sites, and to 1.9 ± 0.1 in sites where tamarisk removal was > 5 yrs ago (PRE = 0.71). Plot data shows the same trend at the first two nodes, but further shows that native richness increases to 2.1 ± 0.2 with increased percentage of gravel in soils (≥ 3.1%) and average annual maximum temperatures < 27.5°C (PRE = 0.75, Figure 4).

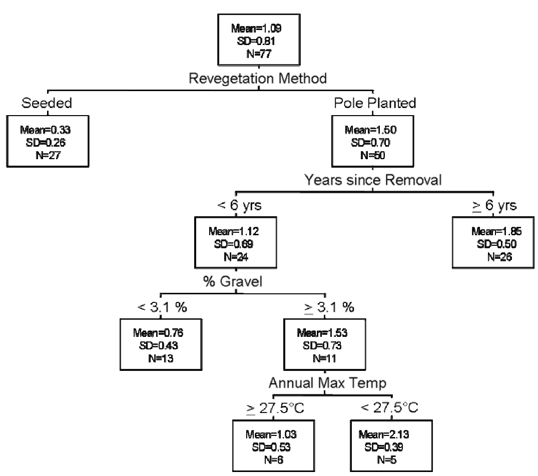

Figure 4. Regression tree for native species richness as a function of area within plots (PRE = 0.75).

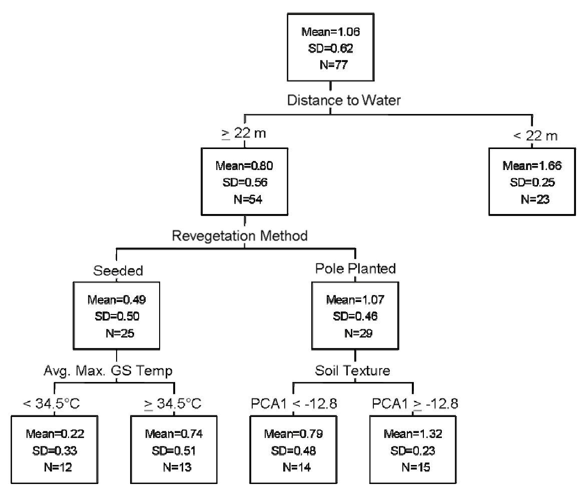

Figure 5. Regression tree for native absolute cover (log transformed) within plots (PRE = 0.68). A value for PCA1 of -12.8 is a loam or sandy loam. Values less than -12.8 generally had at least 50% sand.

Native absolute cover was greatest (50.9 + 5.2%) in plots less than 22m from water and lowest (1.3 + 0.8%) in plots farther from water, seeded and with average maximum growing season temperatures less than 34.5°C (PRE = 0.68, Figure 5). Within sites there was a single node similar to the first node at which native species absolute cover ranged from 56.8 ± 7.8% in sites less than 15m from permanent water source to 12.7 ± 2.4% in farther sites (PRE = 0.67). Absolute cover in seeded plots averaged only 10.0 ± 3.7%, but increased to 29.3 ± 10.4% in plots with sandy soil (>24%) and soil salinity greater than 3.1 mmhos/cm, but less than 23.8 mmhos/cm (PRE = 0.62).

DISCUSSION

Tamarisk relative cover was greatest in sites with low precipitation and high pH. Tamarisk is known to tolerate drought and alkalinity better than cottonwood (*Populus* sp.) and willow (*Salix* sp.) (Cleverly et al. 1997, Horton 2001a, Horton 2001b, Smith 1998). However, those sites with more favorable water and soil conditions for native species had much lower cover of tamarisk which is consistent with the findings of Sher et al. (2000, 2002, and 2003) that tamarisk is a poor competitor with cottonwood and willow. This is also supported by the increased native cover in sites closer to a permanent water source.

Overall, planted species are successfully establishing and becoming a greater part of the community in most sites with time. This is true for both pole planted sites as well as seeded sites. However, soil texture and drainage play a much larger role in success of seeded sites and soil salinity and pH in the survival of poles. The species most often used for pole planting (cottonwood and willow) are not generally tolerant of high pH (Siegel & Brock 1990) and salinity (Jackson et al. 1990, Shafroth et al. 1995, Glenn et al. 1998, Smith 1998, Vandersande et al. 2001), while saltbush (*Atriplex* sp.) (Glenn and Brown 1998, Malcolm et al. 2003) and sacatone (*Sporobolus* sp.) (Stromberg 1996, Marcum 1999) which are commonly used for seeding are quite tolerant.

Pole planting is a preferred revegetation method for many land managers because poles provide more initial cover and habitat than seeding and can survive without irrigation if drilled down to the water table. For this reason and because seeding is often used in harsher sites (deep water table and/or high salinity) it is not surprising that native richness and cover would be higher in pole planted sites than seeded sites. It is also not unexpected that native richness would increase with time in these sites given a healthy riparian habitat.

Native richness was strongly associated with soil texture, with greater richness in sandy soils. Sandy soils have better drainage and lower water holding capacity; they also have less resistance to root expansion. Although it has been observed that drought-sensitive seedlings of both tamarisk and cottonwood do better in clay than sand (Sher and Marshall. 2003), older populations have generally found more sand and gravel in cottonwood stands and more clay in tamarisk stands with clay percentage increasing with age of tamarisk stand (Stromberg 1988, Sher et al. 2002). In this analysis, native cover and planted species cover both increased in seeded plots with sandier soils, again suggesting that soil characteristics that favor native species lead to relatively less tamarisk.

Native richness was also highest when average annual maximum temperatures were not too hot (<27.5°C); however, native species cover was lowest with average growing season maximum temperatures were not too hot (<34.5°C). This is likely due to regional differences in growing season more than actual temperature differences. Growing season at the Rio Grande sites is May – September, while along the Colorado River it is November – March. Thus Rio Grande sites generally have lower annual temperatures (R = -0.63) and higher growing season temperatures (R = 0.96) than Colorado River sites. Given this, the general trend is for greater native richness and cover at Rio Grande sites than Colorado River sites. This could be due to other measured characteristics such as elevation (R = 0.97), annual precipitation (R = 0.72), average annual and growing season minimum temperature (R = -0.96 and R = 0.79, respectively), or average depth to water table (R = 0.61).

In conclusion, our results suggest that there are measurable site characteristics that lead to more successful restoration in terms of native cover and richness, planting success, and tamarisk control. Clearly soils play a large role, with lower salinity and pH and coarser texture favoring native species. Additionally, as expected, water was important, with closeness to permanent water, good precipitation, and good drainage all favoring native species. Finally, success increases with time since tamarisk removal, both in increasing native cover and richness and decreasing tamarisk cover.

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COMPARISON OF GEYER AND MOUNTAIN WILLOW GROWN IN TOPSOIL VERSUS LIME AND COMPOSTED BIOSOLIDS AMENDED MINE TAILINGS

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ABSTRACT

Native willows (*Salix* spp.) have been used to revegetate fluvially deposited mine tailings along the upper Arkansas River near Leadville, Colorado. A greenhouse study was conducted to compare growth and metal uptake characteristics of Geyer (*S. geyeriana)* and mountain willow (*S. monticola*) grown in topsoil versus lime and biosolids amended mine tailings. Biomass, leader length, and tissue metal contents were measured after four months growth. Geyer willow above and belowground biomass and leader length was greater in plants grown in topsoil compared to amended mine tailings. However, soil type did not affect mountain willow growth. Analysis for six metals yielded complex results for the two willow species and soil types. Geyer had higher concentrations of Mn and Pb in aboveground tissues, and Cu in senesced leaves and stems only than mountain when grown in tailings; while mountain leaves contained higher levels of Cd than Geyer when grown in tailings. Both willow species contained foliar Cd levels which were above livestock toxicity tolerance values. Based on growth characteristics, mountain appears better suited for restoration of mine tailings compared to Geyer willow.

INTRODUCTION

Mining activities have negatively affected lands throughout the world (Plass 2000). Prior to environmental regulations, mine tailings, which are processed ore, were left exposed and uncontained, or were often disposed of by dumping into nearby streams and rivers (Richmond 2000). This led to some of the tailings being fluvially deposited along lower stream and river banks. These tailings can be extremely acidic and contain large amounts of heavy metals (Ross 1994, Richmond 2000). The Upper Arkansas River near Leadville, Colorado was historically contaminated with heavy metals from mine tailings. These areas of fluvially deposited tailings are low in pH and devoid of vegetation (USEPA 2003). Restoring these tailings is essential to reestablishing ecosystem function in these areas. Vegetative restoration accomplishes plant and soil community establishment, bank stabilization, and metal immobilization and removal through plant uptake and compartmentalization. In order to establish vegetation in these areas, steps must be taken to create a suitable substrate for growth (Munshower 1994).

Past studies have incorporated various liming agents and/or organic materials into the mine tailings to raise pH and reduce metal bioavailability (Fisher et al. 2000, Bourret 2004, Bourret et al. 2005, Brown et al. 2005), and current suggestions (Archuleta et al. 2003) agree with this strategy. These studies have evaluated the resultant substrate conditions of such amendments to the tailings, as well as the establishment and growth of various types of vegetation planted in the amended tailings. In the Brown et al. (2005) study, extractable (plant available) metal levels were reduced. Also in this study, both ryegrass (*Lolium perenne)* and earthworm (*Eisenia foetida*) survival and metal uptake when grown in biosolids and lime amended tailings were similar to a control soil. Fisher et al. (2000) found that lime and lime plus biosolids amended tailings had lower exchangeable metal concentrations than tailings without lime.

In addition, Fisher found that Geyer willow (*Salix geyeriana*) had greater leader length when grown in lime and biosolids amended tailings, and the pH level of the tailings also increased with the application of lime. In a third study, Bourret (2004) agreed with the previous two on the reduction of exchangeable metals in tailings amended with lime and biosolids, as well as Fisher et al.'s (2000) study that the addition of lime to the tailings increased pH. Results from these studies showed that the tailings should be amended with a liming agent and organic matter to neutralize the pH and make toxic metals less available to plants and animals. These changes would positively affect the mine tailing restoration efforts.

These previous studies have used municipal composted biosolids as the source of organic amendment. Composted biosolids are readily available and low cost, while also having high organic matter content (Sopper 1993) as well as relatively low levels of Cd, Cu, Pb, and Zn (Bourret et al. 2005 and Fisher et al. 2000). Thus biosolids are an effective and economically feasible amendment for soils with high metal and low organic matter content.

Willows have long been considered in efforts to reclaim contaminated substrates due to their adaptability and fast growth (Pulford and Watson 2003). Willows have advantages such as early establishment with extensive fibrous root systems, ability to propagate vegetatively, and tolerance and accumulation of toxic metals that make them exceptionally suitable for restoration efforts (Kuzovkina and Quigley 2005). The use of willows in restoration efforts, specifically wetland restoration, provides wildlife habitat, erosion control, and water quality improvement (Kuzovkina and Quigley 2005).

Studies have shown that willow species differ in their compartmentalization of metals taken up from contaminated soils. Nissen and Lepp (1997) found that tissue concentrations of metals varied between plant parts and between species, with no predicable trend in location of the highest metal concentrations.

This study will further look at differences in tissue concentration of metals between native willow species. According to Dinelli and Lombini (1996), comparing the metal uptake of plants can help determine their ability to grow in contaminated soils. Information on the location of concentrated metals in willow plants used for restoration will have implications on wildlife and grazing considerations in the future. The overall goal for this study was to determine willow species that may be used to restore mine tailings in riparian areas. The first objective of this study was to determine survival, metal content, and root and shoot growth of Geyer and mountain willow (*S. monticola*) grown in amended mine tailings or topsoil under greenhouse conditions. Based on Bourret et al.'s (2005) study, it is expected that mountain willow will be more vigorous in growth and biomass and less affected by metals than Geyer willow. A second objective was to determine the differences in Cd, Cu, Mn, Pb, and Zn concentrations between senesced leaves, new leaves, stem bark, and stems without bark of mountain and Geyer willow grown in amended mine tailings. The location of metals in a plant is important when considering what plant parts animals or insects may be ingesting. Total root and shoot metal contents were also examined.

METHODS AND MATERIALS

This greenhouse study was conducted at the Colorado State University Plant Growth Facilities in Fort Collins, Colorado. Materials (willow cuttings and tailings) for the study were collected at a field site located 8 km southwest of Leadville, Colorado on the historic Smith Ranch. The field site is located within the 500 year floodplain on the east side of the Arkansas River, approximately 50 m from the river bank. Historically, this area was contaminated with mine-waste that originated upstream in what is now the California Gulch EPA National Priority List site (USDOI 2002). The mine-waste was entrained in the river due to several historic flood events which resulted in deposition of tailings up to 60 cm deep along an 18 km reach of the upper Arkansas River. As a result, soil ecosystem function and riparian vegetation have been degraded, and plants and animals have been exposed to high levels of heavy metals (USDOI

2002). When plants are able to grow in this area, willows dominate the riparian shrub community with an understory of grasses, sedges, and rushes (USDOI 2002).

Sample Collection and Experiment Setup

Willow pole cuttings were collected at the field site. Six Geyer and six mountain willow clones were identified and tagged. Six pole cuttings at least 90 cm in length were cut from each clone. Branches were trimmed and the poles were cut into 45 cm sections, yielding two willow stakes from each pole. The top end of each section was identified and sealed with white latex paint as recommended by the Natural Resources Conservation Service (NRCS 1994). In total, 72 cuttings of each species were collected, plus 72 extra for mortality replacement. Cuttings were placed in black plastic bags, labeled, and refrigerated at 3^{o} C until soaked and planted.

Fluvially deposited mine tailings were collected at the field site and transported to the greenhouse in Fort Collins, Colorado. A sub-sample of tailings was collected for moisture analysis. The sub-sample was weighed, dried at 105^{o} C for 24 hours, and reweighed to determine moisture content. The mine tailings were determined to be 68% dry weight. Another tailings sub-sample was analyzed for total and AB-DTPA (Ammonium bicarbonate, diethylene triamine pentaacetic acid) extractable Cd, Cu, Mn, Pb, and Zn, SMP buffer pH (which estimates reserve acidity: Shoemaker, McLean, and Pratt, 1961), pyritic sulfur content, electrical conductivity (EC), and active pH (Table 1). This information was used to calculate liming requirements.

Table 1. Characterization of a topsoil, mine tailings, and biosolids used in a greenhouse study.

Parameter	Units	Topsoil	Mine tailings	Composted Biosolids	Amended mine tailings†
Texture	--	Sandy Loam	Clay Loam		Clay Loam
pH	--	7.9	5.3	5.6	7.0
EC‡	mS cm^{-1}	1.3	3.6	2.9	2.0
NO_3-N	mg kg^{-1}	29.5	40.4	287	6.2
P- AB-DTPA§ (Total)	mg kg^{-1}	10.6	0.6 (621) ¶	(0.115)	4.3
Cd- AB-DTPA (Total)	mg kg^{-1}	0.04	25.3 (50.0)	(0.69)	26.8
Cu- AB-DTPA (Total)	mg kg^{-1}	2.7	95.9 (681)	(4.5)	128
Mn- AB-DTPA (Total)	mg kg^{-1}	2.8	1.1 (1802)	(0.011)	0. 7
Pb- AB-DTPA (Total)	mg kg^{-1}	1.1	15.0 (3059)	(1.6)	7.7
Zn- AB-DTPA (Total)	mg kg^{-1}	0.7	76.5 (1054)	(51.4)	75.9
OM#	%	1.0	7.0	82.7	7.3
SMP buffer pH††	pH		5.5		
Total sulfur	%	0.01	0.2		
Pyritic sulfur	%		0.1		
A:B potential based on pyritic sulfur	Mg CaCO3 ha^{-1}		5.7		

†Mine tailings after amendment and after the greenhouse study (4 months growth & watering)
‡Electrical Conductivity
§Ammonium bicarbonate, diethylene triamine pentaacetic acid (AB-DTPA) extractable (Soltanpour 1991)
¶Values for elemental analysis are expressed as AB-DTPA extractable amounts followed by total elemental content in parenthesis
#Organic Matter
††Shoemaker-McLean-Pratt

Composted biosolids were obtained from the municipal wastewater treatment plant at Gunnison, Colorado and transported to greenhouse storage. A sub-sample of biosolids was collected for moisture analysis. The subsample was weighed, dried at 105^{o} C for 24 hours, and reweighed to determine moisture content. The biosolids were found to be 57% dry weight. The biosolids were analyzed for total and AB-DTPA extractable Cd, Cu, Mn, Pb, Zn, and organic matter (OM) content, EC, pH, and NO_3-N (Table 1).

Topsoil was obtained from a local Fort Collins, Colorado landscape supplier. A sub-sample of the soil was analyzed for texture. The soil had less than 40% clay content and was sandy loam in texture. Organic matter, NO_3-N, and extractable phosphorus (P) contents of the topsoil were analyzed as indicators of nutrient content (Table 1). Soil pH and EC were also determined at the Soil and Water Testing Lab at Colorado State University. Total metals content in the topsoil, including Cd, Cu, Mn, Pb, and Zn, was also analyzed for comparison with the mine tailings metal content.

Based on the 5.5 SMP buffer pH of 28 Mg $CaCO_3$ ha^{-1} and A:B potential of 5.7 Mg $CaCO_3$ ha^{-1} based on pyritic sulfur content, plus 25% (8.4 Mg $CaCO_3$ ha^{-1}) to account for heterogeneity in the tailings and future acidity, the amount of lime addition to the mine tailings was 42 Mg $CaCO_3$ ha^{-1}. The biosolids rate was 224 Mg ha^{-1} and was based on the rate used by Brown et al. (2005). Amendments were weighed using a bucket and a hanging scale and materials were homogenized in a cement mixer.

Experimental Design

A completely randomized design was used with a 2 x 2 factorial arrangement of growth media and willow species. Thirty-six 7.5 L pots, (15 cm wide by 41 cm deep), were filled with biosolids and lime amended tailings and thirty-six with loamy topsoil. The bottom half of the willow cuttings were soaked in tap water for 2 days prior to planting (NRCS 1994). Eighteen Geyer willow and 18 mountain willow cuttings were planted one per pot within each soil type, giving 18 reps per willow species per soil type. Cuttings were inserted approximately 35 cm into the soil, leaving 3-4 buds aboveground. The soil was saturated immediately after planting. All pots were watered with tap water three times per week or as needed to maintain soil moisture near field capacity. Seventy-two extra cuttings of each species were started as above to ensure enough willows established and for plant tissue metal concentration analysis.

Monitoring and Harvesting

One month after planting, each willow cutting was sampled for the number and length of leaders. These data were collected every two weeks for four months, until the end of the experiment. At the end of the experiment, 18 willow cuttings within each species and soil type combination were harvested (for a total of 72 harvested willows). Aboveground growth was clipped from the stake, bagged, dried at 55^{o}C to constant mass, weighed, and recorded. Soil was placed on a 0.5 mm screen and washed from roots using a gentle stream of water. Roots were clipped from the original stake, placed in a sieve, rinsed with sodium hexa-metaphosphate to remove metals from the surface of the roots (Smucker et al. 1982), bagged, dried at 65^{o} C to constant mass, weighed, and recorded.

Aboveground and belowground willow biomass from both soil types was analyzed for metals content after drying and weighing. Biomass tissue was acid digested (Gablak et al. 1994) and analyzed using inductively coupled plasma atomic emission spectroscopy (ICP-AES) for content of Cd, Cu, Mn, Pb, and Zn.

Plants that were not harvested for aboveground and belowground biomass were harvested at the end of the experiment to determine anatomical differences in metal content. Six plants from each species that were grown in amended mine tailings had their senesced leaves collected throughout the growing period and their actively growing leaves removed from the leaders at the end of the experiment. Leaders were then clipped and peeled of the outer layer of bark by hand or with a potato peeler when necessary. Senesced leaves, actively growing leaves, bark, and bark-less stems from the plants were bagged, dried, acid digested, and analyzed for Cd, Cu, Mn, Pb, and Zn using ICP-AES.

When looking at how metal contents may affect the environment, including animals feeding on the plant, total metal content in the aboveground plant tissue becomes a concern. By multiplying the aboveground biomass (in kg) of each willow clone tested by the aboveground metal content (mg kg^{-1}) found from lab analyses, a total metal content (in mg) was calculated. This value represents the metal content in the entire aboveground growth for each willow species grown in the tailings.

Data Analysis

Data were statistically analyzed with SAS version 9.1 statistical software (SAS Institute 2003). SAS PROC MIXED analysis of variance (ANOVA) was used at alpha ≤ 0.05 significance level to determine differences in species, substrate, and plant tissue location.

Leader length, number of leaders, and root and shoot biomass were determined for each cutting (i.e. experimental unit) at the end of the study. Data were grouped by species (Geyer or mountain willow), clone, and soil type (topsoil or amended tailings). Location on the greenhouse table (row and column) was also tested and found to not be a significant factor. From the cuttings originally planted, only those that survived the duration of the study were included in the data analyses. Seventy-two plants (18 reps of each soil and species combination) were analyzed for biomass and leader length differences between species and substrate type. The plants were also analyzed for plant metal concentration differences between species, substrate type, and aboveground or belowground location. Twelve additional plants (6 of each willow species) were analyzed for tissue metal concentration differences between species. The aboveground tissues of these 12 plants were separated into four anatomical parts, living leaves, senesced leaves (collected during the course of the study), stem bark, and bark-less stems.

RESULTS

Of the 72 willows planted for the study, 13 did not establish or survive and were replaced with substitute plants of the same species, soil type, and clone. Of the 13 plants that did not establish, four were Geyer willow in topsoil, two were Geyer willow in amended mine tailings, three were mountain willow in topsoil, and four were mountain willow in amended mine tailings. The 13 cuttings that did not establish represented three of the six mountain willow clones and four of the six Geyer willow clones.

Biomass and Leader Length

Geyer and mountain willow had different aboveground and belowground biomass responses depending on the soil type in which they were grown (Figure 1). Aboveground and belowground biomass of Geyer willow grown in mine tailings was significantly less (57.6 and 66.6%, respectively) than when grown in topsoil. Mountain willow biomass was not affected by growth in amended mine tailings.

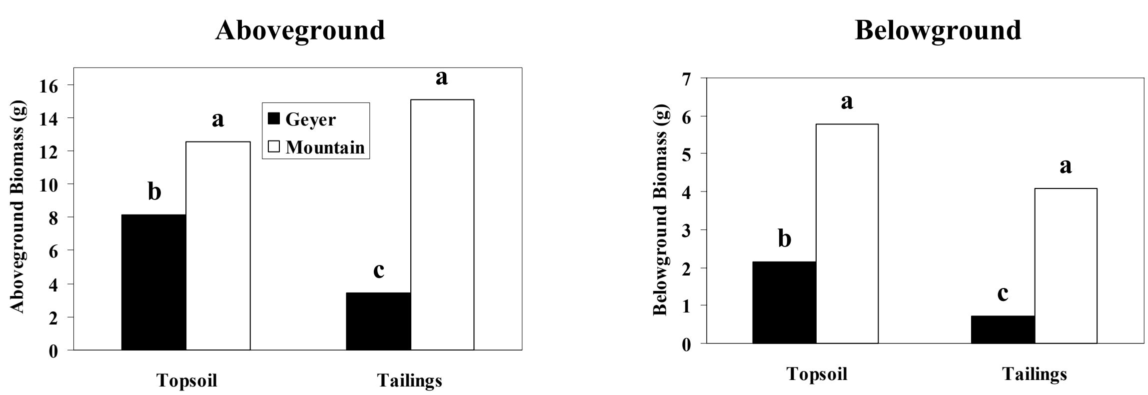

Figure 1. Aboveground and belowground biomass of Geyer and mountain willow grown in topsoil and amended mine tailings. Different lowercase letters indicate differences between the means at p≤0.05.

Soil type affected Geyer and mountain willow average final leader lengths differently (Figure 2). Geyer leader lengths were significantly less (47%) when grown in amended mine tailings than when grown in topsoil. Mountain willow leader lengths were not significantly affected by growth in amended mine tailings.

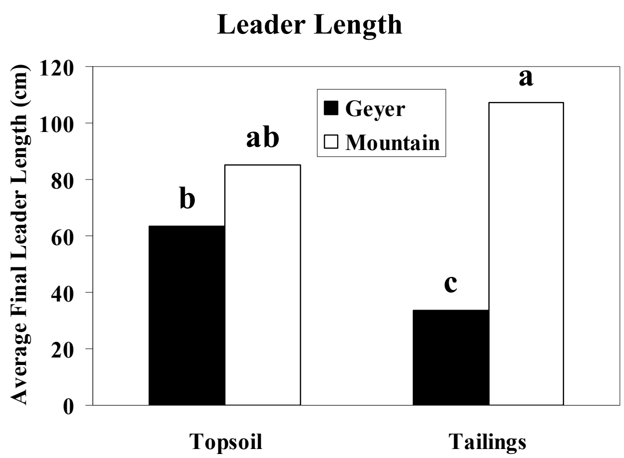

Figure 2. Average final leader length for Geyer and mountain willow grown in topsoil and amended mine tailings. Different lowercase letters indicate differences in the means at p≤0.05.

Table 2: Aboveground and belowground metal concentration differences between Geyer and mountain willow when grown in tailings or topsoil.							
Belowground Metal Concentration							
	Cd (mg kg^{-1})				Cu (mg kg^{-1})		
Soil	Geyer	Mountain	Average		Geyer	Mountain	Average
Tailings	142.2	124.1	133.1 a†		901.4	886.3	893.9 a
Topsoil	3.4	1.8	2.6 b		65.0	43.4	54.2 b
Average	72.8 A‡	63.0 B			483.2 A	464.9 A	
	Mn (mg kg^{-1})				Pb (mg kg^{-1})		
Soil	Geyer	Mountain	Average		Geyer	Mountain	Average
Tailings	899.9	977.5	938.7 a		1087.9	910.3	999.1 a
Topsoil	138.6	116.1	127.3 b		13.6	9.6	11.6 b
Average	519.2 A	546.8 A			550.8 A	459.9 A	
	Zn (mg kg^{-1})						
Soil	Geyer	Mountain	Average				
Tailings	148.0	148.0	148.0 a				
Topsoil	24.3	21.6	23.0 b				
Average	86.1 A	84.8 A					
Aboveground Metal Concentration							
	Cd (mg kg^{-1})				Pb (mg kg^{-1})		
Soil	Geyer	Mountain	Average		Geyer	Mountain	Average
Tailings	36.9	53.3	45.1 a		4.7	3.0	3.8 a
Topsoil	4.1	3.7	3.9 b		0.8	0.4	0.6 b
Average	20.5 A	28.5 A			2.7 A	1.7 B	
†Different lowercase letters in columns indicate significant differences in soil means at P≤0.05.							
‡Different uppercase letters in rows indicate significant differences in species means at P≤0.05.							

Aboveground and Belowground Metal Concentrations

Results of above and belowground metal concentrations are shown in Table 2. Geyer and mountain willow had statistically similar Cd concentrations in aboveground growth and similar Cu, Mn, Pb, and Zn concentrations in belowground growth. Geyer had higher Cd and Pb concentrations in belowground and aboveground tissue, respectively. There were significant differences between soil type and species for aboveground Cu, Mn, and Zn concentration (Fig. 3). When grown in tailings, Geyer willow had higher Mn aboveground concentration than mountain willow, and there was no species difference for aboveground Cu and Zn. Topsoil growth resulted in higher aboveground Cu, Mn, and Zn concentrations in Geyer than in mountain willow.

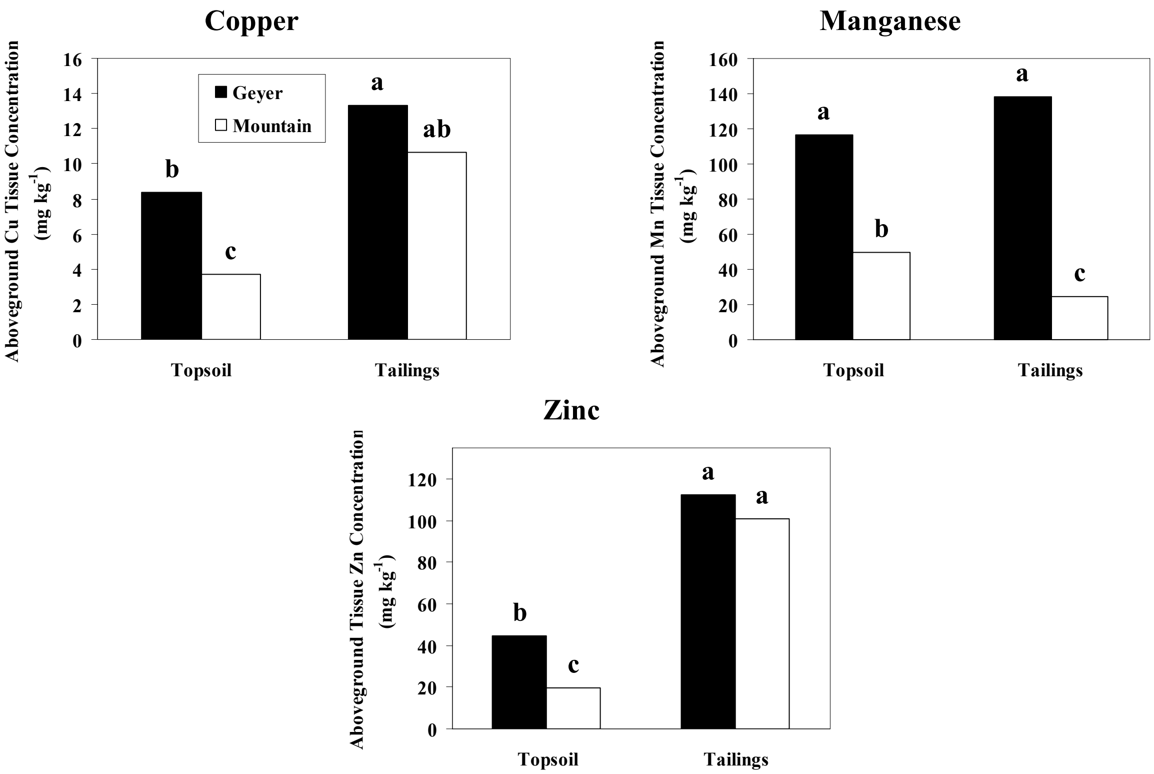

Fig. 3. Aboveground Cu, Mn, and Zn tissue concentrations of Geyer and mountain willow grown in topsoil and amended mine tailings. Different lowercase letters indicate differences in the means at the p≤0.05 level.

Table 3. Anatomical metal content differences for Copper (Cu) and Zinc (Zn) for two willow species grown in amended mine tailings.

	Cu mg kg^{-1}					Zn mg kg^{-1}		
Part	Geyer	Mountain	Average		Part	Geyer	Mountain	Average
New Leaves	9.2	7.2	8.2c		New Leaves	121.1	111.7	116.4 b
Bark	7.6	6.3	7.0c		Bark	82.2	78.7	80.4 c
Stems	23.0	12.5	17.8 b		Stems	35.2	26.1	30.6 d
Dead Leaves	32.4	22.3	27.4 a		Dead Leaves	158.4	143.4	150.9 a
Average	18.1 A‡	12.1 B			Average	99.2 A	90.0 A	

†Different lowercase letters in columns indicate significant differences in soil means at P≤0.05.

‡Different uppercase letters in rows indicate significant differences in species means at P≤0.05.

Anatomical Parts Metal Concentrations

Species differences in metal concentration in the four plant tissues for Zn, Mn, and Pb are shown in Figure 4. Geyer had higher Mn and Pb concentrations than mountain in all plant tissues, and mountain had higher Cd concentration than Geyer in all tissues. Geyer willow had higher concentration of Cu than mountain willow regardless of soil type (Table 3), while there was no difference between species for Zn concentrations. For both species, Cu and Zn concentrations were highest in senesced leaves, followed by stems in Cu, and new leaves in Zn concentration analysis.

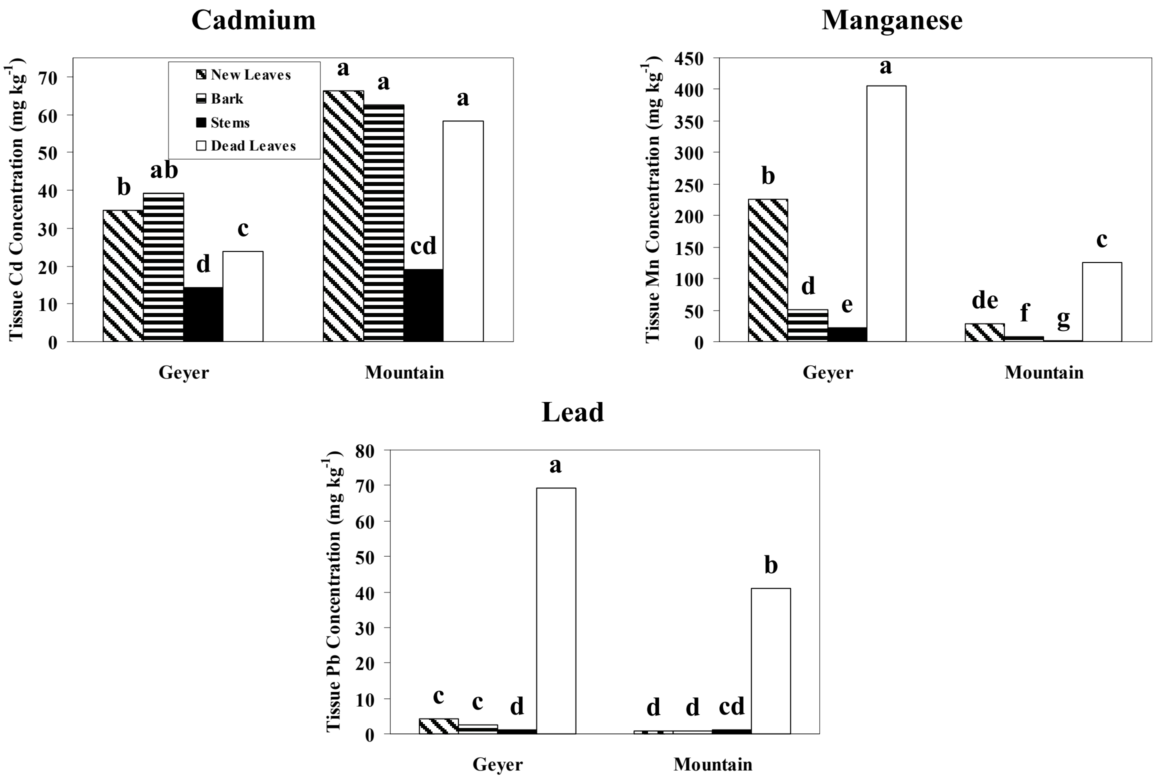

Figure 4. Cd, Mn, and Pb metal concentrations in anatomical parts of Geyer and mountain willow grown in mine tailings. Different lowercase letters indicate differences in the means at P≤0.05.

Table 4. Total aboveground manganese (Mn) and Lead (Pb) metal content for two willow species grown in topsoil and amended mine tailings.								
	Mn mg plant^{-1}					Pb mg plant^{-1}		
Soil	Geyer	Mountain	Average		Soil	Geyer	Mountain	Average
Tailings	0.56	0.39	0.4795b		Tailings	0.024	0.05	0.03328a
Topsoil	0.73	0.58	0.6524a		Topsoil	0.004	0.005	0.004618b
Average	0.6456A	0.4864A			Average	0.01193A	0.02597A	
†Different lowercase letters in columns indicate significant differences in soil means at P≤0.05.								
†Different uppercase letters in rows indicate significant differences in species means at P≤0.05.								

Total Metal Content

Geyer and mountain willow had different total Cd, Cu, and Zn contents in the two soil types (Figure 5). When grown in mine tailings, mountain willow had higher Cd, Cu, and Zn contents than Geyer willow. Mountain willow also had higher Cd content when grown in topsoil. Both Geyer and mountain willow had similar Mn and Pb total contents (Table 4). Watson et al. (2003) also found differences in total metal contents of roots and shoots when looking at two willow species for Cd, Cu, Pb, and Zn. In that study, the willow that showed less metal toxicity based on biomass had higher total metal content in roots and shoots, even though it had less metal concentration than the other willow. Similar to the Watson et al. (2003) study, mountain willow's biomass was not affected by metal toxicity, had higher Cd, Cu, and Zn total metal contents, and had lower Cd belowground and Cu aboveground concentrations than Geyer willow.

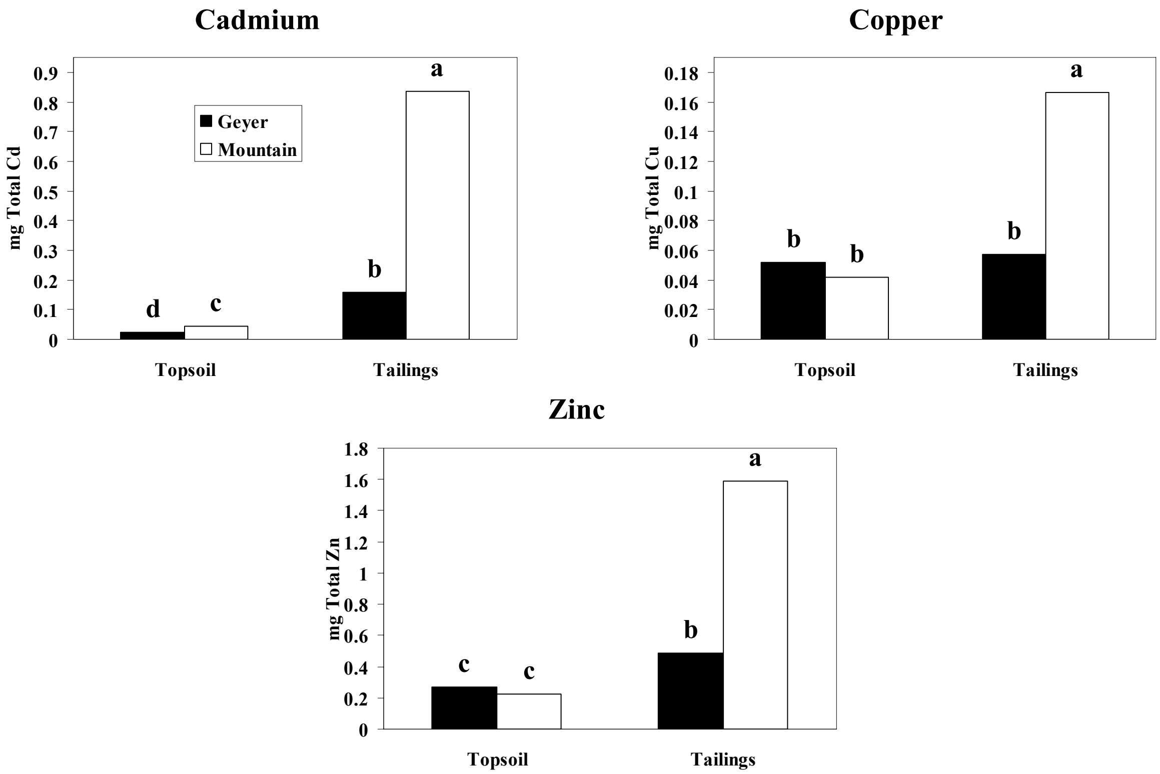

Figure 5. Total aboveground metal content of Geyer and mountain willow grown in topsoil and amended mine tailings. Different lowercase letters indicate differences between means at $P \leq 0.05$.

DISCUSSION

Biomass and Average Leader Length

Biomass of Geyer willow was negatively affected when grown in amended mine tailings versus topsoil (Figure 1). Biomass decreases have been used in previous studies to show a lack of metal tolerance in some willow species (Bourret et al. 2005, Kuzovkina et al. 2004, Landberg and Greger 1994, and Watson et al. 2003a). The greatest measured decrease was a 66% reduction in the roots of Geyer willow. Similarly, Kuzovkina et al. (2004) and Landberg and Greger (1994) found that belowground biomass of willows was affected more by metals than aboveground growth. Mountain willow, however, was not affected by soil type and actually appeared to grow as well or slightly better in the amended mine tailings.

Geyer willow leader lengths were 47% shorter when grown in tailings versus topsoil (Figure 2). Watson et al. (2003a) also used differences in height to distinguish the susceptibility of willow clones to metal toxicity. As seen with the biomass results, mountain willow was not affected by growth in amended mine tailings. Geyer willow biomass and leader length were negatively affected by growth in the amended mine tailings, indicating mountain willow as a preferred species for field restoration efforts.

Soil Chemistry

The lime and biosolids amended mine tailings had a higher pH level than the non-amended tailings, as is the expected result of liming (Bourret et al. 2005, Brown et al. 2005, Fischer et al. 2000). All of the elements analyzed for total metals in non-amended mine tailings were within (Mn) or above (Cd, Cu, Pb, and Zn) the critical soil levels for plant toxicity (Alloway 1995). For many metals, amending tailings with a liming agent and biosolids reduces their availability for plant uptake (Brown et al. 2003, Brown et al. 2005). In this study, however, availability of Cd and Cu actually increased by 6% and 33%, respectively, following addition of the amendments. Availability of Mn, Pb, and Zn decreased by 39, 49, and 1%, respectively, with the addition of lime and biosolids which is similar to findings by Ye et al. (2000) and Brown et al. (2005) for Pb and Zn.

Metal Concentrations in Aboveground, Belowground, and Anatomical Parts

Similar to previous work (Kuzovkina et al. 2004), this study found significant differences in the uptake of Cd and Cu between willow species when grown in the presence of these metals. In this study, Cd was higher in belowground tissues of Geyer willow than mountain, but higher in mountain willow aboveground new and senesced leaves than Geyer. Copper concentrations were higher in all Geyer aboveground anatomical parts (new leaves, senesced leaves, bark, and bark-removed stems). Geyer also had higher Mn in all aboveground tissue and anatomical parts. Mountain willow had higher Pb in aboveground parts, but Geyer willow had higher Pb concentrations than mountain in senesced leaves, new leaves, and bark.

Nissen and Lepp (1997), Punshon and Dickinson (1997), Punshon et al. (1995) and Watson et al. (2003a) show exclusion and low mobility of Cu in willows, so more Cu is held in or bound to the roots than the shoots. The results from Kuzovkina et al. (2004) and this study show that willows hold Cd and Cu in the roots rather than translocate the metals into shoot tissue. Therefore, growing willows in contaminated soils could be used to limit ingestion of high levels of Cd and Cu by wildlife or domestic browsers.

While no metals were found in higher concentration in aboveground tissue than in belowground tissue for either willow species, Zn came close (Table 2, Figure 3). Nissen and Lepp (1997) found that Zn was concentrated in shoots and leaves more than belowground tissues.

As compiled by Kabata-Pendias (2001), excessive or phytotoxic leaf tissue concentrations of Cd, Cu, Mn, Pb, and Zn are 5-30, 20-100, 400-1000, 30-300, and 100-400 mg kg^{-1} dry matter, respectively. When grown in amended mine tailings, plant tissues that were within or exceeded these levels in this study were: all of the Geyer and mountain aboveground tissues (Cd), senesced leaves of both species (Cu and Pb), Geyer senesced leaves (Mn), and new and senesced leaves of both species (Zn). The highest concentration of Cu, Mn, Pb, and Zn for both species was in senesced leaves. This may be a way of the willows getting rid of metals to avoid metal stress (Watson et al 2003a). Also in Watson et al. (2003a) the more metal stressed species of willow (based on a decrease in biomass) translocated higher levels of metals, similar to Geyer in this study.

Previous studies have found that willows have differences in metal uptake based on plant location. Like this study, many others have found metal concentrations higher in the leaves than the stems of willows. In summary, Hammer et al. (2003) found Cd and Zn, Riddell-Black (1994) found Cd, Cu, Pb, and Zn, and Vsylouzilova et al (2003a) found Cd and Pb all with concentrations higher in leaves than in stems. This study had the same results with the exception of Cu, which was found highest in senesced leaves followed by stems, and then mature leaves.

Plant tissue levels of Cd and Zn in Geyer and mountain willow mature leaves met or exceeded those listed as excessive or toxic to plants in Kabata-Pendias (2001). The leaves of both Geyer and mountain willows grown in the amended mine tailings showed symptoms of both Cd and Zn toxicity as described in Kabata-Pendias (2001). However, willows grown in topsoil also showed some toxicity symptoms, including leaf curling, chlorosis, and necrotic or brown leaf margins. This is puzzling because the topsoil did not have phytotoxic concentrations of any of the metals studied.

The uptake and absorption of Fe by animals can be negatively affected when ingesting heavy metals in the diet. Plant tissue concentrations of Cu, Mn, Pb, and Zn were all safely below the minimum tolerance level of domestic animals, except for Pb concentrations in senesced leaves of Geyer and mountain willow. Animals that uptake Pb can experience Fe-deficiency-induced anemia (Wilkinson et al. 2003). In the US, the maximum tolerance level of dietary Pb is 30 mg kg^{-1} (National Research Council 1980). The only aboveground tissues that exceeded this level of lead were senesced leaves, which would not likely be ingested by grazing animals. Cadmium can also decrease the absorption of Fe in the small intestine and reduce the uptake of Cu, an essential micronutrient in sheep (Wilkinson et al. 2003). The Cd dietary maximum tolerable level is only 0.5 mg kg^{-1} (National Research Council 1980). Unfortunately, this level of Cd was greatly exceeded in all aboveground plant tissues in this study and would pose the greatest threat to Fe deficiency in grazers.

Maximum tolerable Cd levels for domestic animals are 0.5 mg kg^{-1} (National Research Council 1980), and levels found in Geyer and mountain willow were 37 and 53 mg kg^{-1}, respectively, for aerial vegetation in this study. Therefore, even though most of the Cd from the soil was kept with the roots, toxic levels were translocated into aboveground growth. Aboveground metal content in plant tissues is especially important in the case of Cd. This metal is much more zootoxic than phytotoxic, and zootoxic at much lower levels than other heavy metals that have been studied (Shtangeeva 2005). Cadmium is a trophic accumulator, so even small concentrations in plant tissue can have detrimental effects on animals ingesting aboveground plant parts. This is a concern for land and wildlife managers working with revegetating mine tailings because of the potential zootoxic affects of ingesting plant tissue with these high Cd concentrations.

Total Metal Content

For Mn, plants of both species grown in topsoil actually had higher total metal contents than those grown in tailings. This was a result of the higher Mn content of the topsoil. Geyer willow had significantly higher Mn concentrations than mountain willow in aboveground growth (138 and 25 mg kg^{-1}, respectively, for Geyer and mountain willow) as well as all the anatomical parts when grown in tailings. However, this did not lead to significant differences in total Mn content between the willows. This is an example of the higher biomass of mountain willow diluting metal concentrations to result in a lower total metal content.

CONCLUSION

Because Geyer willow biomass is adversely affected by growth in mine tailings while mountain willow remains unaffected, mountain willow should be used in restoration projects in this area instead of Geyer willow. While Fisher et al. (2000) found that Geyer willow could survive in amended mine tailings, Bourret et al. (2005) and this study show that mountain willow is better suited for growth in these tailings.

Because Cd is such a zootoxic metal and is trophically concentrated, it is important that restoration of amended tailings be with plants that do not transport large amounts of Cd to aboveground parts that can then be consumed by animals. Unfortunately, both Geyer and mountain willow had large amounts of Cd in their aboveground growth. Because of the high uptake of Cd by both willows, care should be taken in restoration efforts to ensure domestic are kept out of these areas. However, because the biomass of Geyer is so much less than that of mountain, the actual total metal content of the feed (biomass*tissue concentration) is much less for Geyer. So based on Cd alone, Geyer would appear to be the better species to use in this situation.

Future studies should include more long term analyses of metal uptake and plant growth. Brown et al. (2003) indicate that plant metal uptake may be the highest in the first year following biosolids amendment. Dinelli and Lombini (1996) found an initial flush in metal content of willow tissue at the beginning of the growing season in mine spoils, followed by a decline in metal content towards the end of the growing season for Cu and Zn. Such a decline in metal content may be the result of a dilution of metals due to growth (Watson et al. 2003a). Another study (Riddell-Black 1994), however, found that Cd, Zn, and Pb concentration actually increased from the beginning to the end of the season. By conducting a growth and metal uptake study over more than one growing season, similarities or differences between species may be revealed that would contribute to the knowledge needed to find ideal revegetation species for this environment.

Several studies have looked at many species and clones of willows to determine their suitability for growth and survival in metals contaminated soils (Pulford et al. 2002, Vyslouzilova et al. 2003a, Vyslouzilova et al. 2003b, Watson et al. 2003a, and Watson et al. 2003b). More species native to the upper Arkansas River drainage should be studied to find additional species which would be suitable for restoration efforts in this area.

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FOLIAR HEAVY METAL CONCENTRATIONS OF TWO WILLOW SPECIES GROWING IN A CONTAMINATED RIPARIAN ZONE

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ABSTRACT

Heavy metal concentrations in the leaf tissue of two montane willow species, Geyer (*Salix geyeriana* Andersson) and mountain willow (*Salix monticola* Bebb), found growing in a riparian zone impacted by mine tailing were investigated in an observational study. Clones of the two willow species were paired at eight sites and sampled three times (July, August, and September) for leaf tissue metals. Soil around each clone was sampled once (August) for plant available metals. Trace element concentrations of Cd, Cu, Pb, and Zn were 55, 45, 96, and 15% higher, respectively, in mountain willow leaves than in Geyer willow leaves. However, Mn concentrations were 131% higher in Geyer compared to mountain willow leaves. Metal concentrations in the leaves of both species increased from July through September, with the exception of Cu, which decreased during this time period. Results from this study indicated that mountain willow accumulated higher concentrations of metals in its leaf tissue than Geyer willow and leaf metal uptake increased from July through September for both willow species. In addition, high levels of Cd in the leaves of both willow species pose a potential threat to livestock and wildlife throughout the growing season.

INTRODUCTION

A century of mining and several historic flood events in the Leadville, Colorado area resulted in mine waste material (tailing) containing toxic levels of metals being deposited along an 18 km reach of the upper Arkansas River (Swayze et al. 1996). These tailing deposits are devoid of vegetation (URS Operating Services 1999) and toxic material continually erodes into the river, which may adversely impact water quality (Colorado Water Control Division 1988; Walton-Day et al. 2000). In addition, mine shaft and tailing pile drainage was used to irrigate meadows in the Leadville area from the late 1800s until the early 1900s (B. Smith 2002, personal communication) before the detrimental effects of trace metals on animal health and forage quality were noted in 1906 (Levy et al. 1992).

The greatest success in revegetation of tailing material has been achieved when natural pioneer species such as alders (*Alnus* spp.), birches (*Betula* spp.), willows (*Salix* spp.), larches (*Larix* spp.), and pines (*Pinus* spp.) were planted (Good et al. 1985). Willows are particularly important in revegetation efforts because they are an extremely hardy species that opportunistically colonize disturbed and industrially contaminated soils (Grime et al. 1998; Punshon, 1996). This makes them ideal for restoration of riparian systems impacted by mining. In addition, their extensive root system and fast growth help to stabilize stream banks (Gray and Sotir 1992) and create critical habitat and resources for a number of wildlife species (Sommerville 1992).

Previous revegetation studies initiated on fluvial tailing deposits on the Arkansas River near Leadville, Colorado indicate that herbaceous vegetation along with mountain willow (*Salix monticola* Bebb) can be established on these deposits by first amending with lime and organic matter (Fisher 1999; Bourret, 2004). However, efforts to revegetate with Geyer willow (*Salix geyeriana* Andersson) produced lower survival rates than mountain willow and many surviving Geyer willows exhibited signs of chlorosis, slow leader growth, and overall low vigor.

The objectives of this study were to: (i) quantify the metal accumulative and tolerance capacity of two willow species, Geyer and mountain, that were growing within a metal contaminated site; and (ii) determine if leaf metal concentrations in these willows fluctuated throughout the growing season. These willows were chosen for study because they are the two most prevalent willow species in the 18 km reach of the upper Arkansas River near Leadville that has been contaminated with fluvial mine tailing deposits. This type of information on willows is lacking from the literature and is sought after by risk assessors.

METHODS AND MATERIALS

Study Area

This study was conducted on the east bank of the Arkansas River (elevation 2,900 m) 8 km south of Leadville, Colorado (39°12’ N, 106°21’ W). The study site is a riparian shrub community dominated by Geyer and mountain willow. The willows are growing near fluvial mine tailing deposits which are devoid of vegetation because of low soil pH and high metal concentrations. The willows appear vigorous and healthy, despite growing in soil which is moderately impacted from mine tailing and previous irrigation practices (Table 1).

Data Collection and Chemical Analyses

Individual clones of Geyer and mountain willow were paired at eight randomly selected sites surrounding fluvial mine tailing deposits. Leaf samples from each clone were collected on 10 July, 8 August, and 5 September 2002, dried at 60°C for 72 h, and ground through a 2 mm screen in a Wiley mill. Metals in the leaf tissue were determined by digesting 1 g of sample in a solution containing 2 ml perchloric acid and 6 ml of nitric acid at 200ºC for 2 h (Miller 1996). Analyses of trace metals were determined using inductively coupled plasma atomic emission spectroscopy (ICP-AES).

Soil samples from each site were collected on 8 August 2002 by taking eight soil cores (1.9 cm diameter) to a depth of 60 cm around each clone. Soil samples were air-dried, finely ground, and sieved through a 2 mm screen. Soil samples were analyzed for pH, electrical conductivity (EC), and exchangeable metals. Exchangeable metal ions were extracted by shaking 10 g of soil in 50 ml of 1.0 M potassium chloride (KCl) solution for 30 min and then filtering through Whatman #42 filter paper (procedure adapted from Bertsch and Bloom 1996). Concentrations of trace metals in solution were analyzed using ICP-AES.

Statistical Analyses

Statistical analyses were conducted using SAS version 8.2 software (SAS Institute 2002). The PROC MIXED analysis of variance (ANOVA) repeated measures procedure was used to determine statistical differences among months and willow species. The least square means method was used to test parameters for significant treatment differences at P   0.05. All data were tested for homogeneity of variance and normality (Ott 1993) and leaf metal concentrations of Cd and Mn were log transformed to meet these assumptions. All means were presented as non-transformed values for ease of interpretation.

RESULTS

Overall, exchangeable metal concentrations (KCl) in the soil were lower than published phytotoxicity levels (Table 1) (Alloway 1990). However, Cd concentrations were in the phytotoxic range (10 mg kg^{-1}) (Alloway 1990). Concentrations of Cu were below what are considered normal or sufficient levels in soils (20 mg kg^{-1}) (Riddel-Black 1994).

Table 1. Soil characteristics averaged over the 8 sites along the Arkansas River near Leadville, Colorado.

Parameter	Units	Mean	Std Dev	Minimum	Maximum
pH	standard	5.9	0.4	5.3	6.4
EC†	S m^{-1}	1.22	0.88	0.56	3.58
Cd‡	mg kg^{-1}	19.34	11.77	3.31	43.43
Cu	mg kg^{-1}	0.97	0.98	0.37	3.42
Mn	mg kg^{-1}	24.89	13.28	10.92	60.93
Pb	mg kg^{-1}	10.90	14.47	0.96	56.03
Zn	mg kg^{-1}	78.67	17.40	48.29	119.40

† Electrical conductivity.
‡ Potassium chloride (1.0 M KCl) exchangeable.

Table 2. Metal concentrations in willow leaf tissue as affected by species and month of sampling. Willows were growing adjacent to the Arkansas River near Leadville, Colorado in soil moderately contaminated with mine tailing.

	Cd†				Cu		
Month	Geyer	Mountain	Average‡		Geyer	Mountain	Average
	-----------------mg kg^{-1}-----------------				-----------------mg kg^{-1}-----------------		
July	5.2	7.4	6.3c§		4.5	7.4	6.2a
August	6.2	9.3	7.7b		3.9	5.7	4.8b
September	8.3	13.8	11.0a		3.8	5.2	4.5c
Average¶	6.6B#	10.2A			4.2B	6.1A	
	Mn				Pb		
July	142.9	79.4	111.1c		0.8	3.0	2.1c
August	205.2	86.5	145.9b		2.1	4.0	3.1b
September	259.5	97.6	178.6a		3.6	5.3	4.5a
Average	202.5A	87.8B			2.2B	4.3A	
	Zn						
July	465.3	546.5	505.9c				
August	602.6	688.8	645.7b				
September	756.0	867.2	811.6a				
Average	608.0B	700.8A					

† Statistical comparisons for Cd and Mn were conducted on log transformed data, but values are presented as non-transformed for ease of interpretation. All other metals were not transformed.
‡ Averaged over species, because species by month interaction was not significant at P 0.05.
§ Different lower case letters in columns indicate significant differences in means at P 0.05.
¶ Averaged over month, because species by month interaction was not significant at P 0.05.
# Different upper case letters in rows indicate significant differences in means at P 0.05.

The two willow species differed in accumulation of metals in their leaf tissue (Table 2). Trace element concentrations of Cd, Cu, Pb, and Zn were 55, 45, 96, and 15% greater, respectively, in mountain willow leaves than in Geyer willow leaves. However, concentration of Mn was 131% greater in Geyer leaves compared to mountain willow leaves.

In general, concentrations of metals in Geyer and mountain willow leaves increased from July through September (Table 2). Concentrations of each metal were significantly higher from one month to another. The only exception was the concentration of Cu, which decreased 37% from July through September.

Concentration factors (CFs) for Geyer and mountain willow showed a general trend of excluding Cd and Pb and concentrating Cu, Mn, and Zn in leaf tissue (Table 3). Concentration factors greater than 1 indicate accumulation of metals in the leaf tissue while CFs less than 1 indicate exclusion of elements.

Table 3. Concentration factors† averaged over the 8 sites and 3 months of sampling for Geyer and mountain willow growing along the Arkansas River near Leadville, Colorado.

Metal	Geyer	Mountain
Cd	0.34	0.53
Cu	4.33	6.29
Mn	8.14	3.53
Pb	0.20	0.39
Zn	7.73	8.91

† Concentration factors calculated from mean soil and leaf tissue levels given in Tables 1 and 2.

DISCUSSION

Results indicated that there were substantial differences in the uptake of metals by the two species of willow evaluated in this study. Mountain willow accumulated greater concentrations of metals than Geyer willow, with the exception of Mn. These results are consistent with past studies that have found considerable variability among different willow species in their heavy metal accumulation and tolerance (Bourret et al. 2005; Dickinson et al. 1994; Landberg and Greger 1994; Riddel-Black 1994; Punshon and Dickinson 1999). This variability can be attributed to the various willow species having different mechanisms of exclusion, compartmentalization, and binding to specific substances in response to heavy metals (Punz and Sieghart 1993). Landberg and Greger (1994) stated that the mechanism of tolerance is classified into two groups:

Low net uptake or exclusion of metals.
High accumulation of metals in the roots or shoots, where metals are detoxified through compartmentalization within the vacuoles of the plant tissue.

These two mechanisms explain why Landberg and Greger (1994) found that tolerant clones could have either high or low accumulation of metals in their plant tissue. Therefore, net metal uptake and accumulation in willows does not appear to be correlated with tolerance, because the plant could be either excluding or accumulating metals.

In addition to willow leaf metal concentrations varying among species, differences among leaf, stem, and root concentrations have also been noted (Bourret et al. 2005; Nissen and Lepp 1997; Vyslouzilova et al. 2003). Analyses of metal concentrations in woody plants have primarily been focused on foliar analysis over stem and roots because of food chain transfers and metal recycling through leaf litter (Nissen and Lepp 1997). After reviewing literature for pollution assessment, Lussaert (2001) concluded that leaf analysis was the most applied approach over root, wood, bark, and bud sampling. Although leaf analysis is considered the most applied approach, the loci of metal compartmentalization within willow species is important to understanding metal cycling, tolerance, and exclusion. Root and stem sampling would be required in addition to foliar sampling to determine the mechanism of tolerance in Geyer and mountain willow as being exclusion or compartmentalization.

Metal concentrations in the leaf tissue of both willow species in this study, with the exception of Cu, were significantly higher at the end of the growing season than in July. This is consistent with past studies by Riddel-Black (1994) and Vandecasteele et al. (2004) which reported the same trend of Cu exclusion and other metal accumulation in willows at the end of the season, signifying that there was translocation of metals from the roots and stems to the leaves before senescence. A few deciduous plant species have the ability to translocate absorbed metals to their leaves immediately before senescence allowing them to shed the metals with the leaves, consequently limiting metal accumulation in roots or stems (Baker, 1981). The varying leaf metal concentrations throughout the growing season stress the importance of standardizing sampling time to achieve comparable results (Vandecasteele et al. 2002).

The results indicate similar patterns of metal concentration and exclusion within the leaf tissue of the two willow species relative to the soil. Concentration factors (Table 3) show high concentrations of Zn and Mn in the leaves of both species, indicating significant within-plant mobility. This is consistent with past studies involving mountain and Geyer willow which reported higher concentrations of Mn and Zn relative to the soil than other metals (Bourret 2004; Bourret et al. 2005). Conversely, concentration factors for Pb in Geyer and mountain willow (0.20 and 0.39, respectively) indicate a trend of exclusion within the foliage of these willows. These findings are consistent with previous studies which indicate that Pb concentration in plant biomass was lower compared to Cd and Zn, and translocation of this element from twigs to leaves was limited (Vyslouzilova et al. 2003).

Concentration factors for Cu were higher (4.33 – 6.29) in this present study compared to other studies done on Geyer and mountain willow (Bourret 2004; Bourret et al. 2005) and other species of willow (Nissen and Lepp 1997; Riddel-Black 1994). Using data from field and greenhouse studies where Geyer and mountain willow were grown on amended mine tailing, CFs could be calculated for Cu (Bourret, 2004; Bourret et al. 2005). These ranged from 0.2 to 0.3 in the field and <0.01 in the greenhouse. Nissen and Lepp (1997) reported a mean CF for Cu in willow leaves of 0.92 and calculations from Riddel-Black (1994) ranged from 0.075 to 0.11 indicating exclusion of soil Cu from the shoot system, a reflection of the low mobility of this element in plants (Alloway 1990).

In this study, copper concentrations in the top 60 cm of soil were below normal or sufficient levels (20 mg kg^{-1}) for plant growth (Riddel-Black 1994). Based on the higher than expected CFs for Cu, sampling the soil once in August may not have been a true representation of Cu availability to the willows. Bourret et al. (2005) reported that extractable concentrations of Cu were higher in saturated tailing (reducing conditions) compared to unsaturated tailing. Soil reduction releases metals associated with Mn and Fe oxides, which are susceptible to reductive dissolution (Charlatchka and Cambier 2000; Davranche and Bollinger 2000). Oxides dissolve and the metals are released into the soil solution and become bioavailable to plants. Therefore, fluctuations in anaerobic conditions found in riparian areas may promote the dissolution of heavy metals bound to Mn and Fe oxides in pyritic mine tailing (Svendsen 2002). The groundwater level at the study area fluctuated from about 60 to 90 cm from the soil surface through the growing season (Bourret 2004) which would have influenced the availability of Cu to the willows over time. Willows are also known to concentrate their roots just above saturated soil zones (Bourret et al. 2005). Additionally, Vandecasteele et al. (2002) found that volunteer willows rooting in a

60 cm thick uncontaminated cap layer covering polluted sediment were preferentially rooting in the sediment layer, because the layer was well supplied with nutrients.

Concentrations of Cd and Zn in the leaf tissue were above levels considered toxic to plants (3 and 300 mg kg^{-1} for Cd and Zn, respectively) (Kabata-Pendias and Pendias 1992) for both willow species during all months. Riddel-Black (1994) also found Cd concentrations greater than phytotoxic levels in leaves of all four willow species studied throughout the growing season; however, Zn concentrations only reached phytotoxic levels at the end of the growing season. Past studies have shown that many species of willow have high heavy metal uptake and tolerance, especially of Cd and Zn (Brieger et al. 1992; Landberg and Greger 1994; Ostman 1994; Riddel-Black 1994; Nissen and Lepp 1997; Punshon and Dickinson 1999; Vyslouzilova et al. 2003). In addition, willows are often unaffected or stimulated by Cd (Punshon and Dickinson 1999), which may explain why, in this present study, both species of willow were able to survive in soil where Cd concentrations are considered phytotoxic (Table 1).

Foliar concentrations are important indicators for Cd, Zn, and Cu in site-specific ecological risk assessment, since these data are also indicative for food web transfer of metals (Vandecasteele et al. 2004). Larison et al. (2000) reported that ingestion of even trace quantities of Cd (2.63 mg kg^{-1}) from willow leaf buds, new shoots, and stems sampled in the Southern Colorado Rockies influenced not only the physiology and health of white-tailed ptarmigan (*Lagopus leucurus*), but the demographic and distribution of the species as well. Results from sampling the ptarmigan showed toxic Cd concentrations in the kidneys in 44% of adult birds which resulted in irreversible renal tubular damage (Larison et al. 2000). In addition to ptarmigan, herbivores and browsers such as mule deer (*Odocoileus hemionus*), beaver (*Castor canadensis*), elk (*Cervus elaphus*), moose (*Alces alces*), snowshoe hare (*Lepus americanus*), and various livestock are known to consume large quantities of willow (Warren 1942; Roath and Krueger 1982; Armstrong 1987; Larison et al. 2000). Concentrations of Cd in the leaves of Geyer and mountain willow in this present study (6.56 and 10.16 mg kg^{-1}, respectively) were higher than those found in the ptarmigan study, indicating a significant threat to wildlife in the area.

CONCLUSIONS

Results from this study indicate that substantial variability exists in metal uptake between the two dominant willow species found near Leadville, Colorado. This variability complicates a straightforward definition of what are normal and toxic plant concentrations. The increasing concentrations of most elements throughout the growing season within the leaf tissue of both willow species indicates that they translocate absorbed metals to their leaves allowing them to shed the metals during senescence, consequently limiting metal accumulation in roots or stems (Baker 1981). However, only foliar metal analysis was included in this study and to better understand the mechanisms for tolerance and metal accumulation in these two willow species, root and stem sampling would be required.

Although the exact mechanism of metal tolerance is not known in this study, the phytotoxic levels of Cd and Zn in the leaf tissue and phytotoxic levels of Cd in the soil had no apparent detrimental effects on the growth of Geyer and mountain willow given that both willow species were vigorous and healthy. Even though these concentrations do not affect the willows themselves, future research is needed to determine if they are creating detrimental effects on livestock and wildlife in the area.

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ENVIRONMENTAL INTEGRATION OF HIGH ALTITUDE ROADS.
THE SLOPE RESTORATION PROJECT OF "LA LOMBA – BRAÑAVIEJA" ROAD.

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ABSTRACT

Today the environmental integration of new roads constitutes a challenge and an obligation. The first step for a correct integration is the environmental study while elaborating the previous studies, preliminary designs and projects. Many technical aspects must be considered during execution. Among them, we can point out hydrotechnics, bioengineering applications, all kinds of topographic remodeling and operative and planning criteria of the restorer's works.

In this work, we present the recent experience of high altitude revegetation. We explain the project of bank restoration over a singular road in the Cantabria region of Spain where an environmental approach was used to achieve the ecological and functional integration of the banks, beyond an esthetical or visual improvement.

This new operating focus seeks the environmental excellence of the civil works, and constitutes one of the action models of the *Consejería de Obras Públicas y Vivienda del Gobierno de Cantabria*. This focus considers the forecasts of the recent Decree 61/2004 which defines the Singular highways of Special Landscape and Ecological Protection by crossing National Reserves.

AN INTERDISCIPLINARY PROJECT FOR REVEGETATION WITH NATIVE SPECIES IN THE FRENCH PYRENEES MOUNTAINS

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ABSTRACT

The Ecovars 2 project (2005-2007) has been created to extend and enhance the previous project developed by the Pyrenean Botanic Conservatory during the past seven years to promote restoration of damaged sites in the Pyrenees Mountains with native plants. The aim of this project is to promote the use of native species in land reclamation in order to ensure flora conservation. This project gathers different types of organizations: the Botanic Conservatory, an agronomic research institute (INRA of Toulouse) and an agricultural service (SUAIA Pyrenees) in order to carry out simultaneously:

The expansion of expertise and technical support for the revegetation of damaged Pyrenean sites,
The organization of training courses and the elaboration of a practical guide for revegetation with native species in the alpine and sub-alpine Pyrenees,
The development of agricultural aspects of native seed production with on-farm experimental multiplication of plant materials,
The finalization of genetic studies about plant adaptation to local environments,
A collective reflection in a scientific workshop in order to define “collect and use” zones for the species produced and used in land reclamation.

INTRODUCTION

The Pyrenees mountains separate France from Spain and stretch for 450 km, linking up two countries and a principality (Spain, France and Andorra), and two seas (the Atlantic Ocean and the Mediterranean Sea). They reach their summit upon the Aneto Peak (3404 m), above the snowline.

High altitude revegetation in the Pyrenees needs to address the same problems that occur in other mountainous areas. Specifically, topography and climate of such areas make revegetation very challenging with long natural recolonization times on damaged sites, like ski runs, roadsides and other infrastructures, particularly in alpine environments (Dinger and Bédécarrats 2001, Krautzer and Wittmann 2005). The disturbances strip away the topsoil, resulting in erosion and lack of landscape integration. In order to prevent these risks, revegetation is generally carried out, but in the Pyrenees this is always accomplished with cheap and alien seed mixtures at all elevations (Cassan et al. 2003). At first, these practices create a risk of technical failure by the limited vegetation cover and the high nutritional requirements of such species during a long period. Next, those alien species pose a threat for local flora and local ecosystems (Gustafson et al. 2004, Lambinon 1997, Simberloff 2003).

In the early 1990’s, the Pyrenean National Park brought to the fore the need for native seeds for restoration in its central area (mainly in the subalpine belt). Thus, the Park assessed the potential for the use of native seeds and began some experimental studies. By the end of the 1990’s, with the Pyrenean Botanic Conservatory assistance, the information from these studies led to an expertise and technical support activity for all the French Pyrenees, in the subalpine and alpine belts. During this nearly seven year time period, the potential for the use of native seed was evaluated in collaboration with practitioners and nature conservancy authorities. Wild collection of native seed is not sufficient to supply the projected need for these species. For revegetation of projects above 1200 m on the French side of the Pyrenees, it is estimated that approximately 40 tonnes of native seed each year would be required (Malaval 1998). Thus, commercial growers of native seed were needed to provide enough native seed for sowing in National Parks and ski resorts in Pyrenees.

From a market study (Malaval 1998), ecological observations, expertise, technical support, genetic and taxonomic studies (Lauga et al, In Press, Malaval-Cassan et al., In Press) and agronomic experiments, we decided to initiate the creation of a native seeds organization, with the collaboration of other institutes.

PARTNERS AND PARTNERSHIPS

Today, the Ecovars 2 project consists of collaboration between three organizations, acting together in order to promote the use of native seeds and to build scientific and technical tools that allow it. The exact title of the program is "**To conserve, to restore and to promote Pyrenean flora during landscape planning in Pyrenees, in a sustainable way**". Those three organizations are:

1. The Pyrenean Botanic Conservatory is a local public entity, acting for wild flora knowledge and conservation in the Pyrenees. Since the beginning of the Ecovars 2 project, the Conservatory has been the project coordinator and coordinates participation of many professional groups and nature conservancy authorities.

2. The Pyrenean Utility Department for Agriculture (SUAIAP) is a professional public agency, acting in particular for exploration, innovation and emergence of development projects in mountain areas. This organization focuses on development of seed production for ecological restoration.

3. The National Institute of Agronomic Research (INRA) of Toulouse works on research programs linked to agricultural or natural grassland systems. This institute organizes research on locally adapted native species selected for revegetation, linking genetic analysis and stakeholder interests. This work will aid in developing tools for use in local restoration projects.

The three institutions work together to develop partnerships with a variety of organizations and stakeholders, in order to accommodate their revegetation interests and their desire to participate in the program:

Europe, French State, and public institutions as Languedoc-Roussillon, Aquitaine and Midi-Pyrenees Regions, provide financial support for this program;

Ski resorts, forestry services, seed producer syndicates, seed multiplication experts, private environment offices act as active and interested partners;

Nature conservancy authorities, Pyrenean country planning delegation, Pyrenean tourism comity, ski resort syndicates, road and forestry pistes planning institutes, associations for environment protection, and other revegetation stakeholders participate in a monitoring network, in order to evaluate the Ecovars 2 project and guide the program development.

These partnerships form the base of an interactive approach for the innovative kind of revegetation practices we will define together, in order to reach Ecovars 2 objectives (Malaval et al. 2005).

EXPERTISE, INFORMATION AND COORDINATION

This part of the program focuses on the activity of the Botanical Conservatory during the last five years. From studies and experiments, with stakeholder's participation, objectives and opportunities were defined in order to address the issue of the lack of native species for use in restoration projects.

Coordination of such program includes or is coupled with:

- communication, information of all partners and stakeholders,
- network animation, from seed prescribers to seed users,
- expertise and technical support, consultant for land reclamation in Pyrenean sites,
- development of a list of relevant native species for revegetation in the subalpine and alpine belts,
- preparation of stakeholders training,
- collective creation of a technical guide for good practices in land reclamation,
- preparation of a certification process.

An important part of the expertise and technical support activity points out the necessity of a better integrated approach for revegetation. An important task consists, then in considering all stages of landscape planning, from excavation, to soil restoration, before and after revegetation, in order to provide successful restoration.

All these tasks are achieved in collaboration with revegetation practitioners and stakeholders, in order to enable integration of future native seed production into the market. A Website will soon keep public and people in the trade informed about the progress of the program.

SEED PRODUCTION AND UTILIZATION

The main focus of the project is the production of native seeds for revegetation of alpine and subalpine belts in Pyrenees. It includes:

- development of a technical guide for seed production with the farmers-multiplier's profession,
- seed collection in altitude areas,
- assessment of adapted methods for seed multiplication at high elevation (500 to 1000 m),
- prospective study of the future multiplication organization,
- production of multiplying young plants in altitude areas,
- development of a network of seed multiplying farmers.

In 2005, the first seed collections for multiplication were gathered, allowing for production of young plants for spring 2006. The first young plants of the perennial species used in this program are going to be transplanted in fields in April and May 2006, in order to set up the first fields for production of native species this year. The first species that have been collected seeds include:

*Festuca eskia**
*Festuca gauteri**
Festuca airoides
*Carduus carlinoides**
Lotus alpinus
Trifolium alpinum
Senecio leucophyllus
Rumex scutatus
Deschampsia flexuosa

* Pyrenean endemic or Pyreneo-cantabric endemic species.

All these species show propagation characteristics and constraints that are not familiar to traditional Pyrenean farmers. The main task consists in developing techniques to produce seeds in large quantities and at the same time conserving biodiversity.

Previous genetic studies (Lauga and al., In Press) assessed the first collection of seeds in thePyrenees and zones of utilization of such seeds after multiplication. For most of the species, occidental and oriental origins of Pyrenean seeds need to be kept separate during their utilization, because the geographic pattern of neutral diversity showed differences between those regions. The differences in genetic diversity could be related to the existence of different lineages in the post-glacial recolonization processes of the massif during the Pleistocene period (Malaval-Cassan et al. 2005).

The objective of the agricultural part of the program is to develop an independent organization of native seed production, in the Pyrenees, according to the procedures set up in Ecovars2 (seed transfer zones, certification process, practical details for seed collection and biodiversity conservation).

RESEARCH AND DEVELOPMENT

The research program includes:

- determination of genetic structure of plant populations used in restoration, like *Festuca eskia*,
- study of environmental and human factors that could influence population structure.

Evaluation of results from reciprocal transplants has begun in the French Pyrenees. Evaluation will include genetic analysis, and stakeholders interview.

Our active participation approach to restoration using the stakeholders evaluation anf review will allow us to propose relevant tools for restoration practice. For example, this program will contribute to designs for consensual seed transfer zones for plants used in revegetation in the French Pyrenees (Gonzalo-Turpin et al. 2005).

CONCLUSION

This multi-structural approach will allow us to coordinate the technical, scientific and human elements needed to promote commercial development of sources of native seed. We will also be able to define and develop resources for the pricing of the produced seeds.

Moreover, we will promote the use of native species in the Pyrenees, following a precise practical guide, in order to enhance the success of high altitude revegetation.

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THE IMPACT OF AVAILABLE NITROGEN IN MINE SITE REVEGETATION: A CASE STUDY IN THE COEUR D'ALENE (IDAHO) MINING DISTRICT

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ABSTRACT

A variety of soil amendments were evaluated for suitability in revegetation of waste rock piles in the Coeur d'Alene Mining District of northern Idaho. The amendments included biosolids, composts, log yard wastes, and two liquid-based organic treatments. Initial available N (ammonia plus nitrate) values varied significantly with the type of amendment, ranging from 5 ug/g in the liquid-based soil treatments to >850 ug/g in the log yard waste-urea fertilizer treatment. As would be expected, high available N increased the potential for high runoff N. However, within the set of high N plots, plant density was a significant factor in limiting runoff N concentrations. That is, plots with high available N often supported dense vegetation, which tended to decrease runoff N as a result of high plant uptake. The level of available nitrogen also had a strong impact on plant frequency vs. density, species distribution, and the extent of unseeded vegetation. For example, high N amendments promoted a high frequency/low density vegetation profile that was dominated by grass species and contained a low content of unseeded vegetation. Conversely, the low N amendments promoted a low frequency/high density profile with a diverse grass-legume mixture but also a greater density of weeds.

INTRODUCTION

The Idaho Department of Environmental Quality (IDEQ) initiated a study to identify alternative approaches for reclamation and revegetation of waste rock piles in the Coeur d'Alene Mining District. As is the case in many reclamation projects, quality topsoil is a limited resource. Hence, there is an ever present need to identify topsoil alternatives that are locally available and can promote a self sustaining vegetative cover. The overall goal of this study is to examine soil amendments, including biosolids, composts, log yard wastes, and two liquid-based soil treatments for efficacy and cost effectiveness in site revegetation.

A key variable in these amendments is nitrogen availability. The essential role of nitrogen in amino acid and protein formation, as well as enzyme function and chlorophyll synthesis, has been well documented (Tisdale and Nelson 1975; Follett et al. 1981). However, the impact of nitrogen in mine site reclamation goes beyond basic plant nutrition considerations. The dynamic nature of the nitrogen cycle and its intricate association with soil organic matter and soil biota influences a host of secondary properties or processes including water quality, plant diversity, and the ability of seeded vegetation to compete with weed species. For example, Reever-Morghan and Seastedt (1999) reported decreased knapweed biomass as a result of reduced N availability. Reduction of available nitrogen was also used to suppress growth of introduced grasses, which improved the establishment of native species (Wilson and Gerry 1995). Redente et al. (1992) found that high N levels produced more biomass in early seral species but, as N availability decreased, late seral species appeared to gain a competitive advantage. Additional studies have shown that high nitrogen availability leads to higher overall productivity but lower diversity (Willems and van Nieuwstadt 1996; Baer et al. 2003).

Given the importance of these impacts to the long-term success of restoration projects, additional studies (particularly those conducted under differing environmental and topographic conditions) are useful. The demonstration plots installed to study topsoil alternatives provided an ideal opportunity to evaluate available N impacts using side-by-side comparisons. The specific objectives of this study were to evaluate the impacts of available N on 1) runoff nitrogen concentrations, 2) vegetation frequency, density, and diversity; and 3) density of unseeded vegetation.

METHODS

Site Description

The Silver Dollar Mine site is located west of Osburn, Idaho (47º 30.22' N; 115º 59.39' W). The site is dominated by a waste rock pile produced during mine development and sorted from the ore during the mining process. Milling and smelting activities took place off-site so heavy metal concentrations are a minor issue for plant growth relative to low fertility. The waste rock pile rests on a north-facing slope at an elevation of about 2500 feet. Average total monthly precipitation ranges from 1.5 inches in July to 4.5 inches in November, with a total annual precipitation of 38 inches. Average monthly temperatures are 32.9/21.3ºF (max/min) in January and 78.6/47.2ºF in August.

Site Preparation/Plot Installation

The waste rock pile was regraded to a 2:1 (H:V) slope and ten plots (20' X 100') were installed with a berm (3' X 2') separating each plot. Runoff flumes and an erosion trap were installed at the bottom of each plot. The western- and eastern-most plots were reserved for controls; the remaining plots were assigned to participants on a random basis. Project participants (Table 1) were solicited and selected by IDEQ.

Installation of the plots began 25 September 2002 and concluded 23 October 2002. A brief description of amendment materials and application rates is listed in Table 1. Each participant selected their amendment rate and method of application and each plot was seeded, either by hand or by hydroseeding, using a standardized seed mix (Table 2).

Plot Assessment

The plots were inspected monthly from April through August during 2003, 2004, and 2005. Percent germination and relative growth (leaf stage) were evaluated at the beginning of each growing season. In addition, a qualitative assessment of leaf color was made as this can provide clues to nutrient sufficiency/deficiency and plant stress due to diseases and pests. Uniformity of coverage was also noted for each plot.

Plant coverage was assessed using two methods. Plant frequency was determined using a Cover-point optical projection scope (ESCO Associates, Boulder CO). One hundred points were recorded at 1 m intervals along a randomly located transect in each plot. Each point identified an individual plant, rock, bare soil, or litter. Plant density was assessed at two sampling points per plot, 10 m in from the bottom and top of the plot. The specific location of the sampling point was randomly selected - the observer faced away from the plot and tossed a 1-m^2 PVC hoop over their head into the plot. Each individual plant within the hoop was tallied and identified, including plants that were not a component of the original seed mix. The mean value of the replicate density assessments is reported in the following tables and figures.

Table 1. Demonstration plot amendments and project participants.

Plot	Amendment	Affiliation	Rate	Available N (lb/ac)
A	Control (topsoil)	IDEQ	40 yd^3 of topsoil was spread to a depth of approximately 6”	20
B	Biosolid + Woodash (0.75:1)	Coeur d'Alene Wastewater Treatment Plant	26 yd^3 of Class B biosolids mixed with wood ash (0.75:1) was spread to a depth of approximately 4”	2668
C	Potlatch Log Yard Waste	Potlatch Corp.	Log yard fines (<3/4”) were mixed with urea fertilizer (10 % v/v); 48 yd^3 was spread to a depth of approximately 6”	1832
D	Kiwi Power	Quattro Environmental, Inc.	Fertile Fibers Plus, Kiwi Power, Strong Hold + Tacker and Atlas Soil Lock was mixed and applied using the hydroseeder	24
E	Eko Compost	Eko Compost	20 yd^3 of compost was spread to a depth of approximately 4”	348
F	Glacier Gold Compost	Glacier Gold, LLC	20 yd^3 of compost was spread to a depth of approximately 4”	49
G	Biosol	Rocky Mountain Bio Products	83 lb Biosol Mix (7-2-3) plus 5 lb Wood Fiber Mulch seed mix was applied using the hydroseeder. Wheat straw was spread over plot and 4 lb Guardian Tackifier applied.	20
H	Glacier Gold Log Yard Waste	Glacier Gold, LLC	20 yd^3 of log yard waste was spread to a depth of approximately 4”	53
I	Biosolid + Woodash (1:1)	Coeur d'Alene Wastewater Treatment Plant	26 yd^3 of Class B biosolids mixed with wood ash (1:1) was spread to a depth of approximately 4”	918
J	Control (fertilizer)	IDEQ	50 lb of fertilizer (16-16-16), seed mix, and tackifier were applied with the hydroseeder. Bluegrass straw was applied as a mulch on bottom-half of plot	14

Table 2. Seed mix used on the Silver Dollar Demonstration Plots.

Common Name	Scientific Name	Rate (lb PLS/ac)	Weight (%)	Minimum (%)
Slender wheatgrass	*Elymus trachycaulus* ssp. *trachycaulus* var. Revenue	14 lbs	22.3	21.9
Idaho fescue	*Festuca idahoensis* var. Joseph	8 lbs. 7 oz	13.4	13.2
Sheep fescue	*Festuca ovina* var. Covar	7 lbs	11.1	10.9
Mountain brome	*Bromus marginatus* var. Bromar	7 lbs. 11 oz	12.2	12.0
Meadow brome	*Bromus biebersteinii* var. Paddock	8 lbs. 7 oz	13.4	13.2
White Yarrow	*Achillea millefolium*	11 oz	1.1	1.1
Blue flax	*Linum lewisii* var. Appar	4 lbs. 3 oz	6.7	6.6
Rocky Mountain penstemon	*Penstemon strictus*	1 lb. 6 oz	2.2	2.2
White dutch clover	*Trifolium repens* L.	8 oz	0.8	0.8
Canada bluegrass	*Poa compressa*	11 oz	1.1	1.1
Big bluegrass	*Poa ampla* var. Sherman	1 lb. 7 oz	2.3	2.3
Canby bluegrass	*Poa canbyi* var. Canbar	1 lb. 6 oz	2.2	2.2
Cicer milkvetch	*Astragalus cicer*	7 lbs.	11.1	10.9
Fireweed	*Epilobium angustifolium*	1 oz	0.1	0.1
Weed seed				0.5 (Max)
Inert and other crop				1.5 (Max)

Surface runoff was collected monthly in 2003, 2004, and 2005, and each sample analyzed for ammonia-N, nitrate-N, and orthophosphate. In 2003 and 2005, a composite (3x) soil sample was collected from each plot. A standard fertility test (ammonia-N, nitrate-N, available P and K, organic matter, and pH) was determined for each sample. All laboratory work was conducted at the University of Idaho Analytical Sciences Laboratory.

RESULTS

Waste Rock Properties (pre- and post-amendment)

Most of the amendments decreased the plot pH relative to the initial value of 8.3 (Figure 1A). The pH of the amended plots ranged from 6.3 to 8.3 with the 1:1 woodash/biosolid mixture exhibiting the highest pH. Overall, the pH was relatively consistent among the amended plots throughout the study period. The organic matter content varied from ~1% in the controls and liquid-based amendments to 15-34% in the solid-based amendments (Figure 1B).

Each of the amendments significantly increased the available P and K content, with the extent of increase being strongly dependent on the nature of the amendment (Figure 1C, 1D). Available P values (sodium acetate extractable) ranged from <2 to >600 ug/g while available K[1] ranged from 80 to 1000 ug/g. To put these numbers into perspective, available P and K levels in excess of 8 and 100 ug/g, respectively, are considered sufficient for non-irrigated legume and legume-grass pastures in northern Idaho (Mahler 2005). Thus, each of the amended plots contained adequate to excessive P and K relative to typical plant requirements. In several plots, the application rates are well in excess of vegetation needs, thereby increasing the potential for leaching. This is indeed the case as is shown by the runoff P results (Figure 2).

The nitrate-N level in the unamended soil-waste rock was 0.7 ug/g while the amended plots exhibited nitrate-N concentrations ranging from <5 to >800 ug/g (Figure 1E). Similarly, ammonia-N was initially low and varied significantly among the amended plots, ranging from <4 to >60 ug/g (Figure 1F). These values are equivalent to 20 - 3500 lb available N per acre [available N (lb/ac) = (ug/g ammonia N + ug/g nitrate-N) x 4].

Nitrogen in Surface Runoff

As would be expected, runoff nitrogen concentrations closely reflected the available ammonia and nitrate content of the amendments. The highest runoff ammonia- and nitrate-N concentrations (5.3 and 34 mg/L, respectively) were observed in the Potlatch Log Yard Waste (Figures 2A and 2B). This is undoubtedly due to the very high rate of urea fertilizer (10% v/v) applied to the log yard waste, which resulted in an extremely high available N content (~3500 lb/ac) in 2003. Significant N runoff was also observed in the Eko Compost and Biosolids + Woodash II plots. Despite having very high available N (~2500 lb/ac), runoff ammonia- and nitrate-N concentrations were low in the Biosolids + Woodash I plot (Figures 2A and 2B).

In general, runoff N concentrations are decreasing over time. This is most likely due to the combined mechanisms of plant uptake and leaching. The greatest decrease was observed in Plot C (Potlatch Log Yard Waste + Urea Fertilizer). It is important to note that vegetation was almost nonexistent on this plot in 2003, most likely as a result of ammonia phytotoxicity. The lack of vegetation eliminated plant uptake as a runoff control mechanism, thereby facilitating high levels of ammonia- and nitrate-N in the 2003 runoff.

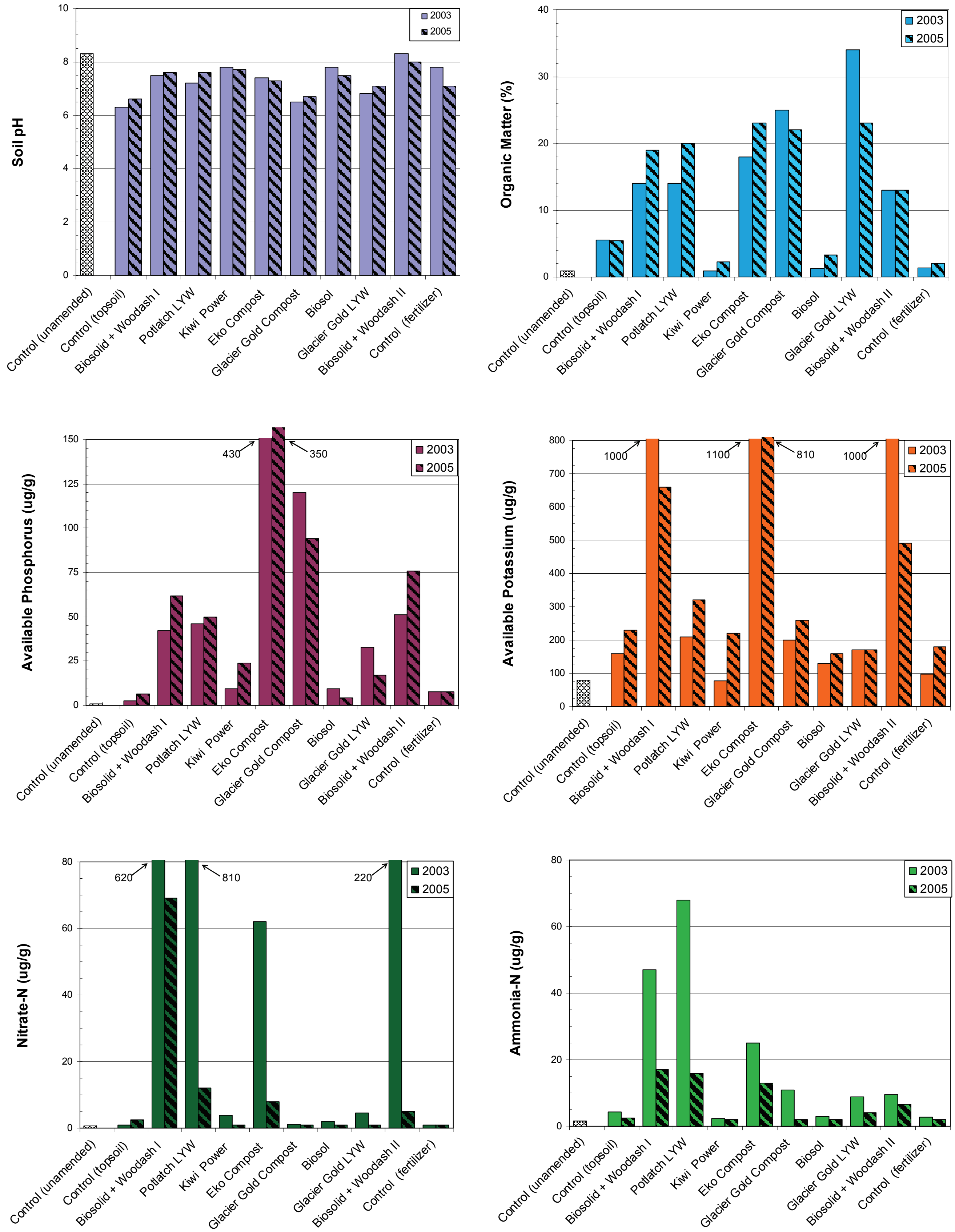

Figure 1. Comparison of soil parameters in each plot in 2003 and 2005: (A) soil pH, (B) organic matter, (C) available P (sodium acetate-extractable), (D) available K, (E) nitrate-N, and (F) ammonia-N.

It is worthwhile to compare the nitrogen dynamics of the Potlatch Log Yard Waste to those of the Biosolids and Eko Compost plots. Each of these plots contained very high levels of available N in 2003 (Table 1). However, significant differences in vegetation density were observed among these plots. Vegetation was nonexistent on the Potlatch plot in 2003 (Figures 3A, 3B), most likely as a result of ammonia phytotoxicity. This lack of vegetation eliminated plant uptake as a runoff control mechanism. In contrast, the Eko and Biosolids plots exhibited heavy vegetative growth, primarily of wheatgrass – a species known for its ability to sequester N. This comparison illustrates the important role of vegetation, particularly N accumulating grass species, in serving as nutrient sinks when nitrogen-rich amendments are used in revegetation projects.

Nitrogen Impacts on Vegetation Coverage

Low N amendments (Kiwi Power, Glacier Gold, Biosol) promoted a low frequency/high density vegetation profile in 2003 (compare Figures 3A and 3B). Field assessments indicated that each of these plots is supporting large numbers of plants; albeit much smaller in stature as compared with neighboring high N plots. Thus, in terms of sheer numbers of plants per unit area, these plots exhibit relatively high plant densities. It should also be noted that plant frequency and density increased in most plots between 2003 and 2005. Plots receiving high N amendments promoted a high frequency/low density vegetation profile. The low plant density values are potentially misleading as these plots are not exhibiting poor performance. To the contrary, these plots exhibited excellent coverage with very large plants relative to the same species growing on the low N plots. It appears that the sheer size of the vegetation is a limiting factor for density in the high N plots. These observations suggest that density data alone can misrepresent the overall quality and performance of a given plot.

Nitrogen Impacts on Plant Diversity and Establishment of Unseeded Species

As Figure 3A illustrates, between 75 and 90% of the total vegetation is accounted for by grass species in the high N plots. A more detailed vegetative assessment of these plots identified wheatgrass as the dominant species (McGeehan, 2006). In contrast, plots receiving lower N inputs (i.e. Controls, Kiwi Power, and Glacier Gold plots) exhibited a greater diversity of forbs (including yarrow, clovers, and milkvetch) mixed with the grasses. This correlation of high N availability, grass dominance, and reduced diversity is consistent with reports by Willems and van Nieuwstadt (1996) and Baer et al. (2003). This relationship has also been reported for grass-legume pastures, where high rates of N (e.g. >50 lb N/ac) lower the legume percentage relative to grasses (Mahler, 2005).

The incidence of unseeded vegetation also varied significantly between the plots, and was strongly associated with the density of grasses. This relationship is clearly illustrated when comparing the grass:forb ratio to the density of unseeded vegetation (Figure 4). A grass-dominated vegetative profile, as observed in the high N amendments, results in a high grass:forb ratio, and a low density of weeds. Greater diversity and a lower grass:forb ratio were observed in the low N plots with a concurrent increase in weed density (primarily knapweed, sweet clover, and black medic).

Several mechanisms have been proposed to explain the relationship between high nutrient availability and low species diversity. Once nutrient limitations are removed, diversity is believed to be controlled by competition for light as a result of dense above-ground biomass, as well as above- and below-ground competition between neighboring roots and shoots (Wilson and Tilman 1991; Rajaniemi 2002; Baer et al. 2003). Additional factors in these competition-based mechanisms include accumulation of surface residue and allelopathy (Grime 1973). Given the range in N availability examined in this study, it appears likely that nutrient competition was a determining factor for species diversity in the low N amendments. Furthermore, these plots supported small and sometimes stunted vegetation with patchy coverage. These characteristics favored the establishment of the invasive weed species. The high N plots effectively promoted growth of taller and denser grass species, thereby triggering light competition, root and shoot interactions, and possibly other competition mechanisms.

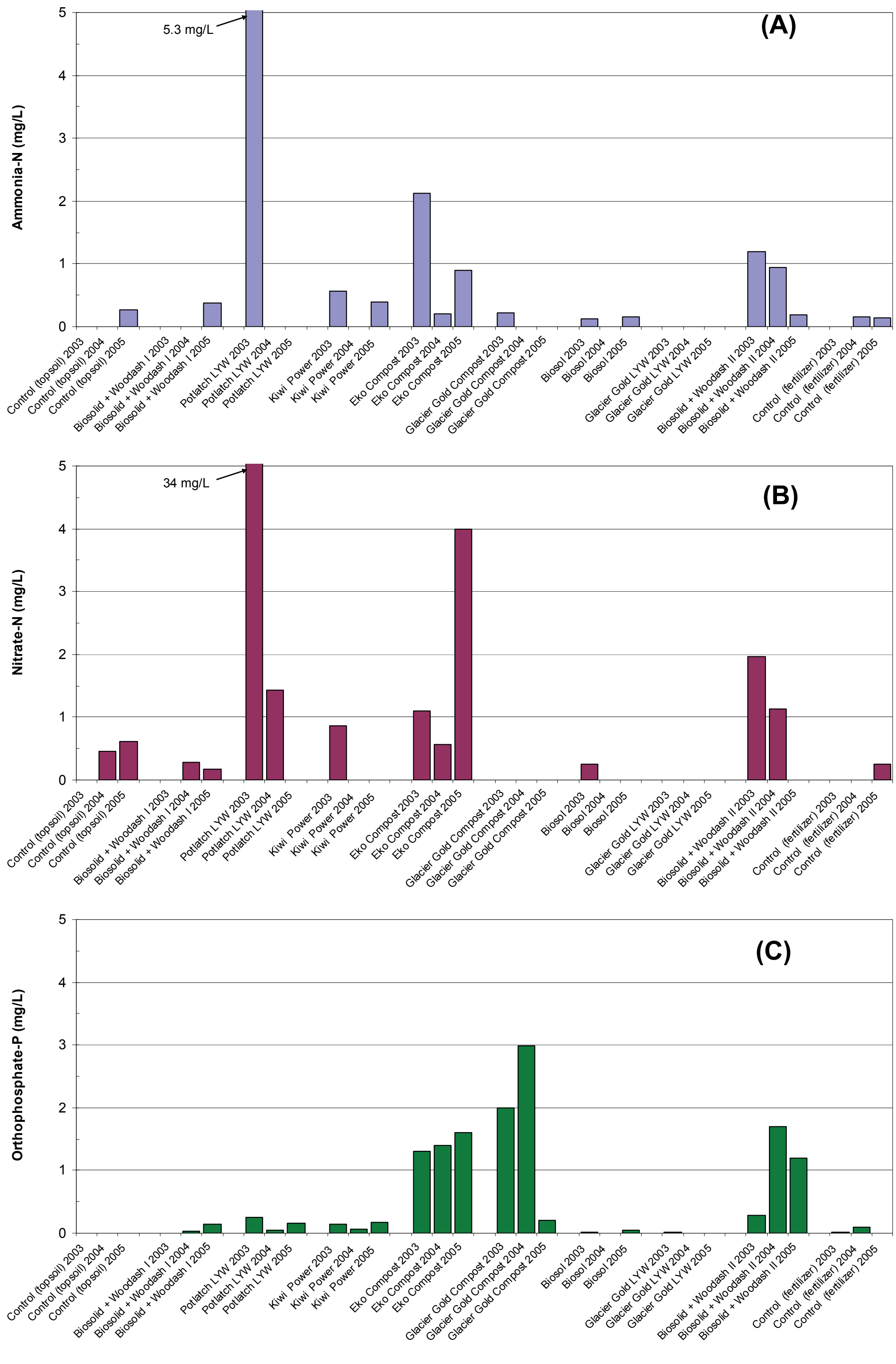

Figure 2. Runoff concentrations of (A) ammonium-N, (B) nitrate-N, and (C) orthophosphorus in 2003, 2004, and 2005.

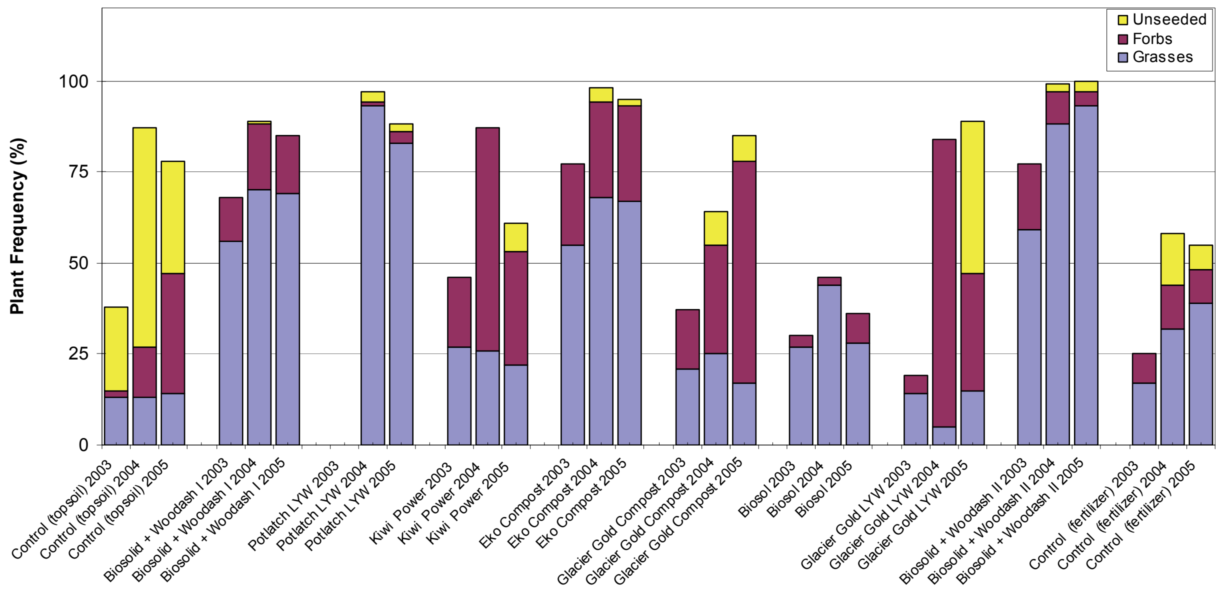

Figure 3A. Comparison of plant frequencies in 2003, 2004, and 2005 across all plots.

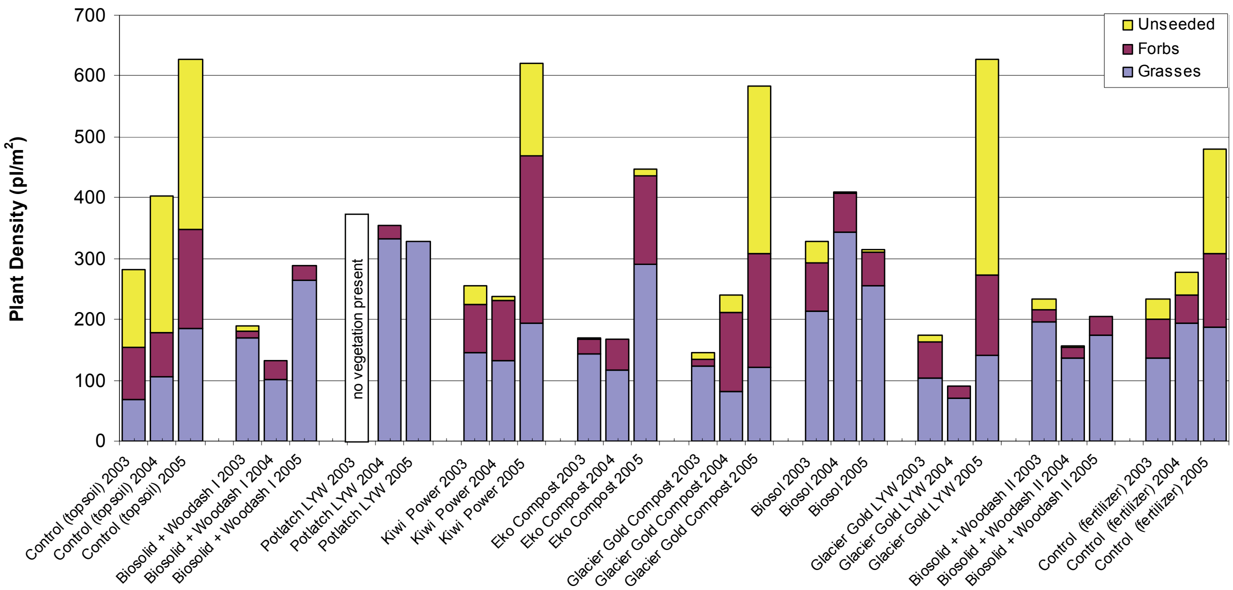

Figure 3B. Comparison of plant densities in 2003, 2004, and 2005 across all plots.

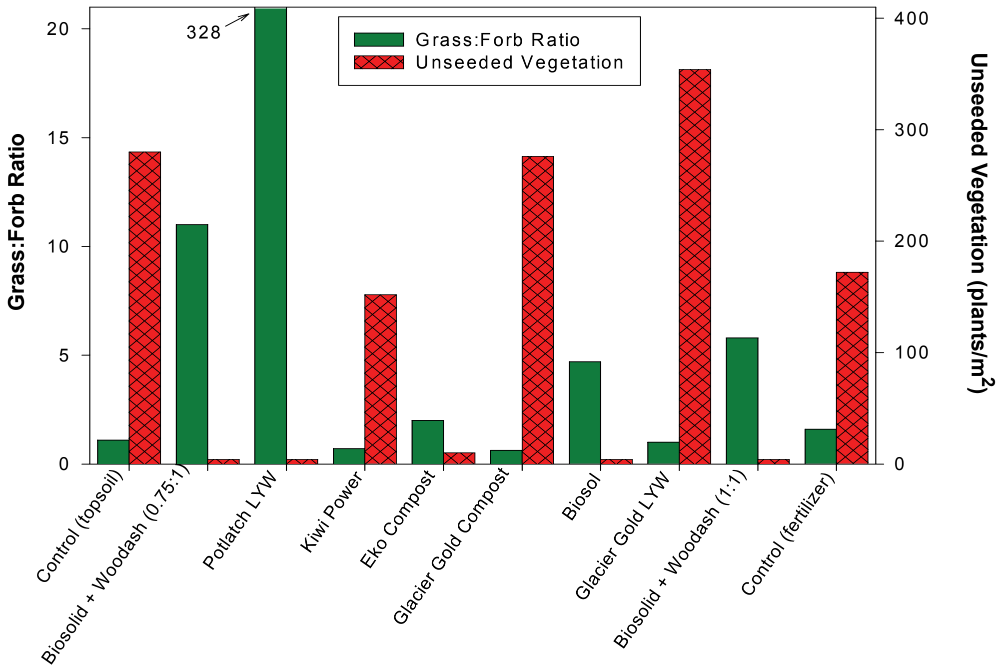

Figure 4. Comparison of grass:forb ratios and unseeded vegetation densities in 2005.

CONCLUSIONS

Aside from problems associated with heavy metals, a key consideration in mine site revegetation is low soil fertility. For this reason, nitrogen fertilization is a necessary practice. In this study, the nitrogen input associated with each amendment produced a significant positive growth response in terms of total plant frequency and density. Ammonia- and nitrate-N concentrations in surface runoff were significantly increased by the amendments. Not surprisingly, amendments with the highest available N were associated with the highest runoff N. However, within the high N plots, plant density was a significant factor. Runoff N was much higher in the plot with low plant density compared with plots exhibiting higher vegetative coverage. Thus, the role of plant biomass as a nutrient sink can be an important factor, particularly when using nutrient-rich soil amendments.

Another compelling impact of N availability was seen in the species distribution and extent of unseeded vegetation, where the level of available N clearly selects for a distinct vegetative profile. High N amendments promoted a grass-dominated, low forb profile, while the low N amendments promoted a more diverse grass-forb mixture. Furthermore, the grass:forb ratio was inversely correlated with the extent of unseeded vegetation. That is, a high ratio (grass-dominated) was associated with low densities of unseeded vegetation. Thus, it appears that high levels of available N provide grasses with a competitive advantage relative to forbs. This has the desirable outcome of controlling unseeded vegetation but at a cost of low species diversity.

It is important to temper these conclusions with the recognition that this was a relatively short-term (3 year) study. Significant changes in the soil media (amended waste rock) and plant communities occurred over the course of this study, and it is likely that these changes will continue. However, it is clear from this and other studies that nitrogen impacts to the entire soil-plant-water system must be considered when evaluating the overall success of a revegetation project.

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REVEGETATION OF THE ROCKY FLATS, COLORADO, SITE

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ABSTRACT

The Rocky Flats, Colorado, Site is a U.S. Department of Energy facility located between Boulder and Golden along the Front Range of Colorado. Cleanup and closure of the former nuclear-weapons component-production facility involved the removal of an Industrial Area the size of a small city. In late 2005, cleanup and closure activities were completed. Disturbed areas were reseeded with native plant species to protect the soil from erosion, maintain water quality standards, and re-establish habitat. Revegetation activities included large-scale seedbed preparation, seeding, and erosion control of upland, riparian corridor, and wetland areas. Two new landfill covers were also seeded. Native seed mixes were developed based on the species common in the native plant communities surrounding the former Industrial Area. Consultations with the U.S. Fish and Wildlife Service addressed disturbances to Preble's meadow jumping mouse habitat areas and with the U.S. Environmental Protection Agency and U.S. Army Corps of Engineers about wetland areas. Re-establishment of disturbed Preble's habitat areas and the creation of several wetland areas in the former Industrial Area drainages are ongoing as mitigation for unavoidable disturbances resulting from cleanup activities. Management and monitoring will be conducted to ensure success.

INTRODUCTION

The Rocky Flats, Colorado, Site (Site) is a U.S. Department of Energy (DOE) facility located between Boulder and Golden, along the mountain front. For over 40 years during the Cold War, the Site produced plutonium trigger components for nuclear weapons. In the early 1990's, site operations ceased and work at the Site shifted to cleanup and closure activities. In October 2006, Site cleanup and closure activities were completed.

The Site was once a small city with an infrastructure that included a fire department, medical facility, water treatment plant, waste water treatment plant, power substations, and numerous administrative buildings and production facilities (Figure 1). As part of the cleanup and closure activities, the buildings and infrastructure in the Industrial Area (IA) were decommissioned and demolished to prepare the way for returning the Site to a more natural ecological state. The landscape in the IA was recontoured to resemble the topography found in the surrounding undisturbed native plant communities outside the IA (an area called the Buffer Zone; Figure 2).

Though most people are unaware, the Buffer Zone at the Site contains a wealth of biological diversity that is becoming increasingly rare along the Front Range with increasing development and urban sprawl. The Preble's meadow jumping mouse (*Zapus hudsonius preblei*), a federally listed threatened species under the Endangered Species Act occurs in the wooded/shrubland streamsides at the Site. Numerous large hillside seep wetlands, riparian wetlands, and ponds are present at the Site. The xeric tallgrass prairie grassland type at the Site is considered increasingly unique along the Front Range with a mix of both tallgrass and montane species. This rich biodiversity will be preserved as most of the Buffer Zone at the Site becomes the Rocky Flats National Wildlife Refuge. The previous IA will be retained under DOE control.


Figure 1. Industrial Area at the Rocky Flats Site in 1995.


Figure 2. Previous Industrial Area after cleanup and closure activities (Fall 2005).

Planning of cleanup and closure activities was planned to avoid disturbance to Preble's mouse habitat and wetlands where possible and minimize impacts when unavoidable. A Programmatic Biological Assessment was written and consulted on with the U.S. Fish and Wildlife Service (USFWS) for disturbances to Preble's mouse habitat. Unavoidable wetland disturbances were consulted on with the Environmental Protection Agency (EPA) and U.S. Army Corps of Engineers (USACOE). To offset the Preble's mouse habitat and wetland disturbances several sections of the re-configured stream drainages in the IA were designed to create/re-establish Preble's habitat and wetlands.

GENERAL REVEGETATION ACTIVITIES

Approximately 650 acres total were disturbed during cleanup and closure activities. Figures 1 through 4 illustrate Site conditions before and after cleanup activities were conducted from different vantage points. After final land configuration work was completed, disturbed areas were seeded with native seed mixtures. Where possible, areas were deep ripped to loosen compacted soils and at many locations Rocky Flats Alluvium was brought in as fill material and soil medium. Several different native seed mixtures were used dependent on the location of the revegetation area. Table 1 lists the species present in each type of seed mix. The flat upper pediment surfaces were seeded with native species typical of the xeric tallgrass prairie at the Site. Hillslopes were seeded with mixed grass prairie species common at the Site. Riparian/wetland areas were seeded with species adapted to the hydric conditions available along the streams and in the wetland areas. Due to project schedule constraints, seeding was conducted throughout the growing season as cleanup activities were completed at each location. Seeding was done by both broadcasting and drill seeding, dependent on the location.

Only graminoid species were seeded initially because it is anticipated that weed control with herbicides will be required during the initial vegetation establishment phase. After initial establishment of grasses, forbs and wildflowers typical the native prairies in the Buffer Zone at the Site may be seeded into the areas to add some diversity to these areas,. Volunteer groups have been conducting seed collecting to provide the Site with species that are not available commercially. It is anticipated that this activity will be continued and will provide additional species of forbs and graminoids for the revegetation areas.

After seeding, various erosion controls were installed to protect the soil from wind and water erosion, and protect water quality. Steeper slopes had erosion matting installed while other areas received different types of hydromulch (aspen fiber, Flexterra®). Most of the flat upper pediment surfaces were protected with crimped straw. At many locations straw wattles and hay bales were also used to protect drainages and other locations where additional erosion control measures were required.

Some of the wetland areas that were completed in early spring of 2005 also had willow stakes and cottonwood poles installed to help diversify the vegetation structure at these locations. Additional planting of stakes and poles is planned for spring 2006 to help continue to enhance these wetland areas. Monitoring and management of the revegetation locations will continue in the future to provide the best opportunity for successful vegetation establishment.

MITIGATION ACTIVITIES

To offset disturbances to Preble's mouse habitat and wetlands, mitigation activities were conducted through the use of both in-situ and new creation of habitat/wetlands. Typically mitigation is conducted to replace the loss of habitat/wetlands resulting from disturbances (development, urbanization) that permanently destroy or eliminate the pre-existing habitat. In those cases, mitigation is often done at another location where "new" habitat/wetlands can be created or established. In the case of Rocky Flats, the cleanup and closure activities were conducted to remove the man-made structures and disturbances, and return these areas to a more native state. In the process of removing

Table 1. Typical Seed Mixtures At Rocky Flats

Flat Pediment Top Locations
Agropyron smithii *Agropyron trachycaulum* *Andropogon gerardii* *Andropogon soparius* *Bouteloua curtipendula* *Bouteloua gracilis* *Buchole dactyloides* *Koeleria pyrimidata* *Sorghastrum nutans* *Sporobolus cryptandrus* *Stipa viridula*
Hillslope Locations
Agropyron dasystachyum *Agropyron smithii* *Agropyron trachycaulum* *Bouteloua curtipendula* *Bouteloua gracilis* *Buchole dactyloides* *Stipa viridula*
Riparian Edges
Agropyron smithii *Agropyron trachycaulum* *Andropogon gerardii* *Bouteloua gracilis* *Elymus canadensis* *Panicum virgatum*
Wetland Areas (actual species varied based on the location)
Agropyron smithii *Agrostis scabra* *Amorpha fruticosa* *Calamagrostis canadensis* *Carex lanuginosa* *Carex nebrascensis* *Eleocharis palustris* *Elymus canadensis* *Elymus trachycaulus* *Juncus balticus* *Juncus tenuis* *Juncus torreyi* *Panicum virgatum* *Poa palustris* *Scirpus acutus* *Scirpus americanus* *Scirpus validus* *Spartina pectinata*

Figure 3. Industrial Area looking west in 2003.

Figure 4. Same location as Figure 3 looking west in late summer 2005.
Note new wetland in foreground.

the man-made structures at the Site, some disturbance to Preble's habitat and wetlands was required to access and complete the cleanup activities. The end result however, was that is it expected that more Preble's mouse habitat and wetlands will be present at the Site, after cleanup and closure activities, than was present before. This will provide a benefit to the wildlife and other natural resources at the Site, since the previous IA will be returned to grassland, riparian streamside, and wetland habitat.

SUMMARY

The Rocky Flats, Colorado, Site cleanup and closure activities were completed in fall of 2005. Land reconfiguration activities transformed the old IA to a more natural landscape, similar to that of the Buffer Zone. Newly created prairie, riparian streamside, and wetland areas were seeded with native species typical of the Site to begin returning the former IA back to natural conditions. Continued monitoring and management of these areas will be conducted to provide the best opportunity for long-term success.

COST-EFFECTIVENESS ANALYSIS OF RESTORATION APPROACHES IN ROCKY MOUNTAIN NATIONAL PARK

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ABSTRACT

Rocky Mountain National Park has adopted guidelines that require the use of locally collected material in restoration projects to conserve the genetic integrity of plant populations. Areas disturbed through human activities are restored through natural revegetation, seeding, transplanting containerized material, or a combination thereof. Costs associated with these approaches vary considerably, emphasizing the need for an evaluation of the cost effectiveness of these methods in the development of late-seral plant communities following disturbance. Vegetative cover and species richness were compared for disturbed sites treated with different restoration methods and associated undisturbed reference areas. Community similarity indices were calculated for each disturbed/ undisturbed pair. Results show differences in the relative effect of treatments depending on effectiveness criteria, making a determination of an optimal restoration approach difficult. Similarity indices show seeded sites had the highest similarity while transplanted sites had the lowest similarity to undisturbed reference areas. Statistical comparisons showed that seeded sites had lower total and native species cover than other treatments, while species richness was not influenced by treatment. Cost-effectiveness analysis of treatments showed seeded sites are more cost effective than other treatments in creating communities with high total and native species cover, while minimizing the establishment of introduced species.

INTRODUCTION

Ecological restoration is a collection of approaches aimed at reversing habitat degradation, destruction, or damage as a result of various types of disturbances (Baldwin et al. 1994). As the recognition of the need for ecological restoration increases and the science of restoration ecology matures, methodologies and concepts that are important for successful restoration have been increasingly developed. Among the concepts crucial to the success of the restoration process are 1) determining realistic goals, 2) developing practical methods for implementing those goals, and 3) assessing the success of these strategies (Hobbs and Norton 1996).

Numerous anthropic disturbances have historically occurred within Rocky Mountain National Park (RMNP) prior to, and following its inception in 1915. Disturbances have included mines, sawmills, homesteads, camps, settlements, lodges, roads, ski slopes and lifts, a golf course, and an asphalt mixing plant. It has been the policy of the National Park Service (NPS) to acquire any in-holdings, remove structures not determined to be culturally significant, and restore these areas either through natural or artificial revegetation efforts to appropriate late-seral plant communities. In recent years, anthropic disturbances in RMNP have occurred mostly in areas of high human use, including the park utility areas, trails, trailheads, picnic areas, campgrounds, and parking lots. The characteristics of these disturbances vary greatly depending on the activities that took place, and thus the approaches to restoration in these areas cover a variety of natural and artificial restoration techniques.

NPS made a management decision in 1985 to use local genotypes of native plants for restoration projects when possible in order to preserve the genetic integrity of the native plant communities. The use of local genotypes in restoration of anthropic disturbances requires the collection of local plant material such as seed or vegetative plant material. Seed collected from local populations can be placed directly onto the site following disturbance or propagated in the park's greenhouse. Salvaged plants or containerized plant material propagated in the greenhouse can be transplanted onto disturbed sites. Since the decision to use only locally derived plant material, RMNP has used natural revegetation, seeding, transplanting containerized plant material, and a combination of seeding and transplanting to restore disturbances (Figure 1).

NPS goals for restoration of anthropic disturbances are: 1) site stabilization, 2) control of introduced species, 3) the creation of an appropriate plant community dominated by late-seral species. The long-term goal of the restoration of disturbed sites in RMNP is the creation of a plant community as similar as possible to the plant community that existed on the site prior to disturbance. The goal is for a disturbed area to "blend in" to the surrounding vegetation to the greatest extent possible. For this reason, adjacent areas of relatively undisturbed native vegetation are often used as a standard for comparison.

Several factors commonly used to determine goals and assess success of restoration projects are: 1) vegetation structure, 2) species diversity, and 3) similarity to surrounding vegetation (Ruiz-Jaen and Aide 2005). However, having multiple goals and means of assessing restoration success may make determining the relative effectiveness of different restoration treatments unclear. A single treatment may not be most effective in all aspects of success criteria, making a conclusion on the most effective approach difficult.

Using only local plant material in restoration projects has placed limitations on the park due to the increased costs associated with the collection and propagation of local plant material. Large differences in costs associated with the various active approaches to restoration (Figure 2) combined with the limitation in funding available for restoration projects in the park emphasize the need for evaluation of the cost-effectiveness of the different approaches to restoring disturbances in RMNP. Cost-effectiveness is an analytic tool which can assist decision-makers in choosing an optimum treatment among several different restoration approaches (Pinjuv et al. 2001). This analytic approach compares different courses of action by evaluating the costs and effectiveness in achieving some specified goal (Quade 1967). Costs are based on labor, equipment, fuel, and other direct costs. Effectiveness is estimated by a treatment's ability to meet pre-defined project objectives. This technique is therefore based on the selection of acceptable measures of effectiveness, which is determined by analyst assumptions of appropriate attributes of restoration success.

OBJECTIVES

The objectives of this study were to: 1) evaluate the relative success of different restoration approaches by assessing the effectiveness of treatments in creating plant communities dominated by native perennial species similar in composition and cover to undisturbed conditions, and 2) analyze the cost-effectiveness of treatments by comparing actual treatment costs to a measure of restoration effectiveness based on cover values of the species present on restoration sites.

METHODS

All data collected for this study were gathered from lands within or directly adjacent to Rocky Mountain National Park (RMNP). Data were collected from 13 disturbed and 13 adjacent undisturbed sites within the montane zone ranging in elevation from 2380 m to 2740 m. The 13 disturbed sites sampled in this study were initially disturbed through anthropogenic activities, such as construction/ improvement of park facilities, installation of utilities, and road, trail and parking lot construction/removal. Sites were selected

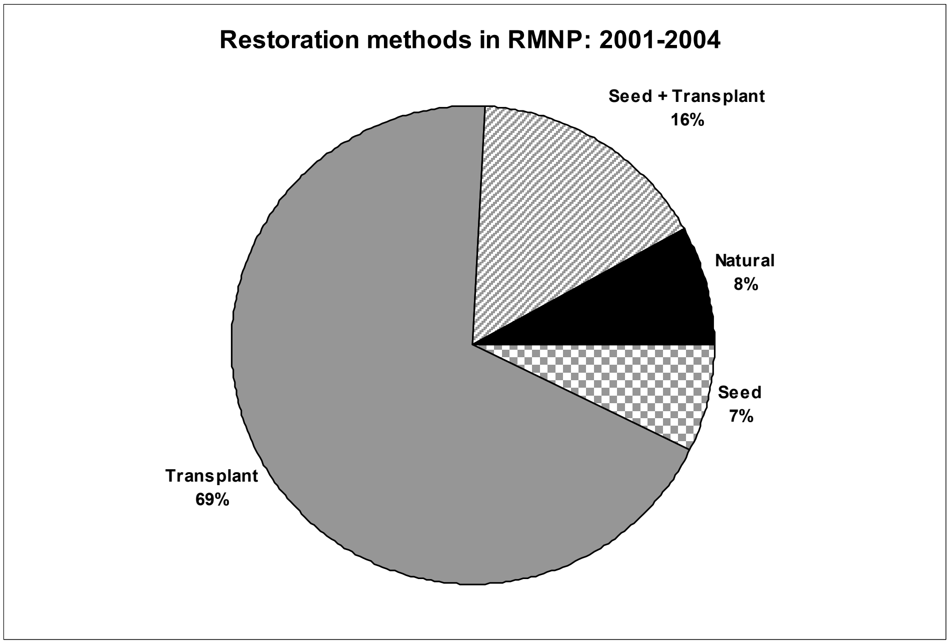

Figure 1. Comparison of methods used in restoration projects, RMNP. 124 restoration projects were evaluated from RMNP records from 1991 to 2004.

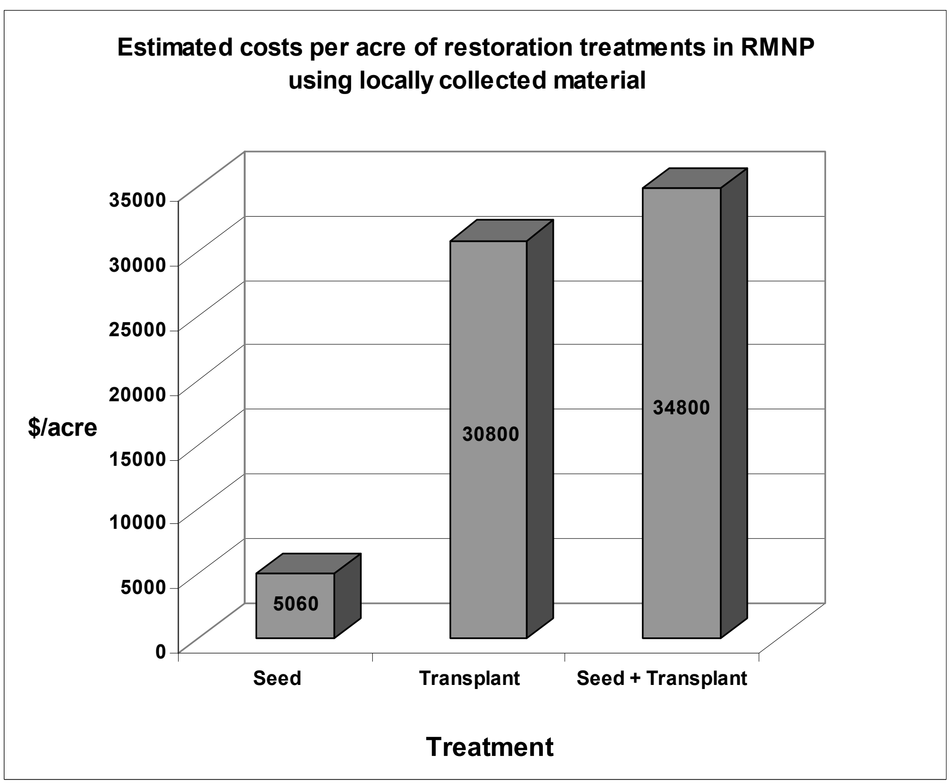

Figure 2. Estimated costs of restoration treatments using locally collected plant material.

Table 1. Description of 13 disturbance study sites, RMNP.

Site	Elevation (m)	Years Since Disturbance	Disturbance Size (m^2)	Restoration Treatment
Fire Management Office	2380	3	790	Seed + Transplants
McGraw Ranch water tank	2440	3	1250	Transplants
MPCG Amphitheater	2490	3	615	Transplants
Beaver Meadows Entrance-N	2500	3	5050	Seed
Beaver Meadows Entrance-S	2500	3	3000	Seed
McClaren Hall	2380	5	1855	Seed + Transplants
Sprague Lake picnic area	2650	5	1370	Transplants
Fisherman's access	2680	5	750	Seed
Jenning's bridge	2680	5	1500	Seed + Transplants
Twin Sisters parking lot	2740	5	200	Transplants
Hot Shot Dorm	2380	8	230	Transplants
William Allen White Cabin	2460	8	100	Seed + Transplants
Wild Basin Kiosk	2560	8	~300	Transplants

Table 2. Hypothetical seed mixture approximating the relative cover values of undisturbed plant communities on reference sites in RMNP. Seed amounts based on seeding rate of 400 seeds/m^2.

Species	% seed mix	Reference % cover	Amount seed needed (grams/ac)	Collection rate (grams/hr)*	Collection time (hr/ac)
Hesperostipa comata	25	15.2	2700	100	27
Bouteloua gracilis	20	7.7	180	10**	18
Koeleria macrantha	20	2.7	775	3	258
Pascopyrum smithii	10	2.5	400	400	1
Pseudoroegneria spicata	10	2.3	456	150	3
Heterotheca villosa	5	3.5	48	20	2.5
Potentilla hippiana	3	2.7	130	40	3
Penstemon unilateralis	3	1.3	8	50	<1
Achillea millifolium	3	1.3	7	40	<1
Gaillardia aristata	1	0.2	2	150	<1

* Winslow and Majerus 2005
** Phan and Smith 2000

based on the time since restoration activities took place, as well as the type of restoration treatment applied (Table 1).

Undisturbed areas adjacent to the disturbance were also sampled for comparison to disturbed areas and are referred to as reference areas. Due to the physical proximity of these reference areas to the disturbed area and the frequency of human activities in these areas of the park, reference areas cannot be considered truly 'undisturbed'. Rather they are areas of similar physical and ecological characteristics to the disturbed site, but have not been subjected to the same intensity or severity of disturbance. Reference sites chosen in this study were dominated by native perennial species typical of undisturbed conditions.

Vegetative cover by species was determined during the 2004 and 2005 growing seasons on disturbed and reference sites. Cover was estimated by use of the line-point transect method, with points read every 0.5 m along a 50 m transect. A minimum of five transects were randomly sampled in disturbed sites within the boundaries of the disturbed area, as well as randomly within an equivalent area for adjacent reference sites. The total number of transects to be sampled was based on the degree of variation in cover values (Bonham, 1989). Floristic similarity of the vegetation was calculated from the percent cover data in order to compare similarity of plant community composition between disturbed and associated undisturbed sites. Sorenson's quantitative index of community similarity estimates community similarity based on cover values of species common to both sampled areas, whereas the qualitative index of community similarity is based on species presence (Mueller-Dombois and Ellenberg 2002).

All data were analyzed using analysis of variance (ANOVA) with the general linear models procedure (PROC GLM) in SAS 9.1. Significant differences were estimated by using the least significant difference method (LSMEANS) at a significance level of p 0.05 (SAS Institute, Inc. 2002). Means for all reference sites within each treatment were averaged in order to reduce site to site variations and typify a more realistic description of undisturbed conditions (Hobbs and Norton, 1996).

Average total costs for seeding were based on a seeding rate of 400 seeds/m^2 of a hypothetical seed mix containing 5 common grasses and forbs at rates approximating the relative cover values on reference sites and estimated from wildland seed collection rates (Table 2) multiplied by the GS-5 pay rate of $12.75/ hr. Based on these estimates, the cost of producing this seed mix by hand collection is approximately $545 per pound. Seeding required on average 80 hours of labor per acre, requiring average costs of approximately $1050 per acre. Total costs of seeding one acre of land were roughly $5060. These costs include seed collection, site preparation, broadcast seeding, and hand raking of the soil surface to incorporate seed and increase seed-soil contact. Estimated transplanting costs were determined from RMNP records from the years 2001 to 2005 and are based on the number of employee hours required for the total number of containerized plants produced multiplied by the GS-5 pay rate of $12.75/ hr. Containerized plant production costs on average $3.80 per plant. This cost includes site preparation and labor involved in planting on site.

Data on application rates of treatments were not available for all sites sampled on this study. Cost-effectiveness analysis was performed on 10 of the 13 sites. Costs of treatments were estimated by determining the size of the disturbance in addition to seeding and/or transplanting rates, then multiplying by the estimated costs associated for treatments. Effectiveness of restoration treatments was estimated by the calculation of an Effectiveness Index (*EI*) based on the absolute cover of native species (N) multiplied by the relative cover of native species (N/N+I, where I is the absolute cover of invasive species):

$$EI = \left[N * \left(\frac{N}{N+I} \right) \right] * 100$$

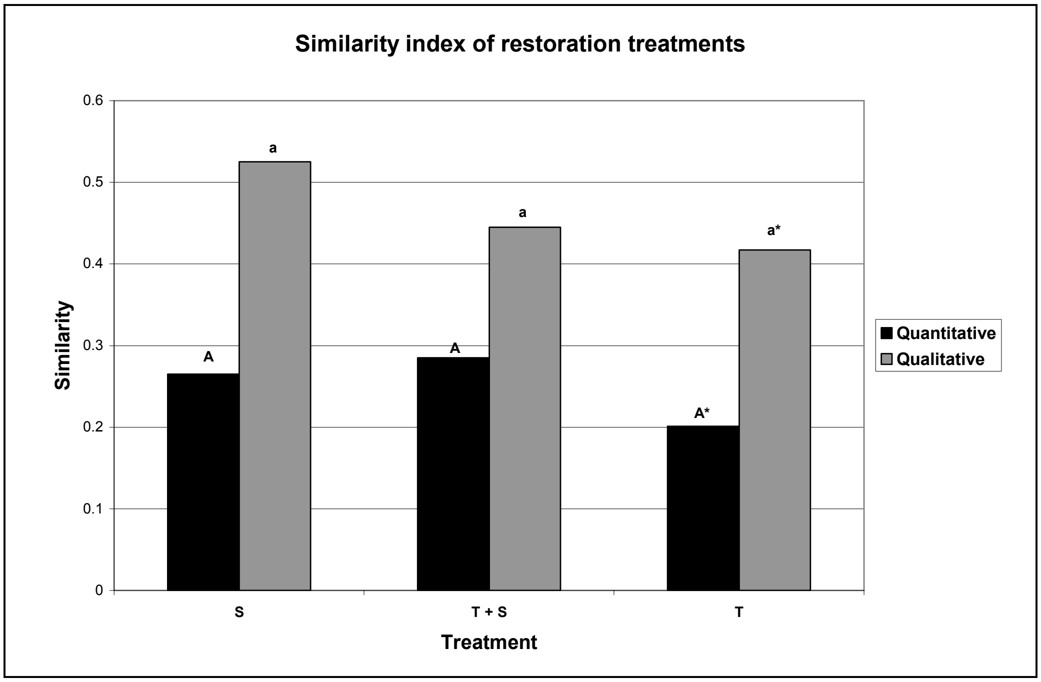

Figure 3. Sorenson's quantitative and qualitative similarity index for disturbed/ reference pairs. Treatments: S - seeding; T+S - transplanting and seeding; T - transplanting. Different letters across treatments represent significance at p 0.05 (* shows significance at p 0.1).

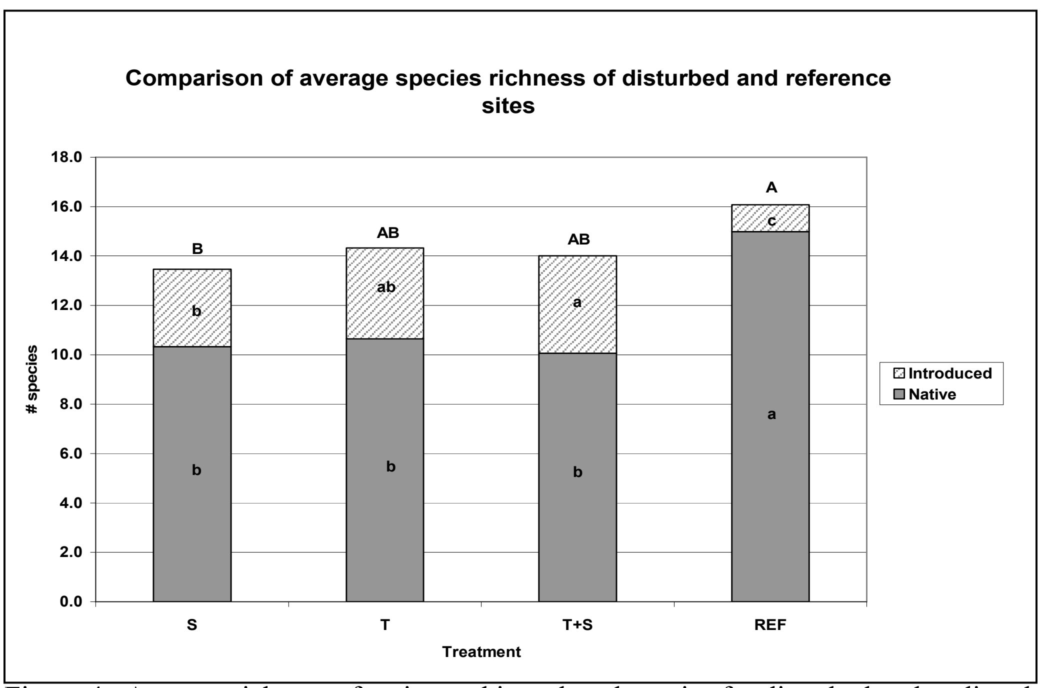

Figure 4. Average richness of native and introduced species for disturbed and undisturbed sites, RMNP. Treatments: S - seeding; T – transplanting; T+S - transplanting and seeding; REF - reference. Different letters across treatments represent significance at p 0.05.

RESULTS AND DISCUSSION

Community Similarity

Community similarity (*SI*) did not differ between treatments based on either species presence or cover values (p 0.05) (Figure 3). Transplanted sites had significantly lower similarity to undisturbed reference sites compared to seeded and seed + transplant sites (p 0.1) (p = 0.095, quantitative *SI*; p = 0.059, qualitative *SI*). Sorenson's quantitative *SI* was highest for seeded sites and lowest for transplanted sites, suggesting that seeding disturbed sites produces plant communities with species cover most similar to undisturbed conditions. Species presence data show sites treated with a combination of seeding and transplanting have the highest similarity, while transplanted sites have the lowest similarity. These data indicate that including seeding as a treatment for restoration of disturbed areas is important for maximizing the similarity of plant community structure between disturbed and undisturbed conditions.

Species Richness

Species richness was not affected by treatment type. Average richness of native species was lower for all treatments compared to reference sites (Figure 4). Treatments did not differ in their average number of native species. The average number of introduced species on disturbed sites was greater than on reference sites. On average, seeded sites had significantly fewer introduced species than sites treated with a combination of transplanting and seeding (p 0.05). However, this difference is an average of less than one species.

Plant cover

Treatments differed in their effect on cover values of plant functional types (Figure 4). Total cover was significantly different between disturbed and reference sites, with the lowest cover values observed on seeded sites. Cover of introduced species was greater for all disturbed sites relative to reference sites, and did not differ among treatments. Native species cover was significantly lower for all disturbed sites compared to reference sites. Seeded sites had significantly less native species cover than transplanted sites or sites seeded and transplanted. All treatments had significantly less native perennial grass cover than reference conditions. However, sites treated with seed and transplanted material had the highest cover of perennial grasses of the three treatments. No significant difference was observed between transplanted sites and reference sites for both shrub and native perennial forb cover. Sites treated with seeding and seeding in addition to transplanting had significantly less shrub and native perennial grass cover compared to reference areas. There were no significant difference in native annual forb or grass cover among the three treatments and reference sites. These results suggest including transplanting containerized material in order to produce cover values of native perennial species most similar to reference conditions.

Assessment of the effectiveness of restoration treatments based on multiple plant community attributes for disturbed and undisturbed sites makes the determination of optimum approaches unclear. Community similarity indices suggest the importance of including seeding for the creation of plant communities similar in cover values and species presence. Analysis of vegetation cover shows transplanting plant material results in the highest total cover as well as cover of native perennial species on disturbed sites. Treatments had no effect on the richness of native species relative to undisturbed conditions.

Cost-effectiveness Analysis

Limitations in financial resources for restoration projects emphasize the need for an analysis of the ratio of cost to effectiveness for different restoration approaches. Comparison of costs to relative effectiveness for alternate treatments can provide important information for managers during the restoration planning process. The calculation of an Effectiveness Index (*EI*) allows for a comparison of costs of treatments to an estimation of the efficacy of treatments in restoring native perennial plant communities. Sites restored with seeding, transplanting, or a combination of the two have differing application rates of these treatments. Sites are transplanted at different densities or seeded at variable rates. Criteria used for estimation of restoration effectiveness, such as vegetative cover, are most likely dependant on application rates. Costs are dependant on the size of the disturbed area as well as the application rates of the treatments. Transplanting or seeding at higher densities will increase costs while potentially resulting in higher total and native species cover. Therefore, treatments are analyzed on a site specific basis with costs estimations normalized to dollars per acre for all sites (Table 3). Thus, this approach allows for a comparison of sites with different sizes of disturbance and application rates of treatments

Cost-effectiveness analysis has advantages of over other economic decision-making approaches. Other analytic approaches to decision-making can mask the assumptions of criteria for effectiveness. For example, cost-benefit analysis can conceal assumptions of effectiveness measures by assigning dollar values to non-market benefits, such as native species cover or species richness (Quade 1967). Calculation of *EI* is an attempt to quantify effectiveness by the cover of vegetation on each site. *EI* not only factors in native species cover, but also total and invasive species cover in order to weight competition from undesirable species. The greater the relative cover of native species, the less competition from introduced species there will be. Thus, *EI* will be low for plant communities with low total cover or on sites where the relative cover of native species is low. *EI* will be high for sites with high total cover in addition to high relative cover of native species. A lower ratio of cost to *EI* is desirable, indicating a lower cost per unit effectiveness.

Cost-effectiveness analysis of restoration treatments in RMNP shows large relative differences between treatments (Figure 5). Sites treated with seeding alone had a lower ratio of cost to effectiveness compared to sites treated with transplanting or a combination of transplanting and seeding. Although seeded sites had lower total cover and relative cover of native species compared to the other treatments, the costs of seeding are dramatically lower resulting in a the least cost per unit effectiveness of all treatments. Both treatments that included transplanting activities produced higher total cover and greater relative cover of native species, however the increased labor costs involved in the production and planting of containerized plant material outweigh the increased effectiveness of these treatments.

Numerous restoration goals may require a variety of criteria for determining the effectiveness of treatments. This can lead to uncertainty in decisions of optimal restoration approaches. Managers are regularly faced with limitations in the availability of financial resources for restoration projects. Given this constraint, preferred restoration approaches are often determined by the combination of costs and the ability to meet project objectives for a given treatment. Cost-effectiveness is an analytic tool which can assist decision-makers with limited financial resources in choosing among several different restoration approaches. Cost-effectiveness analysis of treatments using different sources of plant material show seeding of hand collected seed to be the most efficient use of limited financial resources in the restoration of disturbances in RMNP.

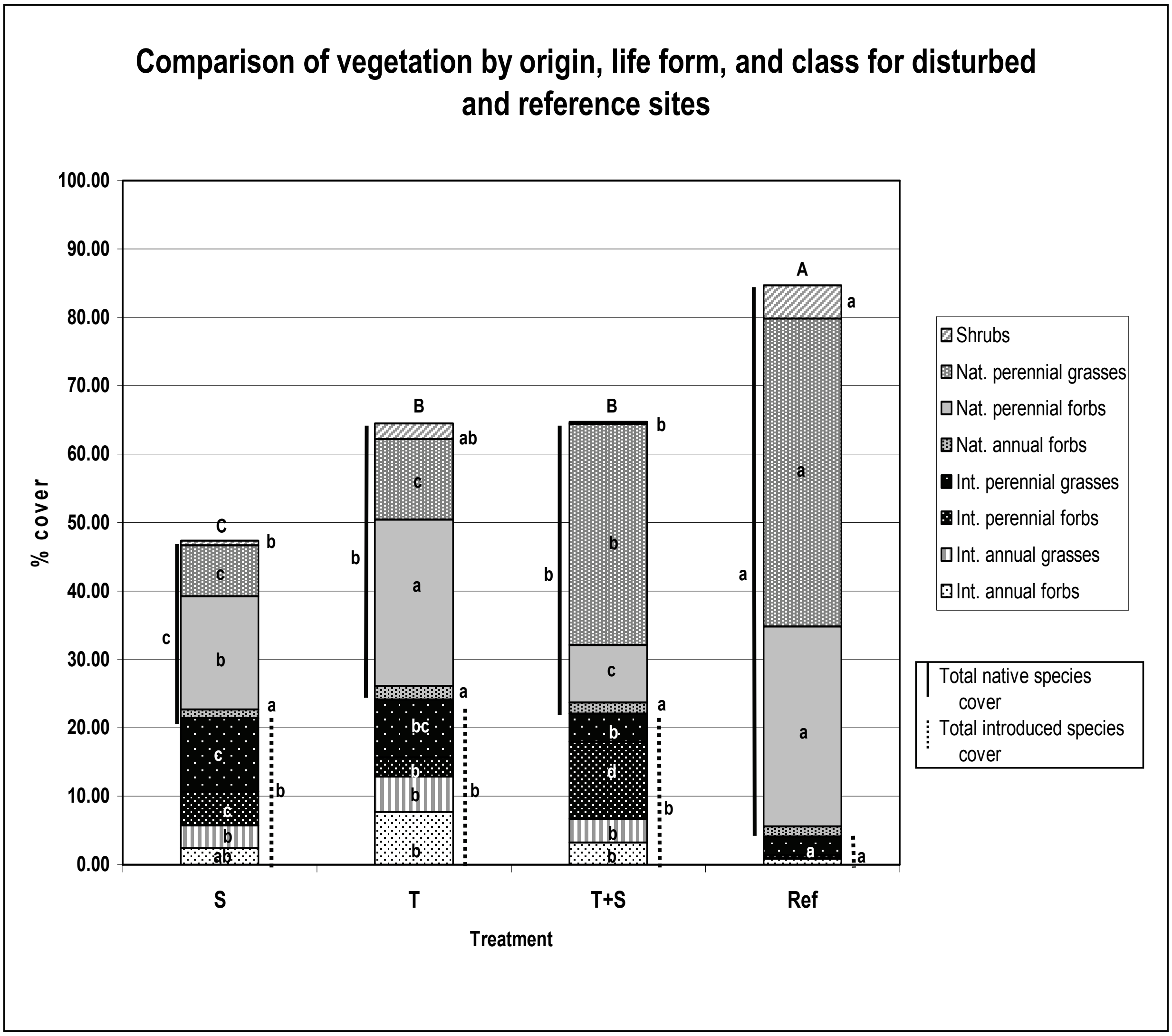

Figure 4. Comparison of cover values for disturbed and reference sites based on origin, life form, and class of species. Treatments: S - seeding; T – transplanting; T+S - transplanting and seeding; Ref – reference. Different letters across treatments represent significance at p 0.05 within each plant functional type (i.e. introduced annual forbs).

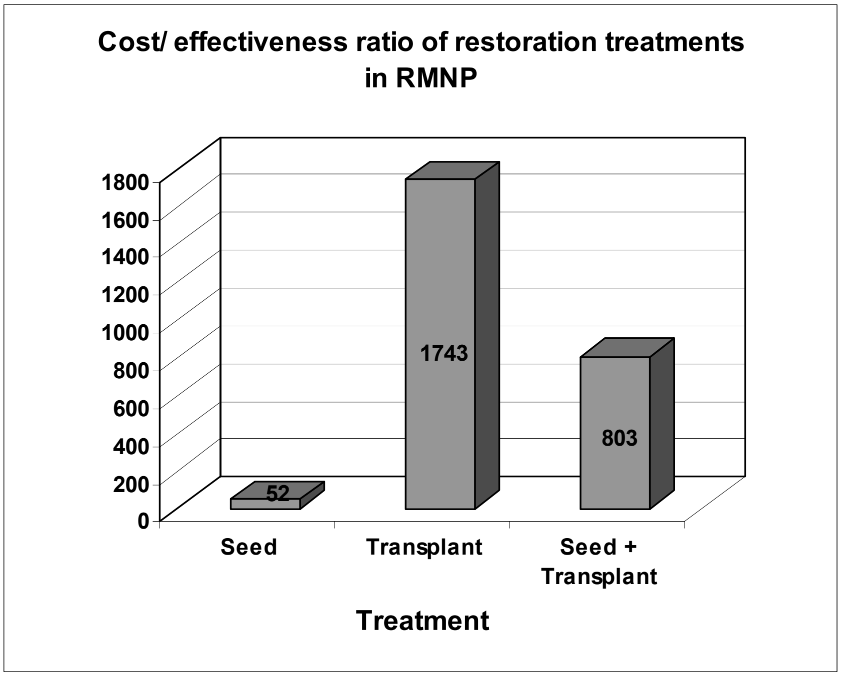

Figure 5. Cost-effectiveness ratio of restoration treatments.

Table 3. Example of Cost-effectiveness analysis calculation.

I. Cost

Site	Size (m^2)	Treatment	Application rate	Total cost (per acre)
Beaver Meadows Entrance – south	3000	seeding	1.43 lb (1.93 lb/acre)	$2100
MPGC amphitheater	610	transplanting	2870 plants (19,000 plants/acre)	$72,200

II. Effectiveness

Site	cover native sp.	cover invasive sp.	relative cover native sp.	*EI*
BME – south	0.31	0.04	0.91	27.2
MPGC amphitheater	0.39	0.28	0.58	22.5

III. Cost-effectiveness ratio

Site	Ratio*
BME – south	77
MPGC amphitheater	3209

* total cost per acre/*EI*

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EVALUATION OF REVEGETATION SUCCESS ON NATURAL GAS WELL PAD SITES NEAR PARACHUTE, COLORADO

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ABSTRACT

The western United States is home to large expanses of public lands with a variety of diverse ecosystems and an abundance of natural resources. Natural gas exploration and production in the West causes a variety of disturbances (i.e. construction of well pads, roads, pipeline right-of-ways) that have the potential to threaten native plant communities, wildlife habitat, and recreational opportunities. Sound revegetation techniques are needed to offset the impacts caused by natural gas development. This project aims to evaluate the effectiveness of revegetation techniques used on natural gas well pad sites developed by Williams Production RMT Company near Parachute, CO. A field study was conducted in 2005 to quantify the success of various revegetation techniques on a sub sample of previously reclaimed well pad sites within six distinct plant communities. Intensive characterization of the current vegetation and soils was conducted at each site to assess the effectiveness of specific seed mixtures, mechanical soil treatments, and soil amendments used in the revegetation process. Creation of a database containing present and historical revegetation data was also a significant component of our work. Results of field data analyses will facilitate the development of reclamation test plots in 2006 to answer questions that will help refine reclamation approaches currently being employed.

MANGANESE AND ZINC TOXICITY THRESHOLDS FOR MOUNTAIN AND GEYER WILLOW

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ABSTRACT

Information on heavy metal toxicity thresholds of willows is lacking but critical for the successful restoration of contaminated riparian areas. In a greenhouse study, two willow species, *Salix geyeriana* and *S. monticola,* were screened for toxicity thresholds to manganese (Mn) and zinc (Zn). Manganese and Zn treatments were applied in solution form from 50 to 1000 mg l^{-1} and 100 to 1000 mg l^{-1}, respectively. The lethal concentration (LC50) values were 3117 and 2791mg Mn l^{-1} and 556 and 623 mg Zn l^{-1} for Geyer and mountain willow, respectively. The effective concentration (EC50-shoot) values were 2263 and 1027 mg Mn l^{-1} and 436 and 356 mg Zn l^{-1} for Geyer and mountain willow, respectively. Shoot tissue values did not increase with increasing treatment concentrations, yet metals in the roots did increase consistently in response to the treatments. Metal levels in the shoot tissues were surprisingly low for Zn (65-139 mg kg^{-1}) and moderate for Mn (1300-2700 mg kg^{-1}). These results suggest *S. geyeriana* and *S. monticola* may be useful in revegetation of sites with high Mn, however, caution is necessary when planting these species in soils high in Zn. This will support the work of restoration ecologists and risk assessors and will help determine specific causes for poor growth of willows planted in contaminated substrates.

INTRODUCTION

The extraction and processing of ores has created a legacy of lands polluted with inorganic toxins. Throughout the mountainous regions of the western United States, mining practices have damaged natural resources by depositing mine tailings adjacent to, or in drainages. This has allowed the tailings to be transported downstream and deposited through fluvial processes along riparian zones. Due to the acidic composition of the tailings and the elevated levels of heavy metals, the tailings deposits are often devoid of vegetation and may threaten human, wildlife, and livestock health. In order to prevent pollutants from traveling further through trophic levels or downstream into municipal water supplies, remediation action must be taken. Phytostabilization, the use of soil amendments and plants to immobilize pollutants in the system, is a long-term and cost effective remediation method being investigated and applied in recent years (Berti and Cunningham 2000).

In the process of phytostabilization, the toxic growth medium must be amended in order to neutralize soil acidity, improve soil structure, and provide organic matter to reduce the available pools of metals. Following this, plant species must be selected that will establish and thrive in local conditions, tolerate the potentially available pollutants, and form large root systems to aid in soil stabilization. Willows (*Salix* spp), shrubs that frequently dominate riparian habitats across North America, are being explored as a genus that fulfills the aforementioned criteria (Kuzovkina and Quigley 2005). Willows have been used extensively in river bank stabilization projects because of their ability to establish from pole cuttings and spread rapidly through volunteer resprouting. The cost effectiveness and likelihood of long-term revegetation success when willows are used in riparian restoration has led researchers to examine the potential of various willow species to tolerate, adapt to, or hyperaccumulate heavy metals (Kuzovkina et al., 2004, Landberg and Gregor 2002, Pulford et al. 2002, Pulford and Watson 2003, Punshon and Dickenson 1997, Vyslouzilova et al. 2003, Watson et al. 2003b). However, there is a distinct void of studies on critical toxicity thresholds for willows.

In order to improve the success of phytostabilization projects, it is necessary to know the toxicity thresholds of plants to the pollutants. Until recently, research on critical toxicity thresholds has been focused on agronomic species which may have significantly different tolerance levels compared to native species and, in particular, to the phreatophytic willow (Ross and Kaye 1994). Information on willow tolerance to heavy metals is limited and research has been predominantly focused on European species (Dinelli and Lombini 1996, Pulford et al. 2002, Punshon and Dickenson 1997, Landberg and Gregor 1994, Verveake et al. 2003, Vyslouzilova et al. 2003). However, there is a common conclusion from these studies. These willow experiments show that tolerance and uptake of heavy metals is strongly species and clone dependent (Pulford et al. 2002, Gregor and Landberg 1999). If willows are to be used in restoration projects in North America, information on species specific toxicity thresholds is needed.

Manganese (Mn) and zinc (Zn) are two common inorganic pollutants found in mine tailings and have been the focus of recent toxicity screening studies on native species used for restoration (Paschke et al. 2000, 2005, Reichman 2001, 2004). These elements are similar in that both are essential to plants in micro amounts and, therefore, are readily taken up and translocated within plants. Consequently, when present at elevated levels in the soil, they can become toxic to plants and interrupt essential metabolic and reproductive processes. Manganese and Zn toxicity can both cause specific deficiencies of other essential elements. Interference with the uptake, translocation, and utilization of iron (Fe) is the primary concern (Chaney, 1993; Kabata-Pendias and Pendias 2001).

In soils, Mn and Zn react similarly to changes in pH as both increase 100 fold in bioavailability with each unit decrease in pH (Kabata-Pendias and Pendias 2001). The biochemical pathway of Mn is unique because bioavailability corresponds closely with redox potential and increases in saturated soil conditions (Bartlett 1988). In addition, Mn-oxides can exacerbate toxic effects of other heavy metals through association, dissolution, and consequent disproportionation of the other elements when there are intermittent anaerobic conditions (Davranche and Bollinger 2000, McKenzie 1989).

Toxicity thresholds are indicators used by ecologists and agronomists to understand plant tolerances to pollutants such as Mn and Zn. The most commonly studied threshold is the shoot phytotoxicity threshold (PT10), also known as the critical concentration, which refers to the tissue concentration of the toxin in the shoot corresponding to a 10% reduction in yield of the shoot. Alloway (1990) states that tissue concentrations exceeding 300 and 150 mg kg^{-1} for Mn and Zn, respectively, indicate phytotoxicity. A review of toxicity studies on agronomic plants by Macnicol and Beckett (1985) demonstrated that PT10 levels for Mn span a far greater range than the critical ranges found for Zn. Manganese PT10 values range from 100 – 7000 mg kg^{-1} (median = 500). The range for Zn spans from 70-900 mg kg^{-1} (median = 260).

The impetus for this study stems from the historic mining activity in the headwaters region of the Arkansas River near Leadville, Colorado. These land use practices have degraded natural resources along an 18 km stretch of the river corridor (Archuleta et al. 2003). It has been determined that phytostabilization is the preferred restoration method for the majority of the 170 tailing deposits that are located in the floodplain (Archuleta et al. 2003). Bourret (2004) and Fisher (1999) have planted willows in previous in-situ phytostabilization projects and found that the shrubs still exhibited chlorosis and stunted growth, suggesting their inability to thrive under the given conditions.

This study was initiated to determine toxicity thresholds of two native willow species, *S. geyeriana* and *S. monticola*, (Geyer and mountain willow, respectively) to Mn and Zn. This information will be useful in identifying causes of the poor growth and chlorosis observed in willows growing in the field.

MATERIALS AND METHODS

Species Selection and Collection

The riparian habitat along the Upper Arkansas River, located at 2,800 m above sea level, is dominated by willow species with an understory of grasses and grass-likes (USDA-NRCS, 1997). Mountain and Geyer willow were chosen for this experiment as they are the two dominant species in this flood plain and are widely distributed across the central Rockies and western United States, respectively (Dorn 1997, USDA-NRCS 2006). Cuttings of dormant willows with diameters of 1-3 cm were collected in October 2004 following senescence and leaf drop. The cuttings originated from 10 distinct shrubs located at various distances from the river and tailing deposits. The stakes were stored for 2 months in burlap sacks in a refrigerator that maintained a temperature range of 1-3˚C. The sacks were misted on a weekly basis and covered with a plastic tarp to prevent desiccation.

Plant Growth Conditions

A sand culture technique was used to determine the toxicity thresholds (Paschke and Redente, 2002 Paschke et al. 2000, 2005). The protocol from the aforementioned studies was followed closely. The only significant alterations were in response to the increased water budget of the willow cuttings as compared to the seedlings of the upland species studied by Paschke et al. (2000, 2005). Willow stakes were cut into 20 cm long sections and planted in 5 by 18 cm cone-tainers (Stuewe & Sons, Corvallis, OR). The growth medium was washed quartz sand with a pH of 6.5-7. A 2.5 cm piece of glass wool was placed in the bottom of each cone-tainer to prevent sand from falling out the drainage holes. The 10 distinct clones were distributed evenly across the trays. The daily ambient temperature in the greenhouse was 27ºC (53% RH) and the nighttime average was 21ºC (75% RH). The photoperiod in this greenhouse was extended to 16 hours with high intensity discharge lights (430 W).

The cuttings were placed under a mister on a 5 minute cycle for the first week of root establishment and then watered by hand daily for 4 weeks. Prior to initiation of treatments, any stakes that had not sprouted were removed and the cone-tainers placed in every fifth hole of the 50 slot Support Trays (Stuewe & Sons, Corvallis, OR) in order to reduce competition for light and space. Number of replicates for each treatment ranged from 32 to 40 willows due to the range of establishment success for each species. At this point, leader measurements and numbers of leaders were recorded to determine if pretreatment growth patterns were equal across the trays.

The treatments included a control plus 6 levels of Mn and separately to a control and 6 levels of Zn in a completely randomized design. Manganese was supplied in solution form using $MnSO_4$ at the following concentrations: 0, 100, 500, 1000, 3000, 6000, and 10,000 mg Mn l^{-1}. Zinc was given in solution using $ZnSO_4$ at the following concentrations: 0, 50, 100, 200, 400, 600, and

1000 mg Zn l^{-1}. The application of metals occurred on Monday, Wednesday, and Friday afternoons and a nutrient solution (a modified Hoagland's solution) was supplied on Tuesday and Thursday afternoons. Since willows are phreatophytic species, they required water each morning and twice on the weekends when no other solution was applied. To ensure saturation of the sand, the solutions were poured into each cone-tainer until there was drainage out the bottom. During the course of treatment, observations were recorded and photographs taken on the visual symptoms of toxicity. This treatment procedure began on January 26, 2005 and was followed for 50 days.

Toxicity Measures and Statistical Analysis

In this study, we determined 3 distinct types of toxicity measurements (Ross and Kaye 1994).

> LC50 (lethal concentration): Treatment level at which 50% of the plants have died at the conclusion of the experiment
> EC50 (effective concentration): External concentration of a pollutant that causes a 50% decrease in plant biomass
> PT50 (phytotoxicity threshold): Plant tissue concentration of the pollutant corresponding to a 50% biomass reduction

The LC50 was determined by recording plants as dead or alive after 50 days. A plant was determined as dead when there were no remaining live leaves. At the end of the treatment period, the shoots and roots of all surviving plants were separated from the original woody stake. Roots were washed free of sand with a gentle stream of water and rinsed over a 0.5 mm sieve. Some roots from each plant had to be tweezed from the glass wool plug. The above and belowground plant parts were dried for 72 hours at 50°C. The dry weight of shoots and roots from each plant was recorded for calculating four separate EC thresholds (EC50-shoot, EC10-shoot, EC50-root, and EC50-plant). The EC50-plant was found by adding shoot and root dry weights and did not include the woody stake from the original cutting because it does not grow. In a growth period this short, new woody material does not have time to develop.

The PT50 was determined separately for the shoots and roots because below ground plant parts frequently accumulate much higher tissue concentrations of heavy metals than the stems and leaves. A subset of 7 plants was randomly selected from each treatment group for analysis of elemental tissue concentrations. In addition, a sample of leaves from 5 randomly chosen individuals from each treatment group was combined for a leaf only analysis. Plant tissues were ground through a 2 mm screen and 0.5 g of material was then digested with 4 ml HNO_3 and 1 ml $HClO_4$ at 200˚C for 2 hours. Tissue analysis was conducted using inductively coupled plasma-atomic emission spectroscopy (ICP-AES).

Statistical analyses were completed using SAS version 9.1 (SAS Institute, 2002). A polynomial trend was fit to the data for each toxicity measurement. The models were chosen based on r^2 and p-values and then used to calculate the specific toxicity thresholds. A 0.05 level of significance was used for each individual analysis. The primary objective of the study was to investigate dose response of different species to Mn and Zn, and therefore, interspecific comparisons were not examined.

RESULTS

Visual symptoms of toxicity

Biomass of all 4 species-treatment combinations gradually declined in response to increasing metal concentrations. By the end of the 50 days of treatment, this was apparent through visual observation and confirmed for above and below ground plant parts through measurement of root and shoot dry weights (Figures 1, 2).

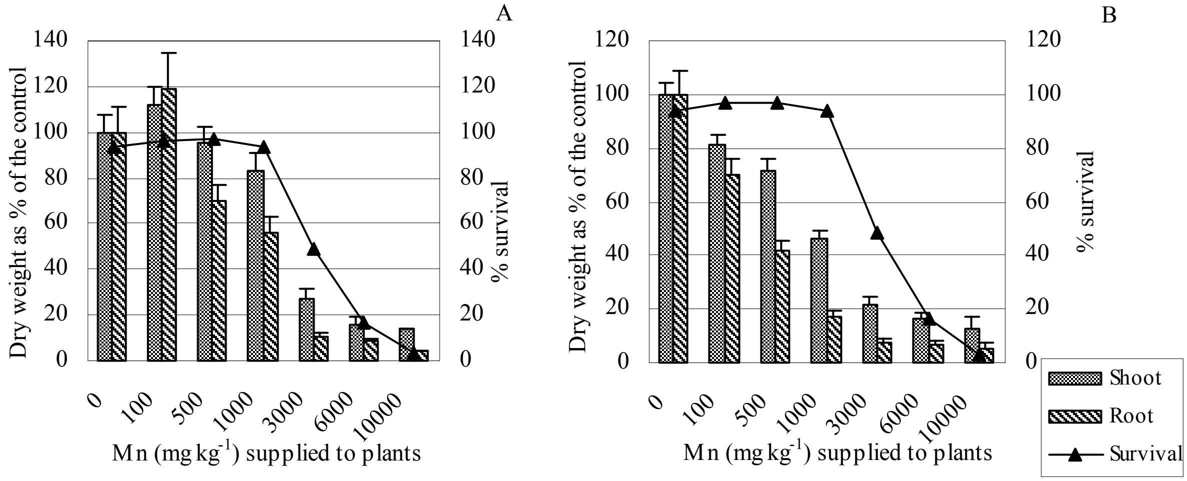

Figure 1. Mean dry weight and percent survival (+/- SE) of Geyer (A) and mountain (B) willow in response to increasing manganese concentrations.

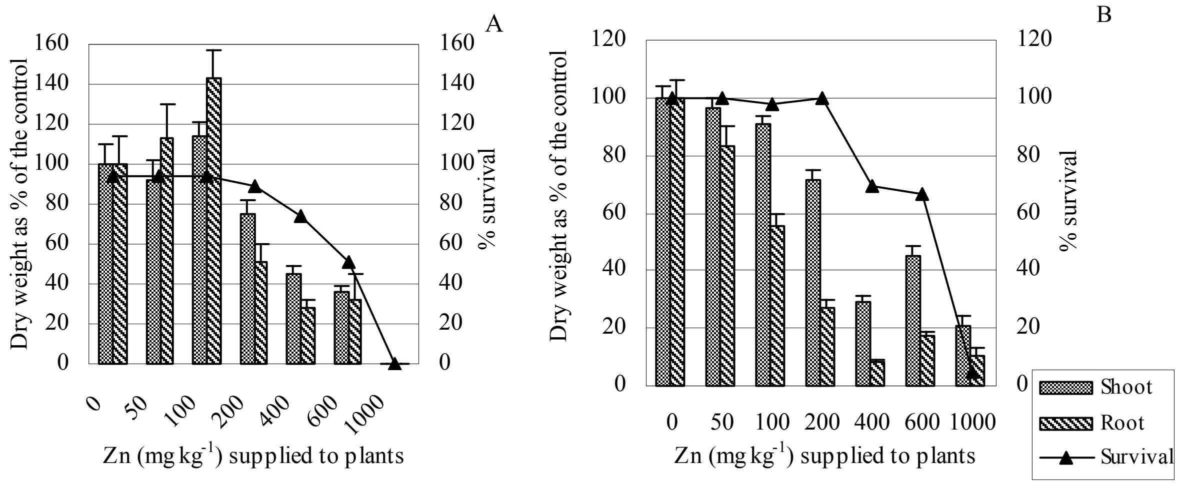

Figure 2. Mean dry weight and percent survival (+/- SE) of Geyer (A) and mountain (B) willow in response to increasing zinc concentrations.

The foliar symptoms of Mn toxicity frequently include necrotic spotting, interveinal chlorosis (usually on older leaves), and crinkling of younger leaves (Marschner 1995). In this study, the mountain willows that received Mn applications displayed the common symptoms, however, Geyer willows developed pale yellow leaves (including the veins) and occasionally some minute black dots became apparent on older leaves.

The common foliar symptom connected to Zn toxicity is chlorosis in young leaves (Chaney 1993). In this study, both species exhibited a strong pattern of interveinal chlorosis in young and older leaves. Often there was a striking contrast between the pale leaf surface and the dark green veinal pattern. Some leaves from Geyer willows slowly yellowed from the margins inward. Additionally, mountain willows that received higher Zn concentrations did not drop their leaves once they were dead; rather, the dead leaves curled and tended to remain on the plant.

Toxicity Thresholds

Mortality occurred beginning with the middle treatment levels for both Mn and the Zn for both species (Figures 1 and 2). The toxicity thresholds, corresponding regression models, R^2, and p-values are presented in Table 1. The LC, EC, and PT-root thresholds were obtainable for all treatment-species combinations. The data for shoot phytotoxicity concentrations did not fit regression models. Therefore, specific PT thresholds are not presented, however, a trend of limited or restricted translocation was observed and described.

Table 1**.** Calculated toxicity thresholds, and the corresponding R^2, p-values and models for the response of Geyer and mountain willow to manganese and zinc treatments.

Threshold type	R^2	*p*	**Model**	**Calculated threshold** [a]
Geyer willow – Manganese				
LC50	0.98	0.0005	$y = 101.8699 - 0.019663*dose + 9.695E{-}7*dose^2$	3117
EC50 shoot	0.32	<.0001	$y = 108.6738 - 0.031109*dose + 2.289E{-}6*dose^2$	2263
EC10 shoot	0.32	<.0001	$y = 108.6738 - 0.031109*dose + 2.289E{-}6*dose^2$	630
EC50 root	0.29	<.0001	$y = 108.1229 - 0.063959*dose + 0.000012*dose^2 - 6.74E{-}10*dose^3$	1135
EC50 plant	0.33	<.0001	$y = 107.5943 - 0.03276*dose + 2.508E{-}6*dose^2$	2094
PT50 shoot	NA	NA	Unable to fit a model	NA
PT50 root	0.18	0.0298	$y = 96.63547 + 0.01517*tc - 3.383E{-}6*tc^2$	6581
Mountain willow - Manganese				
LC50	0.98	0.0003	$y = 106.3591 - 0.024325*dose + 1.48E{-}6*dose^2$.	2791
EC50 shoot	0.57	<.0001	$y = 92.98248 - 0.051513*dose + 9.984E{-}6*dose^2 - 5.65E{-}10*dose^3$	1027
EC10 shoot	0.57	<.0001	$y = 92.98248 - 0.051513*dose + 9.984E{-}6*dose^2 - 5.65E{-}10*dose^3$	59
EC50 root	0.42	<.0001	$y = 84.21103 - 0.077056*dose + 0.000018*dose^2 - 1.065E{-}9*dose^3$	501
EC50 plant	0.58	<.0001	$y = 91.48851 - 0.055864*dose + 0.000011*dose^2 - 6.5E{-}10*dose^3$	890
PT50 shoot	NA	NA	Unable to fit a model	NA
PT50 root	0.57	<.0001	$y = 96.70659 - 0.017455*tc + 8.566E{-}7*tc^2$.	3169
Geyer willow- Zinc				
LC50	0.96	<.0001	$y = 102.6342 - 0.094629*dose$	556
EC50 shoot	0.23	<.0001	$y = 105.8803 - 0.128031*dose$	436
EC10 shoot	0.23	<.0001	$y = 105.8803 - 0.128031*dose$	125
EC50 root	0.17	<.0001	$y = 117.735 - 0.179086*dose$	379
EC50 plant	0.23	<.0001	$y = 107.332 - 0.134283*dose$	427
PT50 shoot	NA	NA	Unable to fit a model	NA
PT50 root	0.59	0.0735	$y = 142.8767 - 0.172519*tc$	539
Mountain willow – Zinc				
LC50	0.93	0.0004	$y = 108.3207 - 0.093555*dose$	623
EC50 shoot	0.56	<.0001	$y = 104.2981 - 0.201863*dose + 0.000139*dose^2$	356
EC10 shoot	0.56	<.0001	$y = 104.2981 - 0.201863*dose + 0.000139*dose^2$	75
EC50 root	0.59	<.0001	$y = 103.128 - 0.557909*dose + 0.001024*dose^2 - 5.58E{-}7*dose^3$	120
EC50 plant	0.52	<.0001	$y = 94.67278 - 0.114038*dose$	392
PT50 shoot	NA	NA	Unable to fit a model	NA
PT50 root	0.83	0.0122	$y = 106.5336 - 0.143509*tc$	394

[a] The LC and EC thresholds are in units of mg Mn or Zn l^{-1} and the PT thresholds are reported in units of mg Mn or Zn kg^{-1}.

Manganese

The LC50 value for both species was near 3000 mg Mn l^{-1} (Table 1). An obvious change occurred for both species between 1000 and 3000 mg l^{-1} during which survival declined from 96 to 41% (Figure 1). The EC50 values for both species demonstrated that growth retardation in roots occurred at lower solution levels than in the shoots (Table 1, Figure 1).

A spike in Mn levels was observed for both shoots and roots at the first treatment level (Table 2). For example, Mn concentrations in Geyer willow increased by 711 and 2600 mg kg^{-1} in the shoots and roots, respectively, from the control plants to the plants that received 100 mg Mn l^{-1}.

Levels of Mn found in the root tissue continued to increase with increasing treatment concentrations (Table 2). In contrast, shoot (stems and leaves) tissue concentrations reached approximately 2000 mg kg^{-1} when treated with 500 mg Mn l^{-1} and then leveled off or declined at the higher treatments. Therefore, though the PT-shoot data did not fit a regression model, it appears that the willows ceased translocation of Mn to the shoot once a particular amount of the metal had accumulated. In this study, the data showed this threshold of restricted translocation to span the range of 1300 to 2700 mg Mn kg^{-1}. The tissue analysis of leaves (from a combination of individuals from each treatment group) revealed consistently higher values in leaves over shoots (Table 2). For the leaves, the upper limit of accumulation for Mn was approximately 3200 mg kg^{-1}.

Table 2. Tissue analysis of shoots and roots for willows treated with manganese or zinc.

Mn in shoots						**Mn in roots**		
Mn treatment mg l^{-1}	Geyer mg kg^{-1}		Mountain mg kg^{-1}			Mn treatment mg l^{-1}	Geyer mg kg^{-1}	Mountain mg kg^{-1}
	WS [a,b]	L [a,c]	WS	L				
0	179 (7)	220	99 (7)	161		0	191 (7)	108 (7)
100	890 (7)	1235	1208 (7)	1706		100	2791 (6)	1139 (7)
500	2093 (7)	2204	2160 (7)	3225		500	4384 (7)	3532 (7)
1000	2287 (7)	3273	1530 (7)	2459		1000	5522 (7)	7672 (6)
3000	1331 (7)	2213	1251 (7)			3000		12373 (3)
6000	2701 (6)		2643 (7)			6000		
Zn in shoots						**Zn in roots**		
Zn treatment mg l^{-1}	Geyer mg kg^{-1}		Mountain mg kg^{-1}			Zn treatment mg l^{-1}	Geyer mg kg^{-1}	Mountain mg kg^{-1}
	WS	L	WS	L				
0	81 (7)	95	34 (7)	46		0	60 (7)	21 (7)
50	88 (7)	92	78 (7)	102		50	140 (7)	147 (7)
100	96 (7)	111	97 (7)	116		100	169 (7)	190 (7)
200	139 (7)	106	82 (7)	119		200	368 (3)	394 (6)
400	102 (7)	94	75 (7)	90		400	619 (4)	470 (3)
600	65 (7)		76 (7)	93		600	553 (2)	488 (3)

[a] WS=whole shoot L=leaf only.
[b] Numbers in parenthesis represent sample size.
[c] Leaf values are derived from a single analysis of a composite sample of leaves from 5 plants.

Zinc

Response to the increasing Zn treatments was evident for both species through decreased growth and mortality (Figure 2). The LC50 value for both species was near 600 mg Mn l^{-1} (Table 1). The shoot biomass of both species was reduced by 50% (EC50) at Zn levels near 400 mg l^{-1}. This threshold was higher for both species than the corresponding EC50-root thresholds. However, the difference was only slight for Geyer in contrast to the noticeably lower EC50-root for mountain willow at 120 mg Zn l^{-1}.

Tissue analysis of the plants that received higher doses of Zn showed a continuous increase of Zn in the roots (Table 2). PT50-root values were 394 and 559 mg kg^{-1} for mountain and Geyer willow, respectively. However, analysis of the shoots revealed minimal change in Zn concentrations. In fact, the maximum Zn that accumulated in the shoots of either species was still within the Zn range considered as normal. The shoot values for Zn ranged between 65 and 139 mg kg^{-1}. In the leaf only analysis, slightly higher values were observed for the lower treatment levels when compared to the whole shoot concentrations (Table 2). The values were still low with the highest values only reaching 119 mg Zn kg^{-1}.

DISCUSSION

When comparing the results from our study to other threshold data, attention to the following details was necessary. First, growth conditions, duration of study, and physiology of species can all greatly influence results. Tables 3 and 4 present data from other studies and only experiments with similar protocols have been included. Even so, it is important to note that plants growing in solution are exposed continuously to the metal while plants growing in sand culture were exposed every other day to the metal, and watered several times in between. Second, the thresholds associated with a 10% reduction in yield should be interpreted separately from those provided for a 50% decline. The 10% growth reduction represents a plant that may be chlorotic, but still be vigorous enough to survive and reproduce. A plant that is 50% of its expected size is much more likely to die under long term exposure and is unlikely to successfully reproduce.

Third, it is important that in this study the treatments were applied in solution form and it was assumed that total and available metal levels were equal. To the contrary, in soils, total Mn and Zn are often 100 fold greater than plant available Mn or Zn (Kabata-Pendias and Pendias 2001). Consequently, because LC and EC values refer to the concentration of the soil solution, it is necessary to correlate these values with best estimates of available Mn or Zn.

Manganese thresholds

There are few LC50 values by which to compare our results. Table 3 presents the low and high end for EC50-shoot thresholds for restoration grasses. Despite differences in physiology between grasses and willows, the EC50 results for mountain and Geyer willow are within EC50-shoot thresholds reported for the grasses. In contrast, the EC10-shoot values for mountain and Geyer willow are slightly higher and 10-fold greater, respectively, than those presented for wheat (Table 3).

As with EC thresholds for Mn, there is a wide range of PT values reported for agronomic plants exhibiting Mn toxicity (Table 4). In this study we were unable to fit the shoot values to a model. However, the range of Mn tissue concentrations (whole shoot: ~1200-2700; leaf: ~1200-3300) corresponded with 5-80% biomass reduction. Therefore, we can extrapolate that the PT10 and

PT50 shoot thresholds would fall within these values. Geyer and mountain willow accumulated moderate levels of Mn in their tissue in comparison with

Table 3. Comparisons of effective concentration (EC) toxicity thresholds reported in other studies for manganese and zinc. [a]

	Manganese					Zinc		
Species	Wheat *(Triticum aestivum)*	Tufted Hairgrass (*Deschampsia caespitosa*)	Slender wheatgrass (*Elymus trachycaulus)*	*Phytolacca acinosa* Roxb		Great Basin Wild Rye (*Leymus cinereus*)	Redtop (*Agrostis alba*)	Big bluegrass *(Poa ampla)*
Type	Agronomic	Native	Native	Hyperaccu-mulator		Native	Native	Native
Percent depression of yield	10	50	50	50		50	50	50
Threshold	**39**	**> 6000**	**886**	**439**		**>500**	**493**	**283**
Growth medium	Sand-culture	Sand-culture	Sand-culture	Solution		Sand-culture	Sand-culture	Sand-culture
Reference	DeMarco et al., 1995	Paschke et al, 2005	Paschke et al., 2005	Xue et al., 2004		Paschke et al., 2000	Paschke et al., 2000	Paschke et al., 2000

[a] EC values are in Mg l^{-1}

Table 4. Comparisons of phytotoxicity shoot (root) thresholds (PT) reported in other studies for manganese and zinc.[a]

	Manganese					Zinc		
Species	Variety of grains and garden vegetables	Wheat	Subterranean clover (*Trifolium subter-raneum*)	Red River Gum (*Eucalyptus camaldu-lensis)*		Variety of grains and garden vegetables	Red River Gum (*Eucalyptus camaldu-lensis)*	Great Basin wildrye (*Leymus cinerus)*
Type	Agronomic	Agronomic	Agronomic	Woody		Agronomic	Woody	Native
Percent depression of yield	10	10	10	10		10	10	50
Threshold	**100-7000 Med = 500**	**570**	**2010**	**6510**		**70-900 Med = 260**	**370**	**2562 (2836)**
Growth medium	Various	Sand-culture	Sand-culture	Solution		Various	Solution	Sand-culture
Reference	Macnicol and Becket, 1985	DeMarco et. al., 1995	DeMarco et. al., 1995	Reichman et al., 2004		Macnicol and Becket, 1985	Reichman et. al., 2001	Paschke et al., 2000

[a] PT values are expressed in mg kg^{-1}

both agronomic and restoration species. However, the few woody species for which there are data have lower tissue thresholds compared to these two willow species.

Geyer willow had consistently higher EC thresholds to Mn than mountain willow. This might have been due to the spike that occurred in Geyer biomass at the first treatment level which was possibly the result of a synergistic effect of Mn on root growth (Blazich 1988). Despite the contrast in tolerance to external Mn, the internal values for the two species were surprisingly close (Table 2).

Table 5. Tissue levels of manganese or zinc reported in willows growing in various contaminated soils.

	Manganese					
Species	Geyer willow	Geyer and mountain willow	Geyer willow	Mountain willow	*Salix fragilis*	*Salix fragilis*
Health status [a]	Healthy	Chlorotic	Healthy	Healthy	NA	NA
Tissue concentration	**356 (leaf) 98 (stem)**	**PP[b]-yr1: 920 PP-yr2: 502 S-yr1: 165 S-yr2: 565**	**259**	**97**	**Stem 34 Leaves 312**	**Stem 969 Leaves 10760**
Growth medium	Pot experiment with amended mine tailing	In-situ amended mine tailings	Growing on floodplain adjacent to tailings deposits.	Growing on floodplain adjacent to tailings deposits.	Pot experiment with soil with 10 mg kg^{-1} available Mn	Pot experiment with soil with 100 mg kg^{-1} available Mn
Age	140 d	1 and 2 yrs	5-20 yrs	5-20 yrs	Age 1 yr-exposure 4 months	Age 1 yr-exposure 4 months
Reference	Bourret, 2004	Bourret, 2004	Bourret, 2004	Bourret, 2004	Small, 1975	Small, 1975
	Zinc					
Species	Geyer and mountain willow	Geyer willow	Geyer willow	Mountain willow	20 species or varieties of European willows	Willow spp
Health status [a]	Chlorotic	Healthy	Healthy	Healthy	Healthy or Chlorotic	Chlorotic
Tissue concentration	**Year 1 628 Year 2 903**	**645 (leaf) 249 (stem)**	**1233**	**867 (leaf)**	**Wood:35-227 Bark:220-458**	**Twigs: 2055 Leaves: 4484**
Growth medium	In-situ amended mine tailings	Pot experiment with amended mine tailing	Pot experiment with amended mine tailing	Growing on floodplain adjacent to tailings deposits.	Field soils contaminated with sewage sludge	Pot experiment with Zn contaminated soils
Age	1 and 2 yrs	140 d	120 d	5-20 yrs	1 yr	17 months
Reference	Bourret, 2004	Bourret, 2004	Fisher, 2000	Bourret, 2004	Pulford et al., 2002	Vyslouzilova et al., 2003

[a] The health status is a basic summation of the visual signs of the shrub as healthy or chlorotic. Most of these soils had elevated levels of multiple pollutants and cause(s) of chlorosis was not clearly determined.

[b] PP =pre-potted
S=staked

The Mn contents of willows growing in amended tailings along the Arkansas River are all below the phytotoxicity range presented here (Bourret 2004, Fisher 1999) (Table 1, 5). In contrast, when you compare the plants exposed to 100 mg kg^{-1} in this study with *Salix fragilis*, also exposed to 100 mg kg^{-1} (Small, 1975), the shoot values overlap (Table 2, 5). There may be several explanations for the relatively high Mn thresholds and uptake (when compared to the median for agronomic species) in willows. However, habitat adaptation is probably a key factor. Reichman (2004) screened seedlings of various woody species and suggested that the high tolerance observed for *Eucalyptus* spp. might be due to evolution of this genus in soils with high Mn availability as a result of waterlogging. The same may be true for willows and, therefore, they may be particularly useful in remediating sites contaminated by Mn.

Zinc thresholds

There are few available data by which to compare our LC50 values for Zn. The EC50 shoot values are comparable with those calculated for native grasses by Paschke et al. (2000) (Table 3). These thresholds will provide practitioners with an upper limit guideline for acceptable Zn levels for willow revegetation projects.

However, results from the tissue analysis indicate some important biochemical processes involved in Zn translocation were different in this system compared to a soil system. While our results show increasing Zn concentrations in the roots with very low and stable levels in the shoots (Table 2), other studies have demonstrated the mobility of Zn into the shoot (Watson 2003a, Verveake et al. 2003). Because tissue analysis of the shoots revealed no consistent increase in response to the treatments, it was only possible to report the range of shoot values for all the treatments (65 -139 mg kg^{-1}). This range falls at the very low end of the critical concentrations (PT10) summarized by Macnicol and Beckett (1985) and within the range of values considered normal (non-toxic) by Kabata-Pendias and Pendias (2001) (Table 4). Zinc concentrations reported for grass shoots are 10 to 100 times greater (Paschke et al. 2000). The most and least tolerant of the tree seedlings studied by Reichman (2001) demonstrated PT10 values closer to the median for agronomic species than to the higher shoot tissue concentration observed for these willows (Table 4).

Alone, these results suggest that Geyer and mountain willow do not translocate Zn. However, the tissue concentrations observed here should be interpreted with caution because the numbers are 10 fold lower on average when compared to shoot concentrations in willows from related studies (Bourret 2004, Fisher et al. 2000, Pulford et al. 2002, Vysouzilova et al. 2003) (Table 5).

Interpretation of thresholds determined in sand culture for plants growing in situ

The inconsistency observed between Zn shoot levels in this controlled sand-culture study with willows growing *in situ* emphasizes the need to consider critical distinctions between the systems. The differences can greatly influence availability and movement of the metals. Both Mn and Zn may be affected by many factors such as: redox levels, pH, temperature, mycorrhizal activity, rhizosphere and root development, complexation or chelation with other elements, antagonistic or synergistic relations with other metals, and compartmentalization into specific plant tissues (Blazic, 1988, Chaney 1993, Kabata-Pendias and Pendias 2001, Manthey 1994, Marschner 1988, Reisenauer 1988, Rufty 1979). For example, the presence of mycorrhizae has been shown to decrease Mn uptake thereby minimizing or alleviating Mn toxicity (Manthey et al. 1994). Furthermore, an increase in temperature has been shown to elevate Mn availability, uptake, and tolerance (Small 1975). Therefore, due to lack of mychorrhizae in this study and because the greenhouse temperatures were considerably higher than ambient temperatures on the Upper

Arkansas River, the tissue levels observed here might be elevated when compared to plants growing *in situ*.

On the other hand, the Zn shoot tissue concentrations observed in this study might be under representing phytotoxicity thresholds (PT). A positive correlation in Zn translocation has been observed with other elements such as Cu (Punshon and Dickenson 1997). In this experiment, Zn was provided in solution on different days from the nutrient solution which minimized the possibility for Zn to bind with other elements. Furthermore, several studies (Bourret 2004, Punshon and Dickenson 1997) have reported a spike in shoot and leaf concentrations of Zn immediately prior to senescence, yet this study did not mimic the completion of the growing season. A study aimed at comparing the thresholds calculated in this study with those determined for the same species growing in soils with known levels of metals would enhance our ability to relate results from controlled studies to field conditions.

Relevance of results for restoration of Arkansas River tailings

In a field study, Bourret (2004) amended tailings with lime and organic matter to increase the pH and reduce availability of metals to willows. Analysis of the tailings prior to amendment showed a pH of 5 and total Zn levels of 5278 mg kg^{-1} (Mn levels were not reported). One year after addition of the amendments, the pH was 6.5 and AB-DTPA Zn had been reduced to 73.8 mg kg^{-1}. Extractable Mn was reported to be 3.8 mg kg^{-1}. The willows that had been planted in these plots continue to show signs of chlorosis and poor vigor. The available Mn concentrations in these amended tailings were well below the LC50 or EC10 thresholds calculated in our study. Also, the leaves of these willows did not have any necrotic spotting as was characteristic for Mn.

In contrast, the level of available Zn (73.8 mg kg^{-1}) in the amended tailings is very close to the EC10-shoot thresholds calculated in this study (75 and 125 mg kg^{-1} for mountain and Geyer, respectively). Additionally, the interveinal chlorosis we observed in response to Zn was very similar to the symptoms exhibited by the plants growing on site. In conclusion, it is unlikely that Mn is contributing to the problems for these willows, yet the EC10 thresholds for Zn combined with the visual toxicity symptoms provide evidence that there may be a link between excess Zn and the chlorosis. Therefore, further studies screening both species and clones of willows endemic to the region for Zn tolerance is recommended.

CONCLUSIONS AND RECOMMENDATIONS

The results of this study show that Geyer and mountain willow have moderate tolerance to Mn and limited tolerance to Zn. This information should support the work of environmental risk assessors and restoration ecologists. Given the potential for use of willows in the restoration of metal contaminated riparian areas, further research should be conducted to establish willows as an effective restoration tool.

The tremendous range of tolerance and uptake demonstrated in previous studies on willow species and clones, and lack of research on willows native to the western US, warrants continued screening of willows for heavy metal thresholds. Specifically, screening for Zn tolerance of willow species and clones endemic to the area is necessary due to the link between the EC10 values and analysis of amended tailings along the Arkansas River. The interactive effect of Mn with other metals can potentially magnify overall toxicity through fluctuating bioavailability in wetland environments. From this study, it appears that willows are fairly tolerant to high levels of Mn. However, because Mn is rarely the sole pollutant in mine tailings, the dynamic and harmful effects of this element with other metals on willows needs to be studied.

Toxicity thresholds provide practitioners with guides for the tolerance of specific species to heavy metals. Because of the complexity of soil-plant systems, it is recommended to use and interpret toxicity thresholds in conjunction with other evidence such as visual toxicity symptoms, seasonal factors, and a complete soils analysis.

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TEN YEARS OF TESTING INDIGENOUS PLANT MATERIAL ON DRASTICALLY DISTURBED LAND IN WESTERN MONTANA

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The Development of Acid/Heavy-Metal Tolerant Releases (DATR) project began in 1995 with the seed collection and evaluate of native indigenous plant material from areas heavily impacted by historic mining and smelting activities in the Upper Clark Fork River watershed in western Montana. The Deer Lodge Valley Conservation District (DLVCD), in cooperation with the USDA-NRCS Bridger Plant Materials Center (PMC), has conducted this project with grant monies from EPA-Mine Waste Technology program and the Montana Department of Justice-Resource Damages Program.

Seven field studies have compared more than 500 local and non-local seed and plant collections in approximately 1,900 plots. The Woody Comparative Evaluation Planting, established in 2000, contains 19 accessions of seven native shrub and tree species (7 indigenous—12 non-indigenous). Top performers of the indigenous ecotypes are common snowberry, ponderosa pine, silver buffaloberry, wax currant, and Woods' rose. At the deep-plowed and lime-amended site on Stuckey Ridge, 87 accessions of grasses, forbs, and shrubs, including two mixes each of indigenous and non-indigenous material, were planted in 2003. Superior performing indigenous species include slender wheatgrass, basin wildrye, bluebunch wheatgrass, big bluegrass, western wheatgrass, and silverleaf phacelia.

Since the project's inception, four plants have been selected for pre-varietal release to the commercial seed industry: Washoe Selected class germplasm basin wildrye, Prospectors Selected class germplasm common snowberry, Old Works Source Identidfied class germplasm fuzzytongue penstemon, and Copperhead Selected class germplasm slender wheatgrass.

PARTICIPANT LIST

We were pleased to have a total of 183 participants at the Seventeenth High Altitude Revegetation Conference. Representatives from one foreign country and 14 states attended the conference (Table 1). As can be seen from the data presented in Table 1, most of the participants came from Colorado; however, people from around the country and from as far away as Spain participated.

For all of you that came, thank you for your participation. Make plans for attending in 2008. The High Altitude Revegetation Conference will be held in February or March, 2008 in Ft. Collins, Colorado. Pass the word to your colleagues, so that the 2008 conference will be a great success.

For current information on upcoming High Altitude Revegetation events, visit our website at www.highaltitudereveg.com.

Warren R. Keammerer
Editor

Table 1. Geographical distribution of participants at the Seventeenth High Altitude Revegetation Conference (March 7-9, 2006).

Geographic Entity	Number of Participants	Percent of Total Participants
SPAIN	1	0.55
UNITED STATES		
Arizona	2	1.09
California	3	1.64
Colorado	149	81.42
Georgia	1	0.55
Idaho	2	1.09
Minnesota	1	0.55
Montana	2	1.09
Nebraska	1	0.55
Nevada	3	1.64
New Mexico	2	1.09
South Carolina	1	0.55
Utah	4	2.19
Washington	1	0.55
Wyoming	9	4.92
Total	183	100.00

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SUMMARY OF SUMMER TOURS 1974-2005

Assembled by Wendell Hassell and Mike Ellis

Since 1974, the HAR Committee has sponsored biennial conferences and annual field trips to unique mountainous revegetation project and research sites. All Conferences have been held at Fort Collins, Colorado, in conjunction with CSU, except the 1980 conference, which was held at the Colorado School of Mines in Golden, Colorado. Summer Field Tours have been conducted at the following sites:

YEAR	AREA TOURED	SITES TOURED
1974	Vail/Climax, CO	Vail Ski Area, AMAX Climax Molybdenum Mine
1975	Empire, CO	AMAX Urad Molybenum Mine, Winter Park Ski Area, Rollins Pass Gas Pipeline
1976	Idaho Springs/ Silverthorne, CO	US Highway 40 Construction, Keystone Ski Area
1977	Aspen/Redstone, CO	Snowmass Ski Area, CF&I Pitkin Iron Mine, Mid-Continent Coal Redstone Mine
1978	Estes Park, CO	Rocky Mountain National Park
1979	Silverton/ Durango, CO	Purgatory Ski Area, Standard Metals Sunnyside Mine Bayfield Range Experiment Program
1980	Vail/Climax, CO	I-70 Vail Pass Highway Construction Revegetation Ten Mile Creek Channelization, Copper Mountain Ski Area, AMAX Climax Molybdenum Mine
1981	Crested Butte/ Gunnison, CO	AMAX Mt. Emmons Molybdenum Project, Western State College, Homestake Pitch (Uranium) Mine, CF&I Monarch Limestone Quarry
1982	Steamboat Springs, CO	Mt. Werner Ski Area, Howelson Hill Ski Jump, Colorado Yampa Energy Coal Mine, P&M Edna Coal Mine
1983	Rifle/Meeker, CO	CSU Intensive Test Plots, C-b Oil Shale Project Upper Colorado Environmental Plant Center, Colony Oil Shale Project
1984	Salida, CO Questa, NM	Domtar Gypsum Coaldale Quarry, ARCO CO_2 Gas Project Molycorp Molybdenum Mine, Red River Ski Area
1985	Cooke City, MT	USFS Beartooth Plateau Research Sites Bridger Plant Materials Center
1986	Leadville, CO	Peru Creek Passive Mine Drainage Treatment, California Gulch/Yak Tunnel Superfund Site, Colorado Mountain College
1987	Glenwood Springs/ Aspen, CO	I-70 Glenwood Canyon Construction, Aspen Ski Area
1988	Telluride/Ouray/ Silverton, CO	Ridgeway Reservoir, Telluride Mt. Village Resort, Idarado Mine, Sunnyside Mine

YEAR	AREA TOURED	SITES TOURED
1989	Lead, SD	Terry Peak Ski Area, Glory Hole and Processing Facilities of Homestake Mining Co., Wharf Resources Surface Gold Mines Using Cyanide Heap Leach
1990	Colorado Springs/ Denver, CO	Castle Concrete's Limestone Quarry, Cooley Gravel Quarry (Morrison), E-470 Bridge and Wetland near Cherry Creek. Littleton Gravel Pit Restoration to Parkland
1991	Central Colorado	Alice Mine, Urad Tailings, Pennsylvania Mine at Peru Creek, Yule Marble Quarry near Marble, and Eagle Mine Tailings and Superfund Clean Up near Minturn and Gilman
1992	Northern Colorado	Rocky Mountain National Park, Harbison Meadow Borrow Pit, Alpine Meadow Visitor Center, Medicine Bow Curve Revegetation, Hallow Well Park
1993	Central and Southern Colorado	Mary Murphy Mine, Summitville Mine, Wolf Creek Pass, Crystal Hill Project
1994	Northeastern Utah	Utah Skyline Mine, Burnout Canyon, Huntington Reservoir Hardscrabble Mine, Royal Coal, Horse Canyon Mine
1995	North Central Colorado	Eisenhower Tunnel Test Plots, Henderson Tailing Test Plots, Wolford Mountain Reservoir, Osage and McGregor IML Site Seneca II and 20 Mile Coal Mines (Steamboat Springs)
1996	Southwest Colorado	UMTRA Site (Durango), Sunnyside Mine (Silverton), Idarado Mine (Telluride), Southwest Seed Co. (Dolores)
1997	Southwest Colorado	Cresson Mine (Cripple Creek), San Luis Mine, Bulldog Mine (Creede)
1998	Lead, SD	Richmond Hill Mine, Wharf Resources, Homestake's Red Placer, Sawpit Gulch, WASP Reclamation Project
1999	Northern New Mexico	Molycorp's Questa Mine, Hondo Fire Revegetation Work, Pecos National Monument, El Molino Site, Cunningham Hill Mine
2000	Central Colorado	Boardwalk at Breckenridge, Eagle Mine, Independence Pass, and Climax Mine
2001	Estes Park, Colorado	Rocky Mountain National Park
2002	Western Colorado	I-70 Glenwood Canyon, CSU Intensive Test Plots, Upper Colorado Environmental Plant Center, Rocky Mountain Native Plants, Union Oil Shale Project
2003	Colorado Front Range Foothills	Hayman, High Meadow, Buffalo Creek and Walker Ranch Fire Sites
2004	Vernal, Utah Area	Upper Strawberry River Drainage Projects and Simplot's Phosphate Mine
2005	Colorado Front Range	Caribou "Mudfest" Restoration Site Lakewood Pipeline Project Cherry Creek State Park Bluff Lake Nature Center

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