Restoration Notes

Restoration Notes have been a distinguishing feature of *Ecological Restoration* for more than 25 years. This section is geared towards introducing innovative research, tools, technologies, programs, and ideas, as well as providing short-term research results and updates on ongoing efforts. Please direct submissions and inquiries to the editorial staff (mingram@wisc.edu and cmreyes@wisc.edu).

Differences in the Effects of Drought upon Restored and Remnant Prairies (Illinois)

Stuart K. Allison (Dept of Biology, Knox College, Galesburg, IL 61401, 309/341-7185, Fax: 309/341-7718, sallison@knox.edu)

This study was motivated by two observations. First, I noted distinctly visible differences in vegetation, and more particularly amount of bare space, between restored and remnant tallgrass prairies when I sampled these sites during a severe drought as part of a long-term monitoring protocol. Restoration at the prairies I study began in 1955 (Allison 2002) and I expected such well-established restored prairies to respond like remnant prairies to climatic events, such as drought. Second, I found no relevant studies as I checked the literature for information on drought effects on restored prairies. Grassland ecosystems, such as North American prairies, regularly experience drought; global climate change models predict more frequent and severe droughts in these communities in the not too distant future (Stewart 1986). Considerable effort has been expended to restore prairie ecosystems across North America since ecological restoration began with tallgrass prairie in the 1930s (Howell and Jordan 1991). Despite the fact that interest in prairie restoration has increased since the 1980s, as has public awareness of the potential effects of global climate change, there has been little study of the effects of drought on prairie restorations. I was surprised by the qualitative differences that I observed in vegetation and decided to statistically analyze the data and document my observations as a starting point for future research.

My research was conducted at six sites in northern Illinois, all relatively flat and located on black mesic silty loam soils. The three restored prairies are located at the Green Oaks Field Research Center in Knox County: East (2.8 ha), South (4.8 ha), and West (7.7 ha). The three remnant prairies are Lost Meadow (0.4 ha), also located at Green Oaks, Brownlee Cemetery Prairie Nature Preserve (0.6 ha) in Mercer County, and Spring Grove Cemetery Prairie Nature Preserve (0.5 ha) in Warren County. All six sites are typical tallgrass prairies in terms of species composition: dominated by C_4 grasses and complex mixtures of forbs (Table 1). The prairies are described in more detail by Allison (2002). All of the sites are managed by prescribed burning, although the regimes differ, ranging from annual fires at two remnant prairies (Brownlee and Spring Grove) to three-year fire return intervals for all of the prairies located at Green Oaks.

I sampled the restored and remnant prairies late in the summer (mid-August to early September) during 2000, 2001, 2002, 2004, and 2005. Sampling consisted of randomly placing 25-m belt transects within the prairies according to their size to avoid oversampling smaller sites, and then identifying all plant species that occurred within 1 m of the central transect line. West and South Prairies contained five transects, East contained four, and the remnants each had three. I used five 0.10-m² quadrats per transect, spaced 5 m apart, to collect data on plant abundance and live plant cover and identifying the plants occurring at 25 points on the 0.10-m² grid.

I compared the restored and remnant prairies by examining the percentage of bare space, the species richness, and the percent cover of the 10 most abundant species as determined by percent cover and the number of quadrats containing the species in each prairie. The top ten species constitute more than 80% of the plant cover and thus can reasonably act as a proxy for the entire plant community.

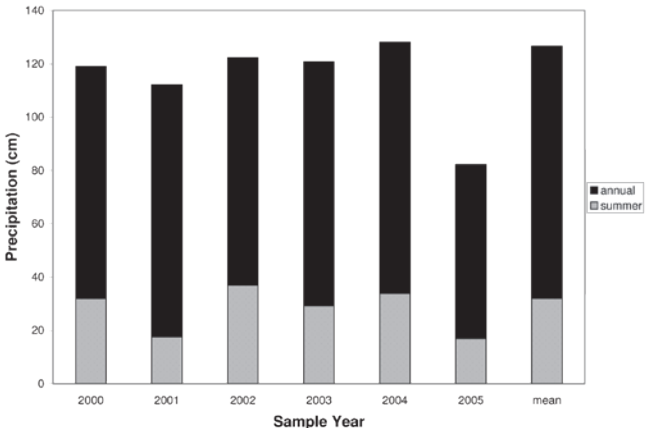

Figure 1. Summer and annual precipitation (cm) as recorded at the weather station at Galesburg, Illinois (ISCO 2006). Mean precipitation is normalized climatic data collected from 1971 to 2000.

Ecological Restoration Vol. 26, No. 2, 2008
ISSN 1522-4740 E-ISSN 1543-4079
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Table 1. Mean percent cover (averaged for all years) and change in cover (owing to drought) for the most abundant species sampled during the summers of 2000–2002 and 2004–2005 in Illinois prairies. Mean percent cover values and changes in cover are given for the ten most abundant species in each prairie (eleven in Lost Meadow owing to a tie). Numbers in bold indicate a decline in mean percent cover during drought years; n.c. means no change.

	Restorations						Remnants					
	East		South		West		Brownlee		Lost Meadow		Spring Grove	
Species	Mean	Change	Mean	Change	Mean	Change	Mean	Change	Mean	Change	Mean	Change
Grasses												
Andropogon gerardii	**35.40**	**-24.0**	30.11	+2.13	22.46	+0.34	31.73	-1.74	**26.03**	**-7.50**	15.63	+1.16
Schizachyrium scoparium	26.96	+17.44	14.94	+6.10	35.64	+9.32	**9.86**	**-0.80**	11.09	+2.90	**9.17**	**-0.38**
Sorghastrum nutans	**2.40**	**-1.10**	**11.17**	**-8.69**	**5.15**	**-4.35**	**7.15**	**-7.15**			**14.40**	**-9.87**
Forbs												
Astragalus canadensis							12.00	+7.60				
Ceanothus americanus							**2.61**	**-1.01**				
Desmodium canadense			**1.02**	**-0.38**					**2.77**	**-0.11**		
D. canescens									1.76	+2.10		
Equisetum arvense							**0.85**	**-0.31**				
Eryngium yuccifolium	2.86	+2.44	**2.27**	**-0.19**								
Euphorbia corollata							3.20	+0.26	2.45	+1.28	6.29	+2.64
Helianthus divaricatus									8.16	+3.30		
H. giganteus	0.32	-0.02			0.64	n.c.	4.96	-3.63			10.18	+2.75
H. hirsutus							8.64	+7.49			**2.45**	**-0.99**
H. mollis	**2.44**	**-1.60**			**17.98**	**-5.74**						
Lespedeza capitata					**2.08**	**-1.04**						
Melilotus alba											**4.53**	**-3.87**
Monarda fistulosa							**5.12**	**-1.13**				
Rhus glabra									**2.19**	**-1.13**		
Rubus allegheniensis									**3.36**	**-0.97**		
Silphium lacinatum			1.98	+0.90	**1.63**	**-1.07**					**14.02**	**-1.09**
S. terebinthinaceum	**2.24**	**-0.94**	2.21	+0.19								
Solidago altissima	6.28	+3.52	**3.97**	**-1.69**	1.15	+0.29			12.69	+4.11	3.04	+2.55
S. gigantea	11.56	+0.34	**20.45**	**-2.29**	**2.72**	**-0.64**			**16.48**	**-7.42**		
S. juncea									1.76	+2.64		
S. rigida			1.25	+0.91	2.34	+0.46						
Veronia gigantea	**2.28**	**-1.48**										

Bare space and species richness were compared by repeated measures ANOVA, followed by a posteriori pairwise comparisons of sample years if the repeated measures ANOVA revealed significant differences among sample years. All statistical tests were performed using Systat, version 5.03 (Systat Software, Inc.).

In 2001 and 2005 summer precipitation was only slightly greater than half of the approximately 32-cm mean summer precipitation for this area (Figure 1). The total precipitation in 2005 was far below the mean (94.5 cm), and so 2005 was considered to be a drought year. However, 2001 received almost exactly mean annual precipitation owing to wet periods in the spring and fall.

Summer drought conditions significantly increased the amount of bare space in restored prairies to more than double the amount in non-drought years ($F = 15.5$, df = 4,452, $p < 0.001$). Although assessing plant growth by measuring a negative—bare space due to lack of plant growth—is not the ideal way to examine drought effects, it does illustrate the fact that remnant prairies may be more resistant to short-term climatic events than restored prairies—even well-established ones.

Species richness was significantly higher in remnant than in restored prairies ($F = 100.513$, df = 1,113, $p < 0.001$), except in 2000. This general pattern was contrary to the expectation that old, well-established prairie restorations would be very similar functionally to remnants. I suggest that even 50 years is insufficient for prairie restorations to converge with remnants.

Three grasses were the most abundant species (Table 1): big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), and Indiangrass (*Sorghastrum nutans*). The other abundant species consisted of several composites (Asteraceae, mostly goldenrods (*Solidago* spp.), rosinweeds (*Silphium* spp.), and sunflowers (*Helianthus* spp.)) and a few legumes (Fabaceae: milkvetches (*Astragalus* spp.), ticktrefoils (*Desmodium* spp.), lespedezas (*Lespedeza* spp.), and the invasive sweetclover (*Melilotus* spp.)). The

ten most abundant species make up 89–96% of the live plant cover in the restored prairies but only 83–89% of live plant cover in the remnants. Most striking is that in the restored prairies the dominant grasses account for 56–65% of the cover but only 37–49% of the live plant cover in the remnant prairies.

The changes in cover for the most abundant species during summer drought show no overall pattern. There was no clear difference between restored and remnant prairies in this respect. Most of the observed species increased in some sites while decreasing in others. The only exception was Indiangrass, which declined in every prairie where it was a dominant species.

A key question raised by my study is why the increase in bare space (or decrease in plant growth) between restored and remnant prairies, such that restored prairies are more negatively affected than remnant prairies? It is possible that there were genetic differences between plants in the remnant and restored prairies, such that remnant prairie plants were better adapted to resist drought than their conspecifics in restored prairies. However, the original restorationists at Green Oaks were careful to harvest local seed, mainly from prairie remnants in cemeteries (much seed came from Brownlee Cemetery Prairie) and along railroads (Schramm 1992, Allison 2002). Thus, it is likely that plants in both remnant and restored prairies were similarly adapted to local conditions.

Several potential mechanisms may be at work, all of which require differences in development or establishment of the plant communities between restored and remnant prairies. It is possible that differences in species composition and species richness may account for differential resistance to drought conditions between remnant and restored prairies (Knapp 1985, Martin et al. 1991). However, the changes among the ten most abundant species are not consistent from prairie to prairie, making it difficult to perceive patterns caused by compositional differences. It is also possible that differences in belowground processes, such as development of roots and/or mycorrhizal associations lead to distinct drought responses (Daniels-Hetrick et al. 1986, Hayes and Seastedt 1987).

In the end, I have found an interesting pattern: drought had a greater negative impact on restored prairies than on remnant prairies, as revealed by significantly more bare space (and thus less plant growth) in the restored prairies and changes in the abundance of the 10 most common species in each prairie. The causal mechanisms are unknown at this time and will have to be determined through future research. Because drought severity and frequency are expected to increase in prairies owing to global climate change, restoration ecologists will have to figure out these relationships to help plan for long-lasting grassland restorations in North America and the rest of the world.

Acknowledgments

I would like to thank the Illinois Nature Preserves Commission for granting me permission to work at the Brownlee Cemetery Prairie Nature Preserve and the Spring Grove Cemetery Prairie Nature Preserve. The Knox College Faculty Development Fund helped support my research. Frank McAndrew, Jennifer Templeton, Mrill Ingram, and Chris Reyes provided valuable comments on a draft of this manuscript. Peter Schwartzman helped me gain access to and interpret the climate data. My reference librarian, Sharon Clayton, was extremely helpful as I searched the literature.

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Restoration in a Failed State: Community-Based Agroforestry in Haiti

Starry D. Sprenkle (Friends of Hôpital Albert Schweitzer Haiti, Haiti Timber Re-Introduction Project and Ecology Graduate Group, Dept of Plant Science, University of California, 1 Shields Ave, Davis, CA 95616, sdsprenkle@ucdavis.edu)

As the poorest country in the Western Hemisphere (Sletten and Egset 2004), the small Caribbean nation of Haiti is often characterized as an environmental and

human disaster. In 1492, Haiti had an estimated 80% forest cover, which declined to 50% in 1900 and 21% in 1945 (Pierre 2001). Agriculture, logging, and charcoal production have reduced forest cover to just 1% today (Woodwell 2002, Woodring et al. 1924). Haiti is mountainous, and as a result of deforestation, much of the topsoil has washed away, leaving severely eroded slopes (Figure 1) and drying aquifers (Pearce 2002). Deforestation has a direct, negative impact on both wildlife and the human communities that rely primarily on agriculture. People living in mountain communities experience higher rates of malnutrition and other poverty-related diseases (Perry et al. 2007), thanks in part to the lower productivity of their lands.

Friends of Hôpital Albert Schweitzer Haiti, a nonprofit organization supporting the nonprofit hospital that has been in operation for over 50 years, has developed the Haiti Timber Re-Introduction Project (HTRIP). Representing a proactive approach to community health that deals directly with the underlying issues of poverty and environmental degradation, HTRIP aims to restore soil fertility and forest cover in the mountains. The primary goal is to establish human-oriented, economically driven forest systems, with ecosystem services and benefits to native flora and fauna as secondary objectives.

Established in 2005, HTRIP is a large-scale experiment in community-based, sustainable agroforestry. Its grassroots approach is a product of dialogue involving the project manager, other staff members, village participants, and local agricultural extension agents. To diminish the risk of illegal harvesting in response to continuing high demand for fuel and commercial timber, the project works directly with individual small-scale farmers who will have a personal stake in the survival of the trees. In this sense, HTRIP is a joint venture with the farmers, who supply land and labor in exchange for seedlings and training. A similar joint-venture strategy had some success in Haiti in the past (Murray and Bannister 2004).

Each year, HTRIP staff identify 10 new locations where land is steep, soil is eroded, and at least 25 local residents are interested in participating. First, HTRIP creates a 1,000-m² demonstration plot in each new community (Sprenkle forthcoming). Second, 20–30 community members attend a year-long series of monthly education sessions (Figure 2) while building and operating a community nursery, and then plant seedlings on their own land after training is completed. Education topics include tree care and maintenance, seed collection and storage, nursery preparation and management, soil conservation, soil organic matter/composting, and agroforestry. As this cycle is repeated annually in each community with a new set of students, HTRIP staff will gradually train local leaders over four years to teach the sessions, manage the nursery, and oversee the plantings. This strategy increases the local, independent capacity for tree planting and keeps staffing costs low.

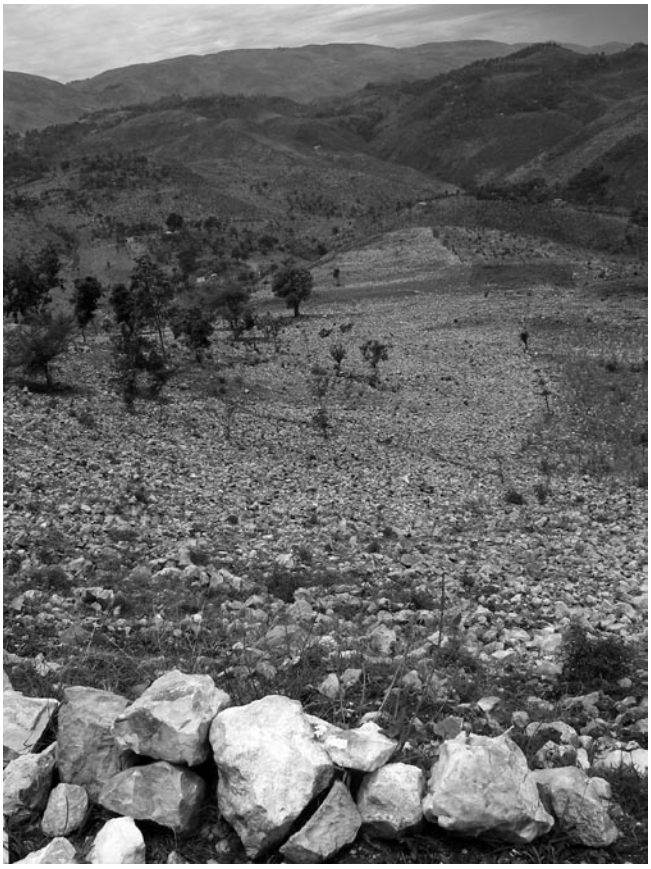

Figure 1. Looking straight down a rocky, eroded slope in the Chaine des Mateux, in the Haiti Tree Re-Introduction Project area. The clusters of trees in the distance are fruit trees growing around small communities. Photos by author

Figure 2. Women discuss the importance of trees during an HTRIP education session. The program uses an interactive, image-based learning method.

The project encourages mixed-species planting to include timber, fruit, and nitrogen-fixing trees. The composition of each nursery is tailored to the local community and depends on multiple factors, such as seed supply (although HTRIP facilitates the trading of seed between

groups), participant interests, and HTRIP recommendations based on environmental conditions. The project has experimented with both native and non-native species, seeking to identify those that survive best and grow fastest. Monitoring provides important feedback for species selection (Sprenkle forthcoming).

Because the goal is sustainable agroforestry, HTRIP has a strong agricultural component. For the first 6–8 years (depending on tree species composition) as the transplanted seedlings grow, the traditional full-sun crops of corn and millet can be cultivated in the reforestation plot, and HTRIP "loans" the farmers crop seed, which each farmer returns to the community seed bank at harvest time for the next year. This partial agricultural subsidy can provide an added incentive for participation in the program. During this period, the trees are thinned to the final 3-m spacing (1,108 stems/ha) through an intermediate harvest of pole-size trees. Once the tree canopy closes, the farmers will have several options: grow forage for livestock, transition to annual shade-tolerant crops, permanently switch to shade-tolerant perennial species like coffee, or switch to nonagricultural land use. Farmers will be assisted in executing their plans, with HTRIP encouraging them to follow only one rule: replant harvested trees.

Several significant challenges arose in two of the ten participating communities in 2006–2007, such that we almost terminated project activities there. The cases illustrate two major concerns of community-based projects in general: corruption and property rights. In one community, a local leader diverted project funds intended for participant meals. The community became discouraged, the demonstration plot fell into disrepair, nursery production faltered, and there were no volunteer work days for the first few months of the 2007 planting season. The project responded by transferring the leadership role to a charismatic, hard-working community member who had continued to labor in the nursery and demonstration plot despite the difficult situation, showing exceptional dedication to the project. After the transition, the community rallied to revive its demonstration plot and establish five new parcels in 2007, and participation in education has increased dramatically. This experience underscores the importance of knowing the local community, choosing a committed local leader, and having the flexibility to address problems as they arise.

A second group is wrestling with the problem of land ownership. This community is comprised mainly of young adults, originally from mountain villages, who moved to the urban fringe to attend secondary school. The adjacent land is deforested and uncultivated, probably because the few large landowners cannot control tree harvest and other activities on their extensive holdings. Like many rural-to-urban migrants worldwide, these individuals are unable to commute to their home communities to farm. They want to participate in the program, but need to find accessible land. The project brought them into negotiation with landowners to work out a share-cropping agreement that explicitly addresses rights to annual crops and tree products. One potential arrangement is for the sharecropper to keep 100% of annual crops (usually the owner takes a portion every year) and the products of tree pruning, and 10–20% of the final tree harvest, in exchange for stewardship of the trees. The landowner would retain the majority of tree ownership. These negotiations are still in progress, but if an agreement is reached, it could be a model for reforestation in places where lands remain degraded and unproductive owing to absentee landownership. Such an agreement would benefit both the landless urban migrants and the landowners, and increase local forest-based environmental services such as flood and erosion control, soil fertility improvements, habitat for beneficial fauna, and improved aquifer recharge.

The HTRIP project seeks sustainability on a number of levels. On an individual plot scale, agroforestry creates an economically sustainable system by pairing agricultural activity for short-term income with forestry for long-term benefits. The project is socially sustainable because it relies on local expert staff, fosters local leadership, and cooperates with other regional groups interested in agricultural and forestry development, including the Mennonite Central Committee and Educational Concerns for Hunger Organization. Although the project is only two years old, already it is exhibiting signs of success. There are 134 graduates of HTRIP's first year of activity (10 communities) who planted almost 20,000 seedlings in 2007. The graduates continue to have contact with HTRIP staff and the local leader. The project has expanded into 10 new communities (20 total), and approximately 500 new students are participating in 2008. Over 50,000 seedlings will be produced and planted by these participants this summer, and these numbers are likely to increase each year. We are currently developing a special socioeconomic monitoring program to evaluate long-term improvements in economic and health status, the key project goal. We hope to continue project activities at least through 2020 or 2025.

Acknowledgments

Project funding is provided by the R.K. Mellon Foundation, Snavely Family Foundation, Linneman Family Foundation, Thorton D. and Elizabeth S. Hooper Foundation, and numerous private donors. The Hôpital Albert Schweitzer (HAS) Haiti provided an invaluable base of infrastructure. The author would especially like to recognize the contributions of Christian M. Snavely, Ian and Lucy Rawson, Shannamar Dewey, and Erlantz Hyppolite to the development and success of HTRIP. Finally, the author thanks Kevin Rice, Mark Schwartz, and Truman Young and his lab for their help in developing this document.

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Does Biodiversity–Ecosystem Function Science Apply to Prairie Restoration?

Amy J. Symstad (U.S. Geological Survey, Northern Prairie Wildlife Research Center, Black Hills Station, Wind Cave National Park, Hot Springs, SD 57747, 605/745-1191, asymstad@usgs.gov)

Biodiversity–ecosystem function (BEF) science investigates how biodiversity (the diversity of organisms, including genetic, species, and landscape diversity) affects ecosystem function, including properties (e.g., aboveground plant biomass, soil organic matter content) and processes (e.g., nutrient cycling, pollination). This relatively new science integrates community and ecosystem ecology (Naeem 2006). Wright and colleagues (forthcoming) propose that integrating community and ecosystem ecology via a BEF approach will result in more successful ecological restorations. The BEF approach assumes that there is an inherent relationship between biodiversity and ecosystem function: diversity and ecosystem function are positively related up to a point, with the greatest increases in function occurring at low levels of diversity.

Grasslands all over the world, including the prairies of North America, can and have served as testing grounds for BEF theory. The evidence from many research studies supports the assertion that biodiversity and ecosystem function are positively related. In these studies, more diverse experimental plots exhibited greater standing plant biomass, lower available soil nitrogen (i.e., greater resource uptake), less invasion by unplanted species, and greater temporal stability of aboveground plant biomass, on average, than lower diversity plots (Tilman et al. 1997, 2001, Biondini 2007, Piper et al. 2007). Based on these results, it should follow that restoration of ecosystem function relies on the restoration of diversity in prairies.

However, two elements of the design of these experiments may make restoration practitioners question the applicability of their results to restoration projects. First, low-diversity treatments in BEF experiments often comprise a random selection of species from the high-diversity treatments (e.g., Tilman et al. 1997, 2001, Biondini 2007). For example, if the high diversity treatment has 32 species, the composition of a 4-species treatment is determined by randomly drawing 4 species from this 32-species pool. This practice is necessary to statistically tease out the effect of diversity itself from that of individual species, but it can yield species combinations that would never be considered for a prairie restoration—a prairie with no grasses, for example. In contrast, low-diversity restorations are usually composed of matrix species (i.e., dominant grasses) because they comprise the majority of biomass in natural prairies. Second, seeding rates, particularly of some forb species, used in BEF experiments are often not economically feasible in restorations. For example, in one experiment (Piper et al. 2007), the seed for treatments of just 12 species would cost over $250,000 per hectare. Both of these elements of experimental design raise two important questions: 1) how does greater biodiversity affect ecosystem function when that greater diversity is accomplished only through the addition of rarer species; and 2) are the extra expense and effort required to establish and maintain higher diversity balanced by the services provided by that diversity?

Experiments designed to answer these questions have yet to be conducted. Building upon the results of previous prairie restorations and grassland plantings provides a practical solution to this problem. The perennial grass plantings of the U.S. Department of Agriculture's Conservation Reserve Program (CRP) are a prime example of the ecosystem approach to grassland restoration, in which the goal is to restore ecosystem function regardless of composition. The CRP was established in 1985 in part to reduce soil erosion on agricultural land. In the program, farmers are paid to plant highly erodible land currently under cultivation into some type of perennial cover. These plantings have achieved the ecosystem function goal of reduced erosion (Young and Osborn 1990). In addition, the plantings, especially those using native species, have produced some unforeseen ecosystem benefits such as soil carbon storage (Baer et al. 2002) and habitat for some wildlife species (Dunn et al. 1993). These ecosystem functions have been

achieved with relatively low diversity—often fewer than five matrix grass species and perhaps a legume—compared to a standard restoration mix (e.g., one restoration company's catalog lists 5–6 grass species and 24–38 forb species for their prairie mixes) or native prairie (60–100 species in 1000 m^2). Experimentally adding a range of numbers and types of species to these existing CRP plantings could help answer the questions posed above.

On the other hand, very diverse prairie restorations exist, but their level of ecosystem functioning has rarely been examined. Most prairie restorations use a community approach, in which the primary goal is to achieve diversity and community composition similar to that of native prairie. For example, the prairie restorations at Homestead National Monument in Beatrice, Nebraska, and at Knox College in Galesburg, Illinois, were planted in 1939 and 1955, respectively. Sixty-seven years after its establishment, the Homestead prairie's community structure resembled native prairie and contained over 100 species, nearly as many as nearby remnant prairie (National Park Service 2006). In 1999, the total number of species at the Knox College restoration was similar to that found in nearby remnant prairies, although the average number of species in sampling plots was lower in the restoration than the remnants (Allison 2002). BEF scientists could benefit from measuring ecosystem function in these and other community restorations of various ages (e.g., Camill et al. 2004), compositions, and levels of diversity, as this would provide an additional means for evaluating the BEF relationship in communities more realistic, both in composition and in size, than many constructed in BEF experiments.

These examples, although by no means exhaustive, are just some ways that BEF science could be conducted in the context of ecological restoration. Both fields are concerned with the same global issue—biodiversity loss and its impacts on the ecosystem services upon which humans rely. BEF science is now focusing on understanding the mechanisms behind species interactions, the traits of species that affect those interactions, and how these affect the relationship between biodiversity and ecosystem function. For example, plots with nitrogen-fixing legume species and C_4 (warm-season) grasses in agriculturally depleted, sandy soil in a Minnesota BEF experiment had greater soil carbon storage than plots without this combination because the legumes increased the input of nitrogen to the system and the C_4 grasses efficiently used that nitrogen in building large root systems (Fornara and Tilman 2008). Similar analyses could help develop restorations that achieve carbon storage or biofuel production goals, as developing markets for both of these commodities could easily involve prairie restorations (Tilman et al. 2006). Greater collaboration between restoration practitioners and BEF scientists would produce similar synergy for other ecosystem services provided by prairie restorations, such as habitat for native bees that pollinate adjacent crops, and soil and nutrient retention to protect surface and ground water. The practitioners must be equal partners in this collaboration, however, because they understand the costs of restoration projects and what information they need to improve restoration success. This need for greater connection between scientists and practitioners is not unique to BEF science and prairie restoration, but the overwhelming concordance of their goals invites mutually beneficial collaboration.

Acknowledgments

Thank you to Justin Wright for valuable conversations on this topic and to Dan Wenny, Brenda Molano-Flores and Marie Curtin for valuable comments on an earlier draft of the manuscript.

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Restoration of Soil Physical and Biological Stability Are Not Coupled in Response to Plants and Earthworms

Bryan S. Griffiths (Teagasc, Environment Research Centre, Johnstown Castle, Wexford, Ireland, bryan.griffiths@teagasc.ie), Qin Liu (Institute of Soil Science, Chinese Academy of Sciences, PO Box 821, Nanjing 210008, People's Republic of China), Huili Wang (Institute of Soil Science), Bin Zhang (Institute of Soil Science), Hsueh L. Kuan (Environment Plant Interactions Programme, Scottish Crop Research Institute, Invergowrie, Dundee DD2 1NE, UK), Blair M. McKenzie (Environment Plant Interactions Programme), Paul D. Hallett (Environment Plant Interactions Programme), Roy Neilson (Environment Plant Interactions Programme) and Tim J. Daniell (Environment Plant Interactions Programme)

Degraded soils typically suffer from loss of organic matter, biological activity, and aggregation, leading to poor plant establishment, erosion, and compaction. There is a close connection between the biological and physical stability of soils expressed as their resistance (ability to retain characteristics despite stressors) and resilience (ability to recover from stress-induced changes) (Griffiths et al. 2007b). Plants and earthworms may restore degraded soil by improving its physical and biological properties. Plants improve physical structure directly by enmeshment of soil particles by the roots and their exudates, and indirectly by stimulating microbial biomass and the production of microbial polysaccharides (Feeney et al. 2006). Earthworms also stabilize physical structure and organic matter (Haynes and Fraser 1998). Surface-active species are known to incorporate surface organic matter and so enhance microbial activity, while burrowers incorporate organic matter at depth and improve aeration and drainage (Lee 1985).

We conducted a greenhouse experiment to compare the effects of plant and earthworm additions on the physical and biological properties of subsoil. Degraded topsoil shares many properties with subsoil: low organic matter concentration, small microbial biomass with limited activity, small seed bank, fewer roots, and poorer aggregation. For this experiment we used subsoil from 20–40 cm depth that was classified as a Dystric-fluvic cambisol and consisted of 71% sand, 19% silt, and 10% clay with a pH (H_2O) of 6.2. Soil was sieved < 2 mm, mixed with fertilizer (110 mg/kg NPK, 14:14:21) and added to 40 containers (50 × 30 × 28 cm) lined with 2-mm diameter polyester mesh on gravel. The containers were watered manually and kept in an unheated greenhouse with additional lighting (16-h day length) for five months, after which we destructively sampled them.

There were five replicates of eight treatments: 1) control (—); 2) organic matter only (—G); 3) plants only (P—); 4) plants and organic matter (P-G); 5) plants and mixed worms (PM-); 6) plants, organic matter, and burrowing worms (PBG); 7) plants, organic matter, and surface worms (PSG); and 8) plants, organic matter, and mixed worms (PMG). We added fresh organic matter (the equivalent of 0.47g chopped dry grass/kg, with a C:N ratio of 4.9). White clover (*Trifolium repens* cv. crusader) and perennial ryegrass (*Lolium perenne* cv. bellevue) seeds were sown at a rate of 30 kg/ha on September 7. Plant height was kept at

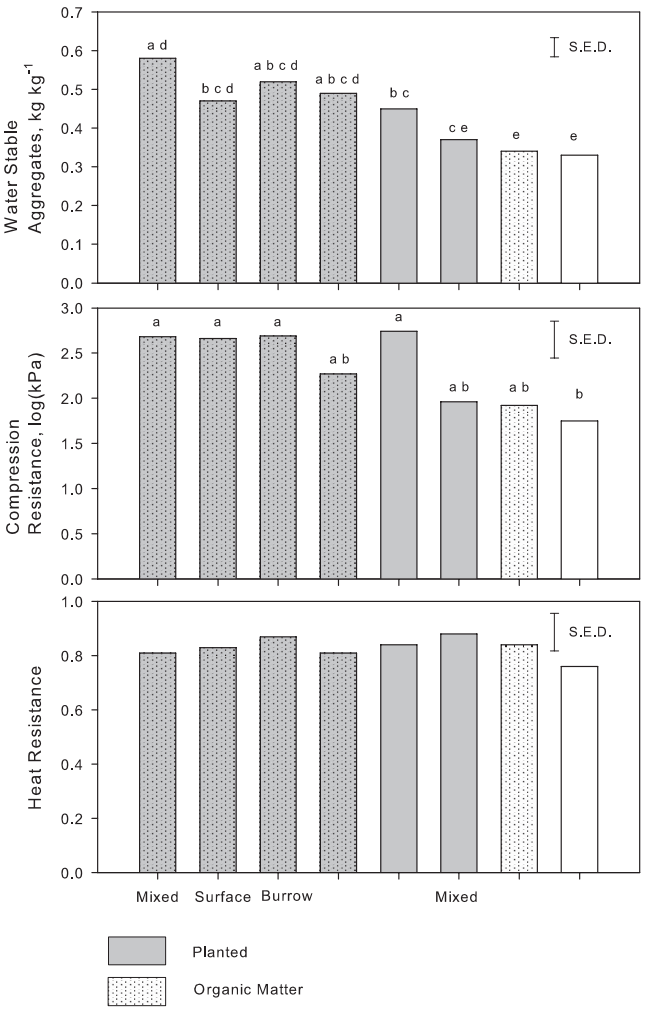

Figure 1. Examples demonstrating that physical stability (water stable aggregates and compression), but not biological stability (heat resistance), responded to restoration treatments. Grey bars indicate planted soils, dots represent the addition of organic matter, and earthworms are indicated on the X-axis. Bars with the same letter are not significantly different at $p < 0.05$; the average standard error of the difference of the mean (SED) from the ANOVA is shown.

Table 1. Mean values for physical and biological soil properties in the form of plant shoot mass (g); microbial biomass measured as total ester-linked fatty acids (ELFA, nmol/g) and community structure indicated by principal component 1 from analysis of ELFA; water-stable aggregates (WSA, g/g), physical resistance (1/Cc), and resilience (RC) to compression; biological resistance (% of control) and resilience (cumulative recovery) to copper or heat stress in soils. Treatments consist of combinations of plants (P), organic matter (G), surface-active worms (S), burrowing worms (B), mixed worms (M), or no additions (-). In the lower section, bold type indicates that the individual treatment factor significantly influenced the soil property in that column.

Treatment	Plant shoot mass	Microbial		Physical Stability			Biological Stability			
		Biomass	Community	Structure	Resistance	Resilience	Resistance		Resilience	
		ELFA	PC1	WSA	1/Cc	RC	Copper	Heat	Copper	Heat
				mean values						
PMG	16.3	51.0	2.5	0.58	2.68	0.54	0.37	0.81	4.19	24.54
PSG	18.8	47.2	2.6	0.47	2.66	0.47	0.26	0.83	4.58	24.41
PBG	24.1	49.9	2.2	0.52	2.69	0.49	0.39	0.87	4.13	24.99
P-G	23.8	45.1	-0.6	0.49	2.27	0.47	0.38	0.81	3.76	24.82
P—	20.6	65.5	-1.0	0.45	2.74	0.52	0.48	0.84	2.90	23.81
PM-	11.6	36.3	-4.3	0.37	1.96	0.40	0.42	0.88	3.85	23.42
—G	n/a	71.7	1.3	0.34	1.92	0.39	0.48	0.84	3.89	23.81
—-	n/a	41.7	-3.3	0.33	1.75	0.36	0.31	0.76	2.50	24.08
				p-values						
Plant	n/a	0.894	0.200	< **0.001**	**0.019**	**0.011**	0.831	0.202	0.141	0.391
Organic matter	0.088	0.656	< **0.001**	**0.007**	0.357	0.334	0.582	0.260	**0.030**	0.070
Worms	**0.036**	0.234	0.497	0.846	0.861	0.456	0.212	0.563	0.196	0.674
Worm type	0.145	0.660	0.631	0.501	0.640	0.904	0.194	0.772	0.917	0.596

12 cm with cuttings returned to the soil surface to mimic animal grazing, stimulate root growth, and provide a more natural earthworm habitat. Earthworms were collected from the field on October 18, kept in moist compost at 15°C for 2 days, split into the two functional groups, and added to appropriate containers.

We analyzed ester-linked fatty acids (ELFA) to determine microbial biomass and microbial community structure (Griffiths et al. 2007a). We measured physical stability as a function of both aggregate stability in water (Tisdall and Oades 1979) and resistance and resilience to compression (Zhang et al. 2005). Biological stability is the resistance and resilience of decomposition processes to either a transient heat stress or a persistent heavy-metal contamination stress in the form of copper (Kuan et al. 2007). Statistical analyses were performed with GenStat (version 9 for Windows, VSN International) and consisted of an unbalanced Analysis of Variance (ANOVA) and principal component analysis of ELFA data followed by ANOVA.

In general, the planted treatments significantly increased soil physical stability (Table 1). In contrast, while microbial biomass increased during the experiment, none of the treatments significantly altered biological stability compared to the control (Figure 1). Moreover, the resulting microbial community structure showed no clear trends. These outcomes suggest that biological stability develops more slowly than physical stability and may require an interaction between microbial community structure and the developing physical structure.

The plants and mixed worms treatment (PM-) gave responses intermediate between the planted and unplanted treatments in terms of shoot mass, microbial biomass, and physical stability. PM- and soil with no additions had significantly ($p < 0.05$) less ELFA than other treatments. The PM- carbon inputs may have been lower (reducing shoot growth and microbial biomass compared to the other planted treatments), but the microbial community structure in this treatment was also very different from the other planted treatments. The results are anomalous and will need to be examined further.

In this study plants played a greater role in soil restoration than earthworms or organic matter. The relatively short time for the experiment might account for the lack of a worm effect (Table 1), although other laboratory studies with worms did record significant effects within a few weeks (such as Haynes and Fraser 1998).

It is not only important to plant the appropriate vegetation for restoration, but also to understand the processes occurring in soil that link physical and biological properties. Some ecological responses, such as biological stability of soil, clearly take longer to restore and would require monitoring for a longer period. Thus, indicators of soil quality should ideally be multifunctional and link above- and belowground processes.

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A GIS Application: Fire Line Location for Prescribed Burns (Texas)

Matt Crawford, Ernest B. Fish and Carlton M. Britton (Texas Tech University, Dept of Natural Resources Management, Fire Ecology Center, Lubbock, TX 79409-2125, 806/742-2841, carlton.britton@ttu.edu)

Fire lines are typically located along pasture fences for prescribed burning in Texas. In rough, hilly terrain this means that fire lines traverse steep slopes and deep canyons, greatly increasing expense and creating hazardous conditions for the people involved in ignition operations. Several variables contribute to a final placement decision, including line widths, wind direction, wind speeds, fuel type, and time of year. Determining where a fire line needs to be located has always been a labor-intensive field procedure requiring considerable time.

We propose an innovative approach to optimize the fireline planning process that will save time and money and enhance safety. Our ultimate goal is to develop a user-friendly GIS application that generates a digital model of an area and draws the best possible fire lines around a burn unit perimeter by predicting the smoothest and safest path. This process should become a useful and simple tool for collecting information and quickly determining the correct management scheme, which can then be exported to a GPS unit to guide bulldozer operators constructing the fire line.

To achieve the end result, several activities needed to be accomplished: 1) comprehensively review terrain analysis literature; 2) establish and test a set of algorithms for a roughness factor (Riley et al. 1999); 3) determine appropriate software and model default values; 4) develop the GIS application; and 5) in the future develop a least-cost model for factors influencing fire line placement.

Our literature review showed that terrain analysis applications were designed for a wide range of uses, from habitat corridor analysis to Mars rover applications. Many of these studies used a combination of three similar types of analyses: terrain trafficability, terrain roughness, and least-cost path (Walker and Craighead 1997). In our study, terrain roughness was found most useful in defining the best route across any area.

Initially we found a set of algorithms that provide the roughness measurements calculated from a Digital Elevation Model (DEM) (Hoffman and Krotkov 1989, Li et al. 2005). These calculations are automated with a script and then incorporated into a Geographic Information System (GIS) (Donlon and Forbus 1999). In one study we reviewed (Moreno et al. 2003), we found an application written by Riley et al. (1999) that measures the amount of elevation difference between adjacent cells of a DEM. This program was written in Arc Macro Language (AML), which is a language specific to Arc/INFO. When the application is applied to DEM data, it provides what we call a Terrain Roughness Index (TRI).

We tested this roughness model using a DEM simulation of a 640-ha pasture on the Rocker Ranch, located in Borden County west of Gail, Texas. Topography ranges from an almost flat canyon bottom to the surrounding walls of the Llano Estacado Escarpment, with elevation ranging from 810 to 890 m (Figure 1). The new DEM-generated grid, which we refer to here as terrain roughness index (TRI), was cut to the study area boundary.

Next, we determined a set of "default" values based on findings from the available applications. Digital Elevation Model data are most often packaged in 30-meter resolution

Figure 1. Photo of the Rocker Ranch pasture (near Gail, Texas) used as the study area, showing vegetation and topography. Photo courtesy of David B. Wester

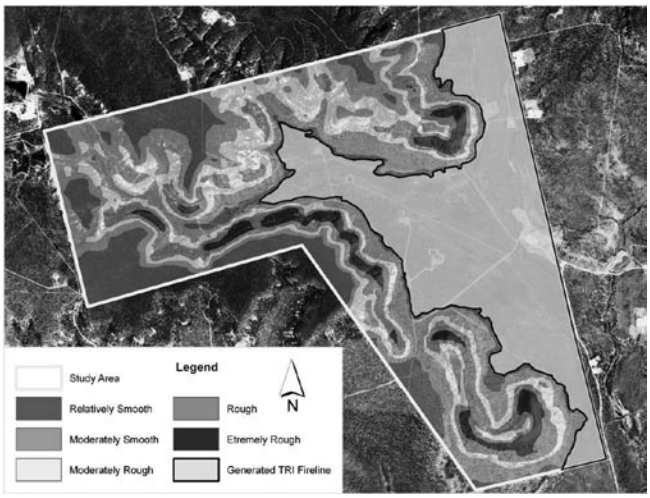

Figure 2. An aerial photo of the study site (Rocker Ranch, Texas) showing terrain roughness categories and proposed fire lines determined from the TRI test.

as a standard. The most important value to be considered is scale; the smaller the scale, the more sensitive the roughness resolution. Scale is based on the ability of the "vehicle" to traverse an area, which for our purposes was a bulldozer or tractor. The TRI application is a simplified calculation tool that measures the differences in elevation and does not consider other variables such as slope and aspect, which will be considered in a later model.

Finally, we developed the proposed application in a form that will enable a user to manipulate, analyze, and process data with ease. We chose ArcGIS 9 (ESRI, Inc.) as the most user-friendly interface with the associated ArcObjects Developer Kit as the best application platform. Visual Basic.NET was the language used to rewrite the program to run in ArcGIS 9. The end result will be converted into a tool within the ArcGIS user interface.

The TRI program was a functional start toward the goal of this study. It calculates the differences in elevation from one grid cell and its surrounding cells. The program is able to create a new surface within seconds. Attempting to create a user-friendly interface for this application has been difficult. After the TRI surface was developed, creating a pathway around the desired area was initially done by selecting the Create Contour button on the Spatial Analyst toolbar and clicking the desired range of values (Figure 2). Automating this process is difficult because every surface is different. Therefore, no standard range of values exists that can be placed in a code to create a program that fits all situations. Presently, however, this set of procedures does significantly reduce the amount of time to plan a prescribed burn.

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Status of Russian Olive Biological Control in North America

Dan Bean (Colorado Dept of Agriculture, Biological Pest Control Program, 750 37.8 Rd, Palisade, CO 81526, dan.bean@ag.state.co.us), Andrew Norton (Dept of Bioagricultural Sciences and Pest Management, Colorado State University, Ft Collins, CO 80523, andrew.norton@colostate.edu), Roman Jashenko (Central Asian Biological Control Laboratory (CABCL), 93 Al-Farabi St, Almaty, 050060, Kazakhstan, romajashenko@yahoo.com), Massimo Cristofaro (BBCA , Via Anguillarese 301, Rome 00123, Italy, mcristofaro@casaccia.enea.it) and Urs Schaffner (CABI Europe—Switzerland, Rue des Grillons 1, CH-2800 Delemont, Switzerland, u.schaffner@cabi.org)

Russian olive (*Elaeagnus angustifolia,* Elaeagnaceae), a Eurasian species, has now become the fifth most abundant woody plant in riparian areas of the western U.S. (Friedman et al. 2005). First introduced to the West in the late 19th century, Russian olive has been promoted for the past 100 years as an ornamental, a windbreak, or even as wildlife habitat (Olson and Knopf 1986). Serious ecological consequences include reduced quality of wildlife habitat and increased competition with native plant species (Katz and Shafroth 2003). Russian olive has invaded riparian areas more slowly than its riparian coinvader, tamarisk (*Tamarix* spp.), but the invasion is now similar in scope and severity and shows no signs of slowing. Biological control may offer a much needed management tool.

Control efforts are controversial, especially biological control, since Russian olive is considered valuable. However, wildlife habitat value may be enhanced by Russian

olive removal (Gazda et al. 2002), and native substitutes are available for other uses. Nonetheless, the perceived value of Russian olive has hindered noxious weed listing, which limits or bans sale or transport of such species. Colorado listed Russian olive in 2001 and banned nursery sales after 2003, but existing trees are allowed to stand at the discretion of local weed-control authorities. Russian olive is on the class C weed list in New Mexico and is listed as potentially invasive, and is banned in Connecticut. Wyoming has recently added the plant to its State Noxious Weed List, and sale or movement of the plant is now banned.

In a classical weed biological-control program, specialist herbivores from the plant's native range are released in the introduced range to suppress populations of the invasive weed. This is an ecological control method, since it involves the reunification of the weed with one or more species of natural, self-propagating controls (for a good general reference on weed biological control, including permitting, see Coombs et al 2004). The concept is straightforward, but the process is complex and time consuming owing to legitimate concerns over potential nontarget impacts of released agents (Louda et al. 2003).

Permitting for importation, open release, and interstate movement of biocontrol agents is regulated by the United States Department of Agriculture, Plant Health and Inspection Service, Plant Protection and Quarantine (USDA APHIS PPQ). USDA APHIS PPQ consults with a panel of experts representing a number of agencies, known as the technical advisory group (TAG), before at least two of the critical regulatory steps (foreign activities and issuing open field permits) in the process. Regulatory details may appear mundane to the end users of biological control but they determine the speed at which agents become available, the safety of introduced agents, and the types of agents that are ultimately available for weed-management programs.

The biocontrol process can be broken into three main steps: the preliminary phase, the foreign activities phase, and the domestic activities phase (Drea 1991). The preliminary phase includes extensive literature reviews to compile existing knowledge of the target and its natural enemies. It also includes an evaluation of the feasibility and necessity for development of a biocontrol program, with a focus on the severity of the invasion and the lack of other control options against the target.

Russian olive is in the foreign activities phase, which begins with extensive and often far-reaching surveys to locate herbivores that feed exclusively on the target plant in its native range and are most likely to suppress the weed in North America. Overseas laboratories then verify specialized host requirements by offering candidate agents nontarget plants, including crop species and plants native to the new range. Special attention is given to species that are closely related to the target, since specialist feeders are far more likely to utilize a close relative of their host than distantly related plants. There are only four North American species in the family Elaeagnaceae, which increases the likelihood of finding biological control agents that are specific enough to warrant introduction. However, two of these species, silver and russet buffaloberry (*Shepherdia argentea* and *S. canadensis*), are on state threatened or endangered species lists, so host-range testing of candidate agents will pay particular attention to these two potential nontarget host species. Agents that can develop on nontarget plants are likely to be eliminated from further consideration.

Three overseas laboratories, the Central Asian Biological Control Laboratory (CABCL, Almaty, Kazakhstan), CABI-Europe (Delemont, Switzerland), and BBCA (Rome, Italy), are doing extensive surveys of herbivores in Kazakhstan, Uzbekistan and eastern Turkey, Iran, and China. These labs have also initiated host range and other characterizations of these herbivores.

In Kazakhstan CABCL has found 30 insect species that commonly use Russian olive as a host**.** These include 17 species that appear to be host-specific enough for consideration as biological control agents. Each of these species has been evaluated using a number of criteria, including damage to the plant in the native range, size and density of the herbivore population under natural conditions, and potential ease of handling and host range testing under quarantine conditions. The three most promising species are undergoing further study in Kazakhstan. Two of the species are psyllids (Psyllidae, Homoptera), relatively small insects that feed by sucking plant juices. The psyllids (*Trioza magnisetosa* and *T. furcata*) under consideration were observed to cause major damage to Russian olive, up to 100% foliage loss, and appear to be very host specific. The third promising agent from the Kazakhstan fauna is a leaf beetle, *Altica balassogloi* (Coleoptera, Chrysomelidae). The leaf beetle causes substantial damage to Russian olive in its native range by feeding on foliage and shoots; it achieves high densities and is relatively common. These features, combined with the relative ease with which chrysomelid beetles can be cultured and tested and their success in past biocontrol programs (DeLoach et al. 2003), makes this species an excellent biocontrol candidate.

CABI and BBCA have been especially interested in the discovery and development of biocontrol agents that feed on the flower buds, flowers, fruit, seeds, or seedlings. Such agents would decrease the reproductive potential of the trees, slowing the spread of the plant without reducing the value of existing trees. This strategy, used successfully in the management of invasive trees in South Africa (Moran et al. 2003), may satisfy most of the concerns of those who value existing Russian olive trees, defusing controversy and possibly accelerating the permitting process. Extensive surveys in three regions of Uzbekistan and preliminary surveys in eastern Turkey, northeastern Iran, and northwestern China have yielded several such promising agents. These include a newly described flower-feeding eriophyid mite, *Aceria angustifoliae*, a fruit-feeding moth (probably

Ananarsia eleagnella), and a fly and a weevil that have not yet been identified to the species level. Other potential agents were also found, including a shoot tip miner, *Temnocerus elaeagni* (Coleoptera, Rhynchitidae), three wood borers, *Chlorophorus elaeagni* (Cerambycidae), *Megamecus cinctus* (Curculionidae) and *Euzophera* sp., and two defoliating moths, *Hyles hyppohaes* (Lepidoptera, Sphingidae) and an unidentified geometrid.

Foreign exploration is continuing and we are optimistic that additional potential agents will be discovered through further and more extensive exploration in the native range of Russian olive. The herbivore assemblage that was found in Turkey differed significantly from what was found in Uzbekistan or Kazakhstan, indicating significant regional heterogeneity in the species composition of Russian olive feeders. This diversity can provide a suite of potential biocontrol agents that in combination may prove effective in the U.S.

The final, domestic activities, phase will begin with the establishment of disease-free colonies of potential biocontrol agents in established USDA quarantine facilities, probably those in Temple, Texas and Sidney, Montana. Additional host-specificity testing may be done, if needed; then field testing and full scale releases will follow, after appropriate permits are obtained (Drea 1991).

An important and frequently asked question is, "When will Russian olive biocontrol agents be available for use in the United States?" The foreign activities phase of Russian olive biocontrol must continue for several more years to ensure program success (Drea 1991), but the domestic phase may soon begin and run concomitantly. It is difficult to predict a date for implementation, since there are many variables in the process, but for a realistic projection we can turn to the tamarisk biocontrol program, which is similar in scope, significance, and controversy. For the current tamarisk program, literature review began in 1987, and foreign exploration and testing started in 1991 and continues to the present day. The domestic phase of the tamarisk program was initiated in 1992, and the first test releases of a biocontrol agent were made in 2001 (DeLoach et al. 2003). Widespread release of the tamarisk leaf beetles (the implementation phase) didn't begin until 2005. Considering the tamarisk time frame, we expect another 10 years before widespread release of the first Russian olive biocontrol agents.

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A Lidar Approach to Evapotranspiration Estimation in Riparian Areas (New Mexico)

W. Eichinger (University of Iowa, Iowa City, IA 52242, william-eichinger@uiowa.edu), J. Prueger (USDA Soil Tilth Laboratory), D. Cooper (Los Alamos National Lab), L. Hipps (Utah State University), C. Neale (Utah State University), H. Holder (Duke University), S. Hanson (U.S. Bureau of Reclamation) and S. Bowser (Bureau of Reclamation)

The river provides 97% of the water used in the New Mexican part of the Rio Grande Basin (Hansen 1996). Because of rapid urban and industrial growth, water demand in the basin has increased 35% over the past 15 years and is expected to more than double in the next 50 years. The U.S. Bureau of Reclamation manages water releases from the reservoirs to maintain the Rio Grande water level while ensuring that water obligations in New Mexico, Texas, and Mexico are fulfilled. The single largest loss is evapotranspiration from the river and adjacent riparian areas. Efficient use of the available water requires accurate estimates of the evaporative water demand along the river, since too little released water results in unmet demands and too much released means "excess" water is "wasted" to the sea.

To accomplish this goal, the Bureau has developed the ET Toolbox, a set of online tools that help water managers

along the Rio Grande gauge water use and allocate it effectively (Brower 2004). The ET Toolbox uses information on vegetative canopy type and daily meteorological data (temperature, humidity, and wind speed) from stations along the river as inputs for algorithms that estimate evapotranspiration. Traditional formulations such as Blaney-Criddle and crop coefficients (Unland et al. 1998, Shuttleworth 1993) badly under-predict evaporative water use in high demand situations, such as hot weather or high winds, because they rely on assumptions about conditions averaged over larger areas. However, areas near the river are a small oasis in the middle of desert, so that evaporation is difficult to measure and highly sensitive to conditions that are not average. Actually measuring evapotranspiration is prohibitively expensive, and so the Bureau of Reclamation is developing more effective, physically based algorithms to estimate water losses.

Conventional evaporation instruments (eddy correlation or Bowen Ratio, for example, [Brutsaert 1982]) measure at a single point located above the canopy, outside the wake caused by the surface vegetation, and yet close enough to ensure only local emissions contribute to the measurement. In the narrow vegetation bands along the river this is often difficult to achieve, but scanning, solar-blind water vapor Raman lidar overcomes this limitation by measuring the three-dimensional water vapor concentration over an area, making it possible to clearly identify the target measurement zone. Evaporation is estimated by determining the evaporative pattern that would cause the measured water vapor distribution, using Monin-Obukhov similarity theory (Eichinger et al. 2000). A comparison of lidar and eddy-correlation estimates of evapotranspiration is shown in Figure 1 (Nichols et al. 2004). The range of evaporation rates shown in the figure demonstrates why the use of point data may lead to unrealistic estimates of evapotranspiration.

The lidar is a laser radar that uses a short pulse (10 ns) of ultraviolet light (248 nm) to excite water vapor and nitrogen molecules. The resulting emissions are used to measure the absolute water vapor concentration to about 3% accuracy (Cooper et al. 1994). Details of this particular instrument, built at Los Alamos National Laboratory, are described by Eichinger et al. (1999). The lidar samples the returning light at 100 MHz with a spatial resolution of 1.5 m along each line of sight. Over the study area, usually about 1 km² in size, approximately 25 million measurements of water vapor concentration, all at different locations and heights above the canopy, are made in a half hour. Averaging the data to generate a three-dimensional map of water vapor concentration and inverting these measurements to obtain evapotranspiration rates is computationally intensive, currently requiring about 4 hours to process 30 minutes of data. A typical study lasts from 10 to 14 days, with the lidar operating more or less continuously.

Maps of evapotranspiration over the study area are generated by assembling all of the lidar measurements within

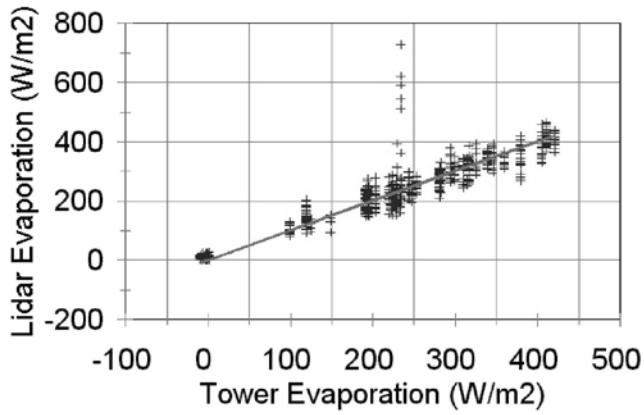

Figure 1. A comparison of evapotranspiration estimates obtained from eddy correlation instruments and lidar measurements within 75 m of the instrument tower at Bosque del Apache, New Mexico (R^2 = 0.88, RMSD = 15.3%). Each point represents a half-hour average. The large differences at 240 W/m² come from an afternoon with extremely high winds that make the eddy correlation values suspect.

each 30-min period. Figure 2 is an example of such a map compared to a thermal map (on right) generated during the same observation period as part of an airborne visible/near infrared imaging effort by C. Neal of Utah State University to extend evaporation measurements to larger areas along the river. The lidar is located on the center of a north-south sand levee. A relatively uniform, dense salt cedar (*Tamarix ramosissima*) canopy is to the east, while less dense cottonwoods (*Populus deltoides* ssp. *wislizenii*) are northwest. The dark strip west of the levee is the drainage canal running parallel to the river. In general, the maps correlate well. As expected, areas of high evapotranspiration (light shades) in the lidar map are associated with cool surface temperatures (darker greys) in the thermal image; conversely, areas of low evapotranspiration (dark grays) are associated with higher surface temperatures (light gray). However, the associations are not exact, thanks to the lidar footprint caused by a slight downwind shift in local moisture conditions (discussed in detail in Cooper et al. 2000). To see Figure 2 in color, visit http://www.wisc.edu/wisconsinpress/journals/journals/er_suppl.html.

The lidar has been used in field campaigns in three areas of New Mexico to provide otherwise lacking surface evaporation estimates. The first trial at Bosque Del Apache Wildlife Refuge provided an ideal opportunity to quantify evapotranspiration from riparian salt cedar and cottonwood stands. Of particular note are the large variations in evapotranspiration over salt cedar canopies (Figure 2): locations differed as much as 43%. This variability parallels differences in crop coefficients measurements, even in the same stand. Because lidar can measure a large area, it is able to obtain more reasonable average estimates.

The second trial quantified open water evaporation from Elephant Butte Reservoir, roughly 60 km long and 2–4 km wide and running north-south in the semiarid middle Rio Grande Valley. Conventional methods, such as Penman

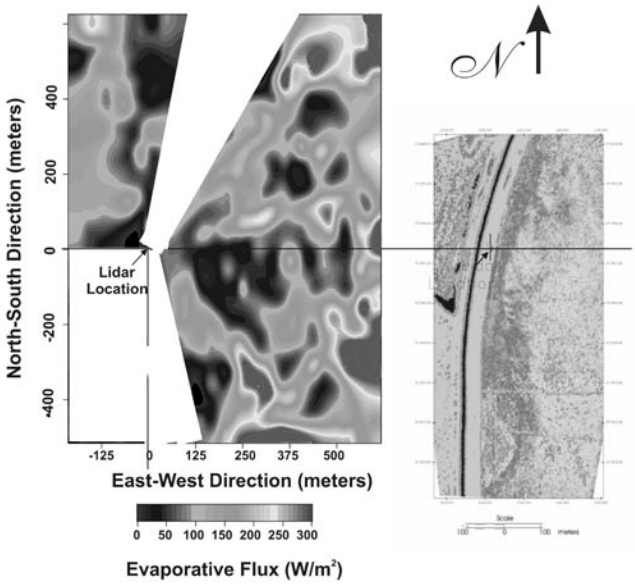

Figure 2. An evaporation map (left) of the Bosque del Apache site; light grays are highest rates, dark grays lowest. To the right is a thermal infrared aerial photograph; light grays are warmest, dark grays are cool. The two plots show the lidar location, are to the same scale, and are aligned horizontally for ease of comparison. The canopy to the east of the lidar is salt cedar and to the north west are cottonwood trees.

Monteith, that rely on solar radiation (Shuttleworth 1993) poorly predict evaporative losses from the reservoir. Wind speed (enhances vertical mixing) and direction (northerly winds tend to have moister air, while dry air comes from the west) greatly influence evaporation. The new lidar-derived algorithms reduced estimation error from over 60% to about 8% (Eichinger et al. 2003).

The current study measures evaporation from riverine sand bars. The field campaign was located at the south end of Rio Grande Nature Center State Park in Albuquerque, where the river has a sand and gravel bed with low, sandy, erodible banks. The presence of numerous wandering bars and middle islands is strongly characteristic of this reach of wide river channel. While data analysis is not yet completed, preliminary results indicate that wet sand bars evaporate at the same rate as the river.

Lidar and state-of-the-art meteorological instruments are too expensive for long-term use. However, lidar is advantageous in locations where instrumentation is not possible or provides ambiguous results. These campaigns are providing new and more accurate algorithms to the ET Toolbox, which in turn will provide better information to water planners. As water demand increases, effective use of a limited resource becomes ever more important.

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Identifying Control Techniques for *Rumex acetosella* in the Presidio of San Francisco (California)

Mark Frey (Presidio Trust, 34 Graham St, San Francisco, CA 94129, 415/561-4148, mfrey@presidiotrust.gov), Jennifer Soong, Jessica Feeser and Samuel Dishy

Sheep sorrel (*Rumex acetosella*) is a dioecious perennial that forms a persistent seed bank (Fitzsimmons and Burrill 1993), spreads vigorously from root buds and by wind-dispersed seeds (IPANE 2004), and is naturalized throughout North America (USDA 2007). As early as 1889, sheep sorrel was listed as one of the "worst weeds" in the United States (Halsted 1889), and is considered invasive or listed as a noxious weed in many states. It is found in a wide range of habitats, but especially open fields and disturbed places. In San Francisco's Presidio, it outcompetes native species, particularly annuals, and invades a broad range of habitats that also contain 15 federally listed endangered plant species.

Sheep sorrel has a dense root system that is difficult to remove and current control techniques routinely leave behind root fragments. We surmised that if we could identify root parameters, in terms of size or depth, at which control efforts are no longer needed, we could then modify current strategies to maximize efficiency. Here, we report on a greenhouse experiment to identify depths at which root fragments pose the greatest risk of resprouting and a field experiment to evaluate potential control techniques.

In January 2006, we filled 42 tree pots (15.2 cm diameter) with sand, the most common (and easily collected) substrate in the Presidio's natural areas. Each pot received one root fragment 12.7 cm long, which fit into the container without overlap. Root fragments were measured with calipers and sorted into six width classes (< 0.5, > 0.5–1, > 1–1.5, > 1.5–2, > 2–3, and > 3 mm). Within each size class, one root was planted at each of seven depths. These ranged in 5-cm increments from the surface to the maximum depth of 30 cm; the typical rooting zone reaches to 20 cm below the surface (Kiltz 1930). Limited greenhouse space and labor prevented the use of replicates. The pots were placed in a greenhouse for a year, kept moist with overhead spray, and monitored for emerging leaves. We rearranged pots every month to minimize confounding effects.

Within one month all roots with diameters less than 3 mm placed at the soil surface produced leaves (Table 1). After two months, roots in every size class buried at 5 cm also produced leaves. In the thickest size class, the root at the surface did not produce leaves, but the root buried at 10 cm produced leaves after three months. No more plants emerged after three months. No roots buried 15 cm or deeper produced leaves.

On January 22–23, 2007, for the field study we established eight pairs of 1-m² plots in a sandy site and seven pairs at a serpentine site within large patches of dense (> 30% cover) sheep sorrel. The perimeter of each plot was cut with a flat shovel down to a depth of approximately 0.25 m, severing all shallow roots to reduce inputs from plants outside the plots. In addition to a control, seven techniques that have been used on the Presidio were tested:

1) *Flame*: use a propane torch to wilt the leaves
2) *Hoe*: use a hula hoe to cut plants just below the soil surface and leave on site
3) *Pull*: without digging up plants, pull and remove from the site
4) *Pick*: with a hand-pick, remove from the site as much of the root and shoot as possible
5) *Smother*: cover vegetation with a black, semipermeable landscape fabric for six months
6) *7.6-cm dig*: remove 7.6 cm of soil, sift through a 0.635-cm screen, and return to the plot
7) *15-cm dig*: remove 15 cm of soil, sift through a 0.635-cm screen, and return to the plot

Table 1. Presence (+) or absence (-) of leaves sprouting from sheep sorrel (*Rumex acetosella*) root fragments buried in greenhouse pots in January 2006. Nothing emerged from depths greater than 10 cm and no changes were observed after April.

Root Width (mm)	Soil Depth (cm)	Date		
		Feb	Mar	Apr
< 0.5	0	+	+	+
	5	-	-	-
	10	-	-	-
0.5–1.0	0	+	+	+
	5	-	+	+
	10	-	-	-
1.0–1.5	0	+	+	+
	5	-	+	+
	10	-	-	-
1.5–2	0	+	+	+
	5	-	+	+
	10	-	-	-
2–3	0	+	+	+
	5	-	+	+
	10	-	-	-
> 3	0	-	-	-
	5	-	+	+
	10	-	-	+

The last was limited to sandy plots because heavy soils made this treatment impractical at the serpentine site. Other treatments were applied to four plots total, two per site. We measured plant cover in each plot immediately before implementing experimental treatments. The labor involved in each treatment was recorded (person minutes per 1 m²) and averaged for each pair of plots. Once a month for a year we estimated percent cover by covering the center 0.25 m² of each plot with a 25-point intercept grid: each point overlying sheep sorrel was treated as four percent cover. Smother plots were included in monitoring once the fabric was removed. To simplify the presentation of data that showed monthly variations within plots, possibly owing to the coarse sampling technique, we averaged plot data in three-month increments.

The pick and both dig methods reduced sheep sorrel cover, while flame treatment appears to be ineffective (Table 2). In sand, the pick and 30-cm dig methods still showed a reduction in percent cover one year after implementation. In serpentine, no treatments showed an effect after one year, but the 15-cm dig treatment reduced cover until the third quarter. All techniques appeared to be less effective in serpentine, and, except for flaming, also took longer to implement owing to denser soil. The seven methods required dramatically different time investments (Table 2).

The greenhouse and field experiments both yielded information that can guide management efforts. Techniques that remove root biomass appear to reduce sheep sorrel cover most effectively. The pick method is the most commonly

Table 2. A comparison of treatments to control sheep sorrel (*Rumex acetosella*) implemented in January 2007 at El Presidio, California. The mean time to implement each treatment (n = 2) was recorded, as well as the mean percent cover (± SE) before (shaded column, n = 2) and after (averaged over three-month periods, n = 6) treatment. Smother plots were included in monitoring immediately after fabric was removed on July 5, 2007.

		Labor	Percent Cover				
Soil	Treatment	(min/m²)	Jan	Feb–Apr	May–Jul	Aug–Oct	Nov–Jan
sand	Control	0	30 ± 2	50 ± 10	33 ± 13	55 ± 12	18 ± 5
	Flame	0.6	48 ± 16	35 ± 15	80 ± 2	85 ± 7	49 ± 14
	Hoe	4	42 ± 14	11 ± 6	35 ± 10	49 ± 11	35 ± 14
	Pull	2	34 ± 6	8 ± 3	35 ± 2	35 ± 9	27 ± 7
	Pick	7	40 ± 12	1 ± 1	8 ± 7	16 ± 6	3 ± 1
	Smother	4	34 ± 6	n/a	n/a	8 ± 1	16 ± 4
	7.6-cm dig	20	32 ± 4	1 ± 1	9 ± 2	11 ± 2	21 ± 6
	15-cm dig	34	34 ± 2	0 ± 0	3 ± 1	5 ± 1	2 ± 1
serpentine	Control	0	54 ± 6	43 ± 4	65 ± 8	52 ± 5	15 ± 4
	Flame	0.4	56 ± 8	26 ± 11	69 ± 2	61 ± 2	47 ± 3
	Hoe	6	46 ± 12	13 ± 7	76 ± 10	57 ± 14	50 ± 9
	Pull	9	64 ± 6	28 ± 10	54 ± 14	61 ± 6	37 ± 8
	Pick	16	74 ± 2	11 ± 6	56 ± 7	48 ± 11	41 ± 5
	Smother	6	56 ± 4	n/a	n/a	9 ± 3	29 ± 7
	7.6-cm dig	31	54 ± 4	1 ± 1	29 ± 5	25 ± 5	17 ± 3

used at the Presidio and it appears to be the best choice to provide long-term reduction in cover while taking only 25% of the time to dig the entire plot. However, further study is needed to assess the relative time costs of each option at larger scales.

Roots left on the soil surface pose a high risk of producing new plants, so practitioners are advised to remove plant fragments when implementing control techniques. Follow-up is essential because roots left on the surface are likely to produce leaves within the first month. Small root fragments left buried at 15 cm or deeper appear unlikely to produce new plants. Practitioners can improve their efficiency by focusing on the top 10 cm of soil and being most concerned about larger roots.

Roots at greater depths take longer to produce leaves, which may lead managers to favor a strategy of repeated applications. However, neither experiment assessed repeated treatments. It may be that choosing techniques that are quick to implement and repeating them at regular intervals would be more effective than any single application discussed here.

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Efficacy of Different Glyphosate Concentrations in Managing Glossy Buckthorn (*Frangula alnus*) Resprouts (Michigan)

R. Gregory Corace III (Seney National Wildlife Refuge, Seney, MI 49883, 906/586-9851 x14, greg_corace@fws.gov), Kenneth P. Leister and Erin Brosnan

Invasive species management is a high priority for many governmental agencies and nonprofit conservation groups with ecological restoration mandates. Since 2001, glossy buckthorn (*Frangula alnus*) has been intensively managed at Seney National Wildlife Refuge (SNWR) and on adjacent Michigan Department of Natural Resources properties in Upper Michigan with some success (Nagel et al. this issue). Owing to concerns for the environmental, human health, and treatment costs associated with herbicide use, utilizing the lowest possible concentration of any pesticide is desirable (Relyea 2005). Because a review of the literature yielded little in terms of the known efficacy of different glyphosate concentrations on managing glossy buckthorn resprouts and because no standard operating procedure exists for the management of this species on

National Wildlife Refuge System lands, we pursued research on the effectiveness of glyphosate (brand name *Gly Star*™) at various concentrations to manage glossy buckthorn resprouts arising from previously treated stems.

Our study area was located on an upland portion of an anthropogenic dike at SNWR dominated by glossy buckthorn. In 2003, mature glossy buckthorn shrubs were treated by first cutting stems with diameters > 2.5 cm, and then applying a 20% active ingredient (a.i.) solution of glyphosate (plus 0.05% nonionic surfactant) to the resulting stumps. However, this treatment produced vigorous resprouting from cut glossy buckthorn stems (see Nagel et al. this issue). In 2006 we set up 60 plots, each consisting of a single multistemmed cut stump, spaced at least 0.5 m apart. We recorded the total number of resprouts and their average height in each plot during initial observations. The mean (± SD) number of pretreatment resprouts was 17.1 (± 10.8). In general, resprouts arising from the same multistemmed stump were around the same height. In 48% of the plots, resprouts were 2–3 m in height, with the remaining resprouts less than 2 m in height.

Typically, glossy buckthorn plants and resprouts < 2.5 cm in diameter have been treated with a 5% a.i. solution of glyphosate at SNWR based on the anecdotal experience of colleagues in Michigan working in various ecoregions and ecosystems. Consequently, to document the efficacy of different concentration of glyphosate in managing resprouts the plots were divided evenly into four treatment groups: 0% (tap water without surfactant), 1.25%, 2.5%, and 5.0%. These concentrations were chosen based on the general experience and advice of colleagues from the State Department of Natural Resources, The Nature Conservancy, and other member organizations of the Michigan Invasive Plant Council, as well as general herbicide label recommendations. All solutions were diluted with tap water according to the label. During dry and stable weather conditions on June 9, 2006, we sprayed the resprouts once using a hand-held (low-volume) pump so that approximately 50% of the surface of all leaves was covered, but not sprayed to the point of dripping off. We monitored the study plots once a week for four consecutive weeks in 2006 (June 19, June 26, July 5, and July 13), and then followed up with approximately biweekly monitoring in April and May 2007. We noted a gradient of stress from chlorosis, to shriveled leaves, to no leaves and brittle stem and we recorded the percentage of stems determined to be dead (i.e., no leaves and brittle stems).

Within one week, the 2.5% and 5.0% glyphosate solutions produced on average 29% and 35% stem mortality, respectively, while the 1.25% solution averaged about 10% (Figure 1). Nonetheless, by the second week post-treatment most stems in most plots were dead. And by the fourth, all resprouts treated with glyphosate were dead (Figure 1). In 2007, we did not observe any change in the dead, brittle condition of resprouts and observed no new growth, providing further evidence of low-concentration glyphosate efficacy.

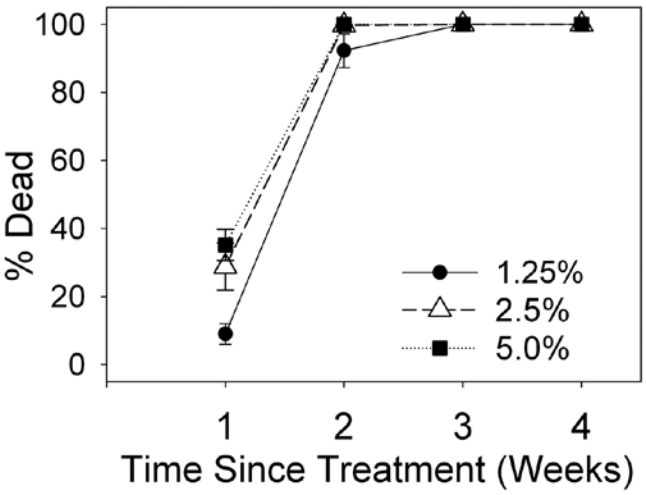

Figure 1. Average (± SE) percentage of dead glossy buckthorn (*Frangula alnus*) resprouts for three concentrations (% a.i.) of glyphosate over a four-week monitoring period in 2006. No dead stems were observed in the control plots.

Successful management of glossy buckthorn and related species likely requires a range of treatment options (Heidorn 1991), with some treatments likely more applicable to a given ecoregion. Although we are not aware of any published work pertaining to the use of glyphosate in the management of glossy buckthorn resprouts, particularly in Upper Michigan, other studies have documented using glyphosate and other chemicals (e.g., triclopyr) or other treatment methods to manage adult plants (Glass 1994, Pergams and Norton 2006), and the efficacy of dormant season herbicide treatments (Reinartz 1997). However, as stated by others (Pergams and Norton 2006, Nagel et al. this issue) eradication of glossy buckthorn and related species likely require follow-up treatments. Our research fills an important information gap by indicating that glyphosate concentrations as low as 1.25% can be effective in killing glossy buckthorn resprouts arising from mature plants previously cut and stump-treated with 20% glyphosate in Upper Michigan.

Acknowledgments

The authors wish to thank the staff and volunteers of Seney National Wildlife Refuge for financial and technical support, especially Tracy Casselman (Refuge Manager) and Laural Tansy (Administrative Technician). Dave Olson (Refuge Biologist) provided helpful edits and comments to early drafts. Field crews were provided as part of a cooperative agreement with the Michigan Department of Natural Resources.

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Interseeding with and without Raking (Illinois)

Stephen Packard (Audubon Chicago Region, 5225 Old Orchard Rd, Skokie, IL 60077, spackard@audubon.org) and Linda Masters (North Branch Restoration Project, Box 2154, Northbrook, IL 60062, masters2001@comcast.net)

In fall 1993 we interseeded six plots in an old field turf with a seed mix of conservative prairie species. Three plots were raked to incorporate the seeds into the soil; three plots were not raked. Ten years later, we counted individual plants of four species that appeared in good numbers in the plots. (A total of nine species appeared to be successfully established, comprising a total of 30.2% of the relative cover of the six plots in 2003. Other conservative species, already present in the area, were not well tested by this experiment.) Two of the four species seemed to benefit from soil disturbance after sowing, and two species did not.

The area chosen for the experiment was former pasture or cropland on Markham silt loam soil in Northbrook, Illinois. (An aerial photo from 1938 shows the site as a mix of pasture and cropland.) The area had subsequently been a Cook County Forest Preserve for many decades. The principal vegetation of the study plots in 1993 (listed in order of total cover in 18 random 0.25-m² quadrats) consisted of tall goldenrod (*Solidago rigida*), early goldenrod (*Solidago juncea*), hawkweed (*Hieracium caespitosum*), meadow fescue (*Festuca elatior*), sedge (*Carex hirsutella*), coneflower (*Ratibida pinnata*), dogwood (*Cornus racemosa*), and bluegrass (*Poa compressa*).

On November 21, 1993, we pounded metal bars into the ground to mark the centers of six circular 6.15 m² plots, selected because they appeared basically similar (a relatively flat surface with few shrubs and generally similar existing vegetation). Each plot was seeded with the mesic prairie turf mix as given in Table 11.1 in Packard and Mutel (1997).Our seed mix consisted of rough seed and chaff (rubbed through screens to break apart the seed heads) mixed about half and half with perlite; we applied this mix at the rate of about one cup of mix per 100 square feet. In three plots (A,C, and E), selected by coin toss after seeding, the seed was raked by hand with four-prong "potato rakes" or "cultivating rakes."

The crew was asked to "rough up" the upper half inch (1.3 cm) of the soil so as to incorporate the seeds. Given the fully vegetated and uneven (bumpy) nature of the soils in these plots (typical in this and many other old field sites in northern Illinois), the raking skipped over some areas, while occasionally raking more than a half an inch deep. Thus, following this treatment, some seeds remain on the surface while others are buried 1 mm to perhaps as much as 20 mm deep. Three plots (B, D and F) were left unraked as a control.

In summer 1994 this area withstood a severe drought. The plots received no special treatment over the subsequent ten years, but were managed as a part of the 36-ha Somme Prairie Grove. This management consisted mostly of spring and fall burning every one to three years, with a hiatus in burning between 1996 and 2001 as a result of county politics. White sweetclover (*Melilotus alba*) and wild parsnip (*Pastinaca sativa*) were pulled by hand from areas like this whenever they appeared.

In August 2003 and May 2004, we relocated the six stakes and remarked the plots. Four species were easily countable in the 6.15 m² plots. Numbers of plants of these four species in the six plots are shown in Table 1. For shooting star (*Dodecatheon meadi*a) and rattlesnake master (*Eryngium yuccifolium*), the total numbers of plants in the raked and control plots were remarkably similar. For purple prairie clover (*Petalostemum purpureum*) and prairie dropseed (*Sporobolus heterolepis*) the totals showed about three times as many plants in the raked compared to the control plots.

Although the variance from plot to plot was great, the apparent pattern was sufficiently suggestive for us to change our management practices. Raking by volunteers is a time-consuming, strenuous, tedious (and unpopular) effort; it also competes for time with work of more proven

Table 1. The numbers of four target species of plants counted, ten years after six plots (A–F) were planted with a mesic prairie turf seed mix.

	Raked Plots				Unraked Plots			
Species	A	C	E	Total	B	D	F	Total
shooting star (*Dodecatheon meadia*)	18	3	52	**73**	10	37	19	**66**
rattlesnake master (*Eryngium yuccifolium*)	14	14	1	**29**	21	4	6	**31**
prairie dropseed (*Sporobolus heterolepis*)	8	9	12	**29**	4	1	6	**11**
purple prairie clover (*Petalostemum purpureum*)	3	4	1	**8**	1	2	0	**3**

importance. Thus, we switched most of our seeding from spring to fall and counted on natural processes to incorporate the seed in the soil. (We continued some seed raking in the case of spring plantings, legumes, rarer seeds, or especially important areas.) As is true for whatever methods we use, we did our best to increase the amount of seed for species that are underrepresented in the restored plots.

All species that we have tested, indeed, nearly all of the region's prairie species, have established well by interseeding in fall or winter without raking. These results have been similar irrespective of soil type, aspect, wetness, etc. This method has not been effective when seed has been planted in dense vegetation (for example, smooth brome, *Bromus inermis*), however, until that vegetation has been opened up sufficiently, for example by two or three years of burning, so that some bare soil is visible in midsummer. Legumes do not do well unless they, and their symbiont bacteria, are already present nearby. We still plant legumes, with their bacterial inoculants, by raking the seed into the turf during the spring.

Many species in the seed mix used were not well testable by this experiment, as some of the species were already present in the area, or because only very small amounts of that species' seed was obtained in 1994. More research under other conditions and in other settings would be valuable.

Acknowledgments

The Forest Preserve District of Cook County acquired the land and supervises management of Somme Prairie Grove Forest Preserve. John and Jane Balaban and many other volunteers of the North Branch Restoration Project contributed expertise and work of many kinds, which were essential to this paper.

Reference

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