Invasive Plant Control by Livestock: From Targeted Eradication to Ecosystem Restoration

G.S. Kleppel, Caroline B. Girard (Biodiversity Conservation & Policy Program, University at Albany, SUNY, Albany, NY 12222, gkleppel@albany.edu, 518/442-4338), Sophia Caggiano, and Erin LaBarge (Department of Biological Sciences, University at Albany)

Targeted grazing (TG), the use of livestock to accomplish specific management objectives, is an effective technique for controlling invasive plants (Launchbaugh et al. 2006 and references therein). Typically, the area targeted for treatment is fenced, and modifications, such as the cutting of dense vines, are made to ensure the effectiveness of the treatment. The animals are deployed for periods of time adequate to severely “damage” the targeted species. Additional treatments, such as herbicide application, may be used as needed after the livestock has been removed.

The effectiveness of TG is illustrated by its use in a cattle pasture invaded by multiflora rose (*Rosa multiflora*) at Glynwood Center, in Cold Spring, New York. Because cattle lack the dentition to browse the thorny rose stems, this invasive has spread rapidly, resulting in the loss of thousands of acres of pasture intended for dairy and beef cattle production. In May 2009, we deployed 13 Boer goats (*Capra hircus*) into an infested (20–30 plants, 1–2 m high, per 10 m²) 0.2-ha portion of pasture at Glynwood Center. The goats remained on pasture until October and readily browsed the multiflora rose. Fecal pellets processed for histological analysis were examined under an inverted microscope, revealing that 42% of the diet of the goats consisted of multiflora rose (Caggiano et al. 2010). Over the course of the season, the vitality (percent of plant with foliated, pliable stems) of multiflora rose in the grazed area declined, and after 2 seasons, many plants were dead (Figure 1). In ungrazed portions of the pasture, multiflora rose appeared healthy. In addition to the invasive species, however, the goats also depleted existing graminoids and forbs and, by September, hay supplementation was needed to support their nutrition. In 2010, the treatment flock was reduced to 6 goats, but hay supplements were necessary by mid-July. We removed the animals in mid-September.

The lesson learned from the project is that successful suppression of the invasive may come at a cost to the larger plant community.

Grazing, like other manipulations of landscapes, disturbs the plant community. However, the disturbance caused by grazing need not disrupt the ecosystem. If the rate of plant biomass removal is lower than the rate of biomass production, overgrazing should not occur. The amount of time animals spend on the landscape is most critical, more so even than the number of animals on that landscape. Wild ungulates tend to pack densely on the landscape but move constantly.

We are using an intensive rotational grazing (IRG) protocol in our studies of the restoration of biologically invaded landscapes with livestock. IRG mimics the distribution of wild ungulates. With IRG, the landscape is divided into paddocks. Animals are stocked at 2–4 times conventional densities (5–9 tons biomass hectare⁻¹) and rotated through the paddock system at high frequencies (2–3 d paddock⁻¹). The grazers open the canopy, and we have observed rapid increases in species richness in grazed paddocks during 30–60 d periods, when grazers are present in any paddock only about 20% of the time (Table 1). We use Romney sheep (*Ovis aries*), a common breed, inexpensively obtained as culls. Before and after the grazing phase of a study, we perform a suite of measurements on...
Table 1. Attributes and results of intensive rotational grazing studies in 2 invaded landscapes in the upper Hudson River Valley of eastern New York. The names of the dominant invasive species are in parentheses. For each study, t is the approximate number of days that sheep were present in any single paddock and (T) is the total period of study (days). Mean (± standard error) canopy height in grazed and ungrazed paddocks are given before and after grazing.

<table>
<thead>
<tr>
<th>Landscape (Invasive species)</th>
<th>Canopy Height (cm)</th>
<th>Species Richness</th>
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<tbody>
<tr>
<td></td>
<td>Ungrazed</td>
<td>Grazed</td>
</tr>
<tr>
<td>Year</td>
<td>(T)</td>
<td></td>
</tr>
<tr>
<td>Wet Meadow (Lythrum salicaria, Phalaris arundinacea)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2008</td>
<td>12(59)</td>
<td>Before</td>
</tr>
<tr>
<td></td>
<td></td>
<td>After</td>
</tr>
<tr>
<td>Riparian (Phragmites australis)</td>
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<td></td>
</tr>
<tr>
<td>2010</td>
<td>6(31)</td>
<td>Before</td>
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<tr>
<td></td>
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<td>After</td>
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*%ΔS = 100% (S_g - S_u) / S_u, where S_g and S_u = species richness in grazed and ungrazed paddocks, respectively.
nd: Species richness data were not collected because the plant community was not sufficiently developed to permit identification of many species.

individual plants of the target invasive species and on the plant community. Measurements germane to the present discussion are canopy height, measured with a meter stick or tape, and species richness measured in 10–25 randomly selected 0.25 m² quadrats in each paddock.

Table 1 summarizes our observations on the use of IRG in 2 invaded landscapes in New York’s Hudson Valley. The first is a wet meadow in an agricultural landscape infested with purple loosestrife (Lythrum salicaria) and reed canary grass (Phalaris arundinacea). These species grow tall, creating dense monotypic stands and considerable shade. During the 2008 and 2009 growing seasons, we studied the use of IRG as a restorative protocol with a paired, 4-paddock rotational system. One paddock in each pair was grazed, the other was not. Sheep were moved from 1 paddock to the next at ~3 d intervals, an entire rotational cycle requiring about 12 d. However, we removed the animals from the site on hot, humid days and during severe storms. Details on the grazing protocol and a discussion of results obtained during the 2008 season are provided by Kleppel and LaBarge (2011).

Prior to the onset of grazing during the 2008 season, the difference between mean canopy heights in paddocks that would not be grazed (hereafter called “ungrazed”) and paddocks that would be grazed (hereafter called “grazed”) was not significant (ANOVA; F = 1.415; df = 159; p > 0.05). After the grazing phase of the study, the mean canopy-height difference was significant (F = 74.3; df = 159; p < 0.001) and species richness, S, was 25% higher in grazed than ungrazed paddocks.

Similar results were obtained in 2009. Prior to grazing, mean canopy heights in ungrazed and grazed paddocks were not different (F = 0.011; df = 119; p > 0.05). After grazing, the mean height difference in grazed and ungrazed paddocks was significant (F = 8.19; df = 119; p < 0.01). Although S increased by only 1 species in the grazed paddocks, it declined by 20% (from 20 in 2008 to 16) in 2009, in the ungrazed paddocks. The difference between species richness, %ΔS, in grazed and ungrazed paddocks was 62.5%.

The second landscape studied was a small (0.14 ha) riparian zone near Saratoga Springs, New York, heavily infested with common reed (Phragmites australis). We followed the grazing protocol of Kleppel and LaBarge (2011), but the small area of infestation limited the number of paddocks in this study to 2 grazed and 1 ungrazed. A third paddock, in an area that was not invaded by common reed was set up to keep the sheep off the invaded site long enough to allow the grazed paddocks adequate time to “rest.” The sheep were moved among the 3 paddocks at ~3 day intervals.

Prior to sheep deployment, the differences in mean height of common reed in the ungrazed and grazed paddocks was not significant (ANOVA; F = 0.8; df = 59; p > 0.05) (Table 1). After grazing, the mean canopy height difference was significant (F = 12.72; df = 59; p < 0.05). Over the course of the study, S declined from 17 to 15 species in the ungrazed paddock, but increased from 17 to 23 species in the grazed paddocks, a difference of 53.3%.

Although elevated species richness associated with grazing is not a new observation, the rates at which S increased, are noteworthy (Table 1). Grazing can stimulate plant growth by opening the canopy (increasing insolation), aerating the soil, and mobilizing soil nutrients and microbes (Frank 2008). Such processes operate at physiological time scales, and opportunistic vascular plant responses, including seed germination and rapid mid-season growth rates, would be expected. Studies are underway to explore some of these processes further.

Connell (1978) observed that ecosystems exposed to moderate levels of disturbance tend to be more diverse than those exposed to either weak or intense disturbances. Grazing represents a disturbance to the plant community. When foraging is light, as when a few deer forage in a meadow, the intensity of disturbance is weak. Conversely,
in conventional agricultural systems, with livestock present on the landscape for weeks to months, the disturbance caused by grazing can be intense, even disruptive, as our Boer goat study illustrated (Figure 1). Neither scenario would be expected, according to Connell’s intermediate disturbance hypothesis, to support high S. If, however, IRG, which applies intense grazing pressure over brief periods of time, creates a disturbance intermediate between light and conventional grazing, then relatively higher S would be expected in the IRG-grazed landscape. The intermediate disturbance hypothesis may help resolve inconsistencies in the observed impacts of grazing reported in the literature (see Kleppel and LaBarge 2011), and further consideration of this hypothesis would seem worthwhile.

Acknowledgments
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References
Caggiano, S., A.O’Connor and G.S. Kleppel. 2010. Using goats as to control a population of 

Tipped Over Duck Nest Box Traps Turtles in a Restored Wetland (Ohio)
Denis Conover (Dept of Biological Sciences ML 0006, University of Cincinnati, Cincinnati, OH 45221-0006, denis.conover@uc.edu), Wayne R. Wauligman DDS (2170 Quail Run Farm Ln, Cincinnati, OH, 45233, 513/922-4430, wrwpgw@aol.com) and Karen Cody (Fernald Nature Preserve, 7400 Willey Rd, Harrison, OH 45303, 513/648-4899, brightcody@aol.com)

Duck nest boxes are often erected in both public and privately owned wetlands to provide nest sites for cavity-nesting ducks, such as wood ducks (*Aix sponsa*) and hooded mergansers (*Lophodytes cucullatus*). There may now be millions of these boxes erected in wetlands in the United States and Canada, since many state and federal agencies and nonprofit organizations have disseminated plans for building nest boxes and funded programs to distribute them for more than 30 years. These boxes are beneficial to ducks and have helped wood duck populations

Figure 1. A downed duck nest box in a dried up pool at Shaker Trace Wetlands (above) that contained many turtles. In 2007, the box was still moist inside and contained several dead and 1 live midland painted turtles (*Chrysemys picta marginata*), 2 dead box turtles (*Terrapene carolina carolina*), and 1 live common snapping turtle (*Chelydra serpentina*) (below). Photos by Denis Conover (left) and Karen Cody (right).
to increase, but they must be properly monitored and maintained. This is usually on an annual basis, typically during the winter or just before the breeding season, but should probably be done more frequently because we have discovered that if a pole gets tipped over and the box gets into the water, these duck nest boxes can serve as death traps for turtles. Our goal is to reduce suffering and death of turtles by warning land managers about the threat to turtles that downed duck nest boxes can pose.

In August 2007, a downed duck nest box was found in a dried up pool of an ephemeral lake in the Shaker Trace Wetlands, a restored wetland in southwestern Ohio (Conover and Klein 2010). In the spring and early summer, this particular lake is typically 1–1.5 m deep, but by late summer even the deeper depressions may be completely dry. When first discovered, the box was still moist inside and contained many dead turtles, which were then removed (Figure 1). While the 3 species of trapped turtles at Shaker Trace are not endangered in Ohio, other wetlands such as the Beaver Creek Wetlands, Spring Valley Wildlife Area, and Cedar Bog harbor rare species such as the spotted turtle (Clemmys guttata).

Apparently the turtles crawled or swam into the box when it was submerged in the pool and couldn’t find their way back out. Some baited traps designed to live-capture turtles work on this principle. It is also possible that the turtles entered the box when water levels were higher than the hole and became trapped when water levels in the pool receded below the hole. If this is the case, the risk to turtles would be greater in ephemeral wetlands. The cause of death could not be determined, but possibilities include drowning, overheating, starvation, or possibly even bites from the snapping turtle (Chelydra serpentina) that was in the box (Figure 1). To prevent more turtles from becoming trapped, we propped up the box and left it open (Figure 2) until the land manager could properly re-erect it. We recommend that nest boxes be monitored throughout the year so that they can be repaired and reinstalled promptly as needed.

Acknowledgments
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References

Keystone Role of Beavers in a Restored Wetland (Ohio)

Denis Conover (Dept of Biological Sciences ML 0006, University of Cincinnati, Cincinnati OH 45221-0006, 513/556-0716, denis.conover@uc.edu)

The Shaker Trace Wetlands is a restored wetland area in southwestern Ohio (Klein 1992). Several years ago, a small number of American beavers (Castor canadensis) took up residence in an ephemeral lake in the wetlands. These beavers cut down willows (Salix spp.)—thus allowing more light to reach other vegetation—built a lodge, and dug an extensive system of canals (Figure 1). There is no stream flowing into the lake, so no dam. In some places, this lake can be over 1 m deep during the spring and early summer, but by late summer it may dry up completely owing to normal dry conditions. During the driest periods the canals dug by beavers provided an important reservoir of water for amphibians, fingernail clams, snails, and other aquatic organisms (Figure 1). During canal excavation, the beavers churned the soil and exposed seeds from deeper in the soil seed bank. In addition to these ecological benefits, the beavers provided wildlife viewing opportunities for visitors to the wetlands. After a couple of years in this ephemeral lake, the beavers abandoned their lodge and moved on, probably to a river or pond less likely to dry up.

Within a couple of years of the beavers’ departure, the extensive growth of willows and other woody vegetation made mechanical removal with a Hydro-Axe machine necessary (Conover and Klein 2010). This operation opened extensive areas to more light, and the large tires of the Hydro-Axe machine left deep tracks in some areas. Churning the soil exposed seeds from deeper in the soil seed bank. The Hydro-Axe operation served to increase plant diversity in the wetlands and provided more habitats for herbaceous species. The tire track depressions hold water during the summer drought period longer than surrounding areas, giving larval amphibians more time to develop into adults.
Snails, fingernail clams, and other aquatic species also benefit from water held in these depressions. In other words, the Hydro-Axe operation provided services at a cost of about $5,000 for 20 ha—services that had been provided for free by the beaver colony before they moved on.

Beavers were extirpated from Ohio by 1830 (Chapman 1949) but have been making a comeback during the last few decades. In some parts of the country, beavers have been reintroduced and have provided rapid improvements in hydrology, riparian vegetation, and wildlife habitat (e.g., Albert and Trimble 2000). In restored ephemeral wetlands similar to the Shaker Trace Wetlands in Ohio, beavers can play a keystone role by cutting down woody vegetation and digging canals that hold water for longer periods into the dry season, and by churning the soil, exposing seeds buried deeper in the soil seed bank. Beavers could be encouraged to remain in such wetlands by scooping out some deeper pools. This would benefit aquatic organisms by holding more water longer into the drought period, and it might enable beavers to remain in the wetlands. Other parts of the wetlands could still be allowed to dry up during drought periods, discouraging the establishment of fish that could prey on amphibian larvae. Desirable larger trees in the wetland area can be protected from beavers by wrapping aluminum, chicken wire, or steel screen around the trunks up to a height of 80 cm (Albert and Trimble 2000).

References

Shifting Baseline Syndrome as a Barrier to Ecological Restoration in the American Southwest
Tong Wu (Northern Arizona University, School of Forestry, Ecological Restoration Institute, PO Box 15017, Flagstaff, AZ 86011, 928/523-7182, tong.wu@nau.edu), Michael Anthony Petriello (Northern Arizona University, School of Forestry) and Yeon-Su Kim (Northern Arizona University, School of Forestry)

More than a century of human encroachment on an industrial scale has eroded the resilience of forests across the southwestern United States, resulting in ecosystems that depart profoundly from historical conditions (e.g., Murphy et al. 2007). Regional ponderosa pine (Pinus ponderosa) forests, since at least the mid-20th century, now experience high-severity crown fires that are inimical to both ecological health and the socioeconomic security of local communities (Table 1). The situation is exacerbated by climate change, which has increased the intensity of and area burned by fires (Westerling et al. 2006). This has galvanized an expansion of ecological restoration treatments—mechanical thinning and prescribed burning aimed at reducing forest density and ground litter. These measures significantly decrease the intensity of wildfires and help re-create the structure, composition, and functions of a healthy forest system (Fulé 2008). However, despite demonstrated successes and cogent scientific reasoning, a barrier to effective landscape-level implementation exists (Hjerpe et al. 2009).
We believe that many of the factors contributing to this barrier are consistent with shifting baseline syndrome (SBS), where a discrepancy between social perceptions and ecological realities contributes to ineffective management as human communities fail to recognize change, considering recent ecological conditions to be normal and thus perceiving them as baseline (Papworth et al. 2009). Despite broadening application of the SBS concept, the implications for restoration are scarcely noted in the existing literature; when SBS is mentioned in the context of restoration, the references we have found explore the association accurately but fleetingly (e.g., Pauly 1995). The analysis presented in this paper is, to the best of our awareness, the first study to examine the likely occurrence of SBS in a restoration scenario.

A common obstacle in diagnosing SBS is the lack of empirical data on altered ecological conditions (Papworth et al. 2009). In this case, the ecological history of southwestern ponderosa pine forests is well understood, and abundant research indicates that the ponderosa pine ecosystems of today are vastly different from those of earlier generations. Additionally, a key tenet of SBS is that it is a social phenomenon (Papworth et al. 2009), and the weight of current scientific and policy-relevant research suggests that existing discrepancies between social perceptions and ecological conditions are due to a failure of the former to keep pace with changes in the latter (e.g., Ostergren et al. 2008).

We believe “generational amnesia” is the form of SBS that describes the socioecological dynamics of forest restoration in southwestern ponderosa pine forests. Generational amnesia refers to a loss of or change in ecological knowledge when earlier generations fail to pass on their experiences to later generations, thus altering perceptions of ecological conditions and interpretations of past anthropogenic modifications (Pauly 1995, Papworth et al. 2009). As a result, the baseline used to interpret current ecological conditions is closer to a single human lifetime than to longer, multigenerational time frames. From an extensive survey-based study of social attitudes toward forest restoration in north-central Arizona, Ostergren and others (2008) found that rural residents—those living closer to forests—were less likely to support restoration treatments such as tree removal. The authors infer that this is because rural residents “are accustomed to relatively dense forest stands and thus may have difficulty perceiving a heavily thinned forest as a ‘healthy’ forest” (Ostergren et al. 2008, 57). The contemporary misunderstandings of the meaning and values of forest restoration are, in many ways, to be expected (Hjerpe et al. 2009). The aim of restoring regional forests to resemble an ecosystem that hasn’t existed for decades, if not hundreds of years, is to make a difficult perceptual demand on residents who don’t benefit from any continuity of local ecological knowledge.

When the forest ecosystem more closely resembled the target conditions of contemporary restoration efforts, the primary tenants of the land were Native American communities (and, to a lesser extent, early Hispanic settlers). They had been, however, displaced to reservations and other marginal lands or otherwise disenfranchised by the widespread expansion of logging, industry, and other enterprises. This disruption in the intergenerational transfer of local ecological knowledge was especially pronounced as a result of the adoption of an efficiency-oriented European-style forest management approach. As Langston (1996, 250) writes, in the late 19th and early 20th centuries, the U.S. Forest Service believed that “if light burning was an Indian practice, then by definition it was superstition, not science.” Surface fires were rejected for being incompatible with the prevailing forestry paradigm, which centered on a belief in the superiority of technocratic expertise (Murphy et al. 2007).

Forest restoration is a complex process touching upon multiple social and ecological dimensions. Although SBS may not be the chief obstacle to expanding restoration treatments, we believe it plays a significant role in explaining the factors shaping public acceptance of forest management policy. In Arizona and states where forest restoration remains an urgent issue, additional surveys of local residents’ perceptions of current forest conditions and their awareness of regional ecological history are needed for a more definitive diagnosis of SBS. Strategies for promoting landscape-level ponderosa pine restoration must confront this issue.

Local stakeholders are likely to lend broader support for restoration treatments if they recognize, for instance, that before the implementation of fire suppression policies generations ago, surface fires (both natural and manmade)
were integral in maintaining the historical structure of the forest. Encouragingly, there is evidence that environmental education efforts by nongovernmental organizations and the Forest Service have been successful in aligning public perceptions and expectations with the objectives of forest restoration (e.g., Ostergren et al. 2008). Engaging stakeholders and educating them on the characteristics of healthy forests and the goals of restoration are the most promising avenues to overcoming generational amnesia presently endemic in the region. Additionally, drawing upon untapped sources of ecological knowledge within the community can also inform management decisions. For instance, an oral history project chronicling Native American institutions may help build cross-generational bridges to traditional practices, such as controlled burning, and shed light on dormant knowledge about regional ecosystems. Aligning public understanding with the reality of fundamentally altered biological conditions is a necessary (though perhaps insufficient) condition for achieving sustainable forest management.

We expect that SBS will become a relevant issue not only for restoration in the American Southwest, but also for many other sites across the world. In nearly all contexts, ecological restoration involves community stakeholders. We recommend survey-based assessments of public understanding of the means, objectives, and context of ecological restoration so that SBS is a key component of the social science research agenda. In scenarios where SBS is a likely problem, rigorous documentation of where social perceptions and ecological realities fundamentally differ, and the reasons and the causes for these discrepancies, can help practitioners gain greater support among community stakeholders.

References


Promising Results Restoring Grassland Disturbances with Native Hay (Alberta)

Peggy Ann Desserud (Dept of Renewable Resources, 751 General Services Bldg, University of Alberta, Edmonton AB, T6G 2H1, desserud@ualberta.ca) and M. Anne Naeth (Dept of Renewable Resources, 855 C General Services Bldg, University of Alberta, Edmonton AB, T6G 2H1, anne.naeth@ualberta.ca)

In Alberta, much of the once dominant rough fescue grassland has been lost to cultivation, overgrazing, and intensive oil and gas activities. Few attempts to restore rough fescue plant communities have been successful (Elsinger 2009, Desserud et al. 2010). Plains rough fescue (Festuca hallii) is a perennial bunch grass, slow growing and long lived, requiring 2 to 3 years to become established from seed. Rough fescue is an erratic seed-setter, seldom producing seed (Johnston and MacDonald 1967). The objective of our study was to assess the potential of native hay as a seed source for restoring rough fescue grassland.

The benefits of native hay include no cost for seeds, a natural mix of adapted native species, protective mulch for emerging seedlings, no requirement for special seed processing or seedling, and increased ground cover. However, the relative hardness of prairie grasses requires specialized harvesting equipment, and seed viability is unreliable. The highly variable production of seed set and the resulting dominance of species in seed at time of harvest influence seed viability in native hay (Romó and Lawrence 1990).

We found no previous research involving native hay for rough fescue grassland restoration. Experiments using native hay to restore grasslands were successful in Germany (Kiehl et al. 2006), England (Jones et al. 1995, Edwards et al. 2007), and Idaho (Gates 1962). In Idaho, native hay resulted in successful native grass establishment, while fertilizer and seeding with sawdust and conifer mulches had poor results (Gates 1962). In contrast, no native seedlings emerged from native hay application in mixed-grass prairie restoration in Saskatchewan (Wilson et al. 2004).

The study area is located in Alberta, Canada, in uncultivated rangeland in the Central Parkland natural region. Topography is an undulating complex of small depressions and hills. The soils are Dark Brown Chernozems on loam...
### Table 1. Mean (± SD) cover (%) for selected plant species on the native hay and seeded pipeline ROWs, showing year of growth, adjacent native grassland control, and initial germination (%) from soil seed bank and native hay. Differing letters indicate significant differences within a ROW at \( P < 0.05 \). Shading indicates non-native or weedy species.

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<thead>
<tr>
<th></th>
<th>Native Hay ROW</th>
<th>Seeded ROW</th>
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<tbody>
<tr>
<td></td>
<td>% Cover</td>
<td>% Germ</td>
</tr>
<tr>
<td>Grasses</td>
<td></td>
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<tr>
<td>Annual rye</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Bluegrasses</td>
<td>4.5 (6)a</td>
<td>40(13)b</td>
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<tr>
<td>June grass</td>
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<td>0</td>
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<tr>
<td>Northern wheatgrass</td>
<td>0</td>
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</tr>
<tr>
<td>Plains rough fescue</td>
<td>10 (11)</td>
<td>12(23)</td>
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<tr>
<td>Rocky mountain fescue</td>
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<td>0</td>
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<tr>
<td>Slender wheatgrass</td>
<td>1.3 (3)a</td>
<td>2.0(6)a</td>
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<tr>
<td>Western porcupine grass</td>
<td>5.5 (9)a</td>
<td>0b</td>
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<td>Forbs</td>
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<td>Lamb's quarters</td>
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<td>3.5(9)</td>
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<td>9.7(2)</td>
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<tr>
<td>Shrubbs</td>
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<tr>
<td>Prairie rose</td>
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<tr>
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<td>Total Cover</td>
<td>47.9a</td>
<td>81.4b</td>
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<td>Bare ground</td>
<td>0a</td>
<td>10(13)b</td>
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<tr>
<td>Litter</td>
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In August 2005, an energy company removed topsoil from the native hay ROW (15 × 150 m) before pipeline installation, which they spread back within 1 month after construction, and left the ROW unseeded. They cut hay in adjacent grassland on July 16, 2006, after plains rough fescue peak flowering in central Alberta in June and before midsummer seed shattering. A modified combine, with more durable and sharper than traditional crop blades, was used to cut about 67 m³ of hay in grassland approximately 50 to 200 m from the pipeline and immediately spray it on the ROW to a depth of 2 to 3 cm.

In July 2007, a different energy company removed topsoil from the seeded ROW (3 × 150 m) before pipeline installation and spread it back after construction. In August 2007, they seeded the ROW (approximately 15 kg/ha) with annual rye (Elymus sp.), slender wheatgrass (Elymus trachycaulus), and Rocky Mountain fescue (Festuca saximontana).

We collected monthly precipitation data at a well site (Byemoor) about 30 km east of each pipeline between April 2007 and August 2009, which we averaged with Environment Canada data from weather stations 35 km south (Craigmyle), 25 km northwest (Big Valley), and 30 km west (Trochu) of the pipelines, forming a circle around the pipeline areas.

To evaluate native hay seed content, we randomly collected 10 hay samples and spread each approximately 1 cm thick over 3 cm of potting soil (1:4 vermiculite and peat) in trays (10 × 15 × 5 cm). To assess seed bank potential of the seeded ROW, we collected 10 soil samples (15 × 15 × 6 cm) from the newly reclaimed ROW, which we spread approximately 2 cm thick over potting soil in trays (10 × 15 × 5 cm). When the surface began to dry, we watered all trays with tap water, approximately every 2 days. We enumerated emerging seedlings and removed them once identified for a 3-month period.

We sampled the native hay ROW in 2007 and 2008, and the seeded ROW in 2008 and 2009, in July, when...
the majority of grass species were mature. With 2 50-m transects, randomly located and each containing 5 subplots (20 × 50 cm) spaced 10 m apart, along the native hay and seeded ROWs, we assessed foliar cover of all species, litter, and bare ground. During the first year for each ROW, we sampled vegetation, litter, and bare ground in adjacent native grassland, 15 m from the ROW, to serve as an undisturbed control.

We subjected data to one-way ANOVA with Tukey’s post hoc test and independent sample t-tests for pairwise comparisons at 1% level of significance (p < 0.05) using PASW (vers. 18.0, SPSS, Chicago IL) and Excel (vers. 2007, Microsoft, Redmond WA). We used nonparametric multiple response permutation procedure (MRPP), operating on Sorenson (Bray-Curtis) distance measures, to evaluate significant differences between seeded plots and controls using PC-ORD (vers. 5.31, MjM Software, Gleneden Beach OR). The MRPP generates a chance-corrected within-group agreement value (A), which evaluates the difference between species composition of grouped plots. The lower the A value, the more similar are the groups (McCune and Grace 2002).

The adjacent native grassland at both sites was dominated by plains rough fescue, shortbristle needle and thread (*Hesperostipa curtiseta*), prairie Junegrass (*Koeleria macrantha*), western wheatgrass (*Pascopyrum smithii*), slender wheatgrass, bluegrasses (*Poa* spp.), sedges (*Carex* spp.), and an abundance of forbs, such as northern bedstraw (*Galium boreale*) and yarrow (*Achillea millefolium*). The dominant grass species in the native hay were slender wheatgrass, plains rough fescue, bluegrasses, and western wheatgrass. The seed bank from the seeded ROW included plains rough fescue and bluegrasses (Table 1).

In the first year, western wheatgrass had the greatest cover on the native hay ROW, followed by plains rough fescue, shortbristle needle and thread, and bluegrasses (Table 1). Rough fescue plants were seedlings approximately 3 cm in height. First year’s growth on the seeded ROW was dominated by seeded slender wheatgrass and several weeds (Table 1). Bare ground averaged 10% on the native hay ROW and 30% on the seeded ROW, and neither had litter.

Cover of slender wheatgrass (p = 0.207), total forbs (p = 0.833), native species (p = 0.198), and litter (p = 0.283) was similar in both ROWs the second year. The native hay ROW had greater cover of western wheatgrass (p = 0.018) and bluegrasses (p < 0.001). Less bare ground occurred on the native hay ROW, although the difference was not significant (p = 0.234). The seeded ROW had no
rough fescue, while the native hay ROW had 12% cover (Figure 1).

Comparing the native hay ROW second year growth to the adjacent grassland showed similarities in rough fescue ($p = 0.011$), slender wheatgrass ($p = 0.032$), western wheatgrass ($p = 0.043$), and bluegrass ($p = 0.047$) cover. The native hay ROW had fewer forbs ($p < 0.001$) and native species ($p < 0.001$), and more litter ($p = 0.001$). Total vegetation cover was 68% of the control. In contrast, the seeded ROW had less rough fescue cover ($p < 0.001$), more slender wheatgrass ($p = 0.040$), and greater bare ground ($p = 0.024$), as well as fewer native species ($p < 0.001$) and more litter ($p = 0.002$) than the adjacent grassland in the second year. No differences were found in bluegrass ($p = 0.056$), western wheatgrass ($p = 0.668$), and total forb ($p = 0.423$) cover. Total vegetative cover was 20% of the control. The MRPP analyses showed the native hay ROW was more similar to controls ($A = 0.198, p < 0.001$) than the seeded ROW ($A = 0.383, p < 0.001$).

Monthly precipitation varied over the years of the study. In 2006, the year that the native hay treatment occurred, the accumulated precipitation before treatment (April–June) was 250 mm, followed by 27 mm in July during the haying and first-year sampling, and 45 mm in August. In 2007, the accumulated precipitation from April to June was 148 mm, followed by 37 mm in July (when the second year growth in the native hay ROW was sampled) and 46 mm in August (when the seed treatment occurred). In 2008, the accumulated precipitation was 30 mm, followed by 53 mm in July (when the second year growth of the seeded ROW was sampled). While 2006 was a wetter season than 2007, both native hay and seeded ROWs experienced similar precipitation levels in August immediately following treatment.

This experiment supports the hypothesis that native hay cut from rough fescue grassland is a viable seed source for restoring disturbances. All species that emerged on the native hay ROW were found in undisturbed grassland. Our results were consistent with those from European (Jones et al. 1995, Kiehl et al. 2006, Edwards et al. 2007) and American (Gates 1962) grassland restoration experiments.

Of particular note in our experiment was the emergence of rough fescue seedlings in the first year, and their continued growth over the following year. This is a promising result given the failure of rough fescue establishment, even when seeded, on other oil and gas disturbances in the area (Elsinger 2009, Desserud et al. 2010). As expected, the seeded ROW was dominated by seeded species in the first and second year. Despite the occurrence of rough fescue in the seed bank of the seeded ROW, only a small amount of rough fescue appeared in the first year, possibly remnant plants from the initial topsoil stripping, and none appeared in the second year. Cover on the native hay ROW met provincial reclamation regulatory requirements, unusual after only 2 years of growth. It included cover exceeding 65% of control, similar species, and no non-native species. The seeded ROW, while admittedly affected by a dry growing season, did not meet criteria, having low total cover and few species similar to the control.

Applied hay would have increased ground cover, which likely accounted for the reduction of weedy species on the ROW, similar to what occurred in the experiment by Jones and others (1995). The limited amount of bare ground, commencing in the first year, is in direct contrast to the seeded ROW and what Elsinger (2009) and Desserud and colleagues (2010) found on seeded ROW even 30 years after recovery. While precipitation before seeding was greater at the native hay site than the seeded site, precipitation during and after the seeding month was similar, suggesting first year germination may have been comparable. The second-year growing season of the seeded ROW was very dry, probably accounting for lower vegetation cover; nevertheless, species composition of perennial grasses should not have been affected (Holmes and Rice 1996).

Seasonal timing of hay cutting is important in determining which seeds will be available and viable. Since our experiment targeted rough fescue, the hay was cut when its seeds were mature. To obtain a full suite of native grassland species, Edwards and others (2007) recommended cutting hay several times, such as in early, middle, and late summer. Kiehl and colleagues (2006) had success baling hay from a donor site and transporting it; however, further research into the longevity of native hay bales is needed. Being able to store native hay for future use would be important for well-site restoration, which may take place several years after construction, or for retaining species, such as rough fescue, that do not produce seeds every year.

This experiment showed that native hay has potential to provide early species establishment and a diverse plant cover similar to predisturbance grassland conditions. Since only 1 native hay site was available for study, extrapolation of the results to other sites is not strong. Nevertheless results are promising and warrant further study to evaluate timing of hay harvesting, how native hay responds to storage, and optimal coverage.

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References
The genetic diversity and structure to facilitate adaptation to climate change and other environmental perturbations (e.g., Johnson et al. 2010). Commercial seed mixtures of non-native species and genetically uniform varieties threaten local diversity. Consequently, efforts to develop native seed sources are receiving considerable attention.

During the 7th Society for Ecological Restoration European Conference in Avignon, a special session focused on the successes and challenges of producing and using native plant material on a regional scale. European and American participants highlighted common issues encountered in developing native seed supplies (Figure 1), creating new market niches, and adapting seed certification procedures for use with native materials with the goal of sharing effective solutions and devising new approaches. Here we share the key findings and next steps outlined in this special session.

Several biological and technical challenges hinder the development of native plant programs at local or regional scales, such as: 1) identifying species-specific seed zones derived from ecological studies and provisional seed zones based on climatic and environmental variables (Johnson et al. 2010); 2) developing genetically diverse, ecologically adapted materials (Johnson et al. 2010); 3) formulating strategies to track plant materials from wildland harvest through agricultural production as well as to manage stock seed or other types of plant materials (Figure 2); 4) developing seed technology for diverse woody and herbaceous species; 5) understanding pollinator requirements and potentially managing wild pollinators in seed fields; 6) identifying cultural practices, including pest and disease control, for maximizing seed production; and 7) developing effective strategies and equipment for reestablishing native plant communities (USDI BLM 2009).

Major political and economical obstacles include sustaining funding for research and development, creating new market niches for seed growers, and creating and maintaining collaboration among researchers, seed regulatory agencies, the private seed industry, and private and public end users.

One of the main topics in the session was the limited European production of native plant material owing to high costs and lack of propagation experience. Native seed production is often organized by local nongovernmental organizations (NGOs) or very small companies, and seed quantities and range of species are limited. Moreover, the lack of administrative support for native plant material leads to widespread use of low-cost commercial seed mixtures containing horticultural and agricultural cultivars and wildflower seeds of unknown or nonlocal origin. Use of easily propagated and widespread cultivars ensures the continuous availability and affordability of these mixtures but ignores the importance of local genotypes. Conrad (2007) and Tischew and others (2010) evaluated grassland restorations to counteract impacts of infrastructural projects on natural systems in Germany. Approximately
70% of the projects used standardized commercial seed mixtures. However, using these mixtures did not help to reach the species-richness objective. Deficits in grassland restoration projects owing to the use of non-native seed mixtures have been well documented (Kiehl et al. 2010). In the future, restoration, as well as revegetation of grasslands outside cities or agricultural areas, should be accomplished only with seeds of local origin produced regionally or harvested from local species-rich near-natural grasslands. To facilitate this objective, legal support must be secured. The German Law for Nature Conservation demands that only species of local provenance will be used after 2020 in restoration and revegetation projects, thus confining the use of nonlocal ecotypes and cultivars to cities, agriculture, and forestry.

In recent years, the European Union (EU) has supported several major projects to expand propagation and use of native plants in ecological restoration. One example is the project “Seminatural Grassland as a Source for Biodiversity Improvement” (www.salvereproject.eu), a collaboration of 8 partners from 6 EU countries. Main objectives are the development of best practice methods for harvesting seed mixtures from species-rich seminatural grasslands and optimization of near-natural revegetation methods for restoration or establishment of species-rich grasslands.

For those few European countries where wild seed is marketed, availability and demand are both low. Most seed is not certified; consequently, the danger of a grey market without proof of indigenous origin is growing. It is crucial to define transnational seed zones and quality assurance standards for collecting, producing, selling, and using native seeds, thus requiring a certification system with control mechanisms and the power to impose sanctions. At the moment, only Germany and Austria have certification systems for native seeds.

Unfortunately, with the new EU Commission Directive 2010/60/EU, trade with regional ecotypes will become increasingly difficult in Europe because the directive states that only 5% of the traded cultivars can be sold as “wild” seeds to avoid competing with commercially-produced cultivars. This restriction would be controlled by EU member states, resulting in a major administrative effort. Moreover, the new directive allows the propagation of wild seeds outside their region of origin. If seeds are propagated at considerable distances from their source, cross-pollination with nonlocal ecotypes and adaptation to climatic conditions at the propagation site (rather than the restoration site) are likely. In addition, propagation in foreign countries hampers effective certification, and the unintended import of invasive non-native species cannot be prevented completely. Finally, annual variations in harvests from species-rich native grasslands preclude compliance with standardized reporting and yield regulations, thus hampering local restoration efforts.

Participants from the United States focused on the federal Native Plant Materials Development Program (NPMDP), which was created in 2001 to provide plant materials for restoration on federal public lands (USDI 2002, USDA 2002). Primary goals are to supply and manage native plant materials through research and collaboration with the private-sector native seed industry.

As an example, the United States Bureau of Land Management (BLM) NPMDP supports programs primarily on arid and semiarid lands of the interior western United States where BLM lands are concentrated (USDI BLM 2009). Seven regional programs focus on long-term research and development of plant materials for specific biogeographic areas. The Great Basin Native Plant Selection and Increase Project, one of the earliest regional programs, focuses on research required to develop seed zones, plant materials appropriate for these zones, cultural practices for agricultural seed production, and wildland seeding strategies.

Despite the considerable progress made in the United States over the last decade, several challenges persist. Acceptance of native species and genetically diverse, regionally appropriate plant materials by federal agencies continues...
to increase but is hindered. This is in part owing to a lack of experience in using these materials and a history of seeding commercially available cultivars of non-native and native species. Seeds of many native grasses and most native herbs cannot be collected from wildland stands in adequate quantities for immediate use, but must often be produced in agricultural fields because of extensive non-native species invasions and plant community fragmentation. This necessitates development of species-specific cultural practices for seed production and delays the availability of larger quantities of seed. Seed growers are often reluctant to take on new species because of production and marketing uncertainties, which occur because seed purchases are driven by fluctuating budgets and in the Intermountain West by unpredictable seed demands for postfire seedings. The location and total area burned annually vary widely, making it difficult to predict the market for specific regional materials from 1 year to the next. Efforts to overcome these problems include increased seed warehousing, contracting for production of materials with narrower distributions, and encouraging development and use of regional materials by multiple users. A national system for certification of wildland-collected and agriculturally produced native seed is in place and continues to undergo modification as needs for native plant materials increase and approaches for their development and maintenance evolve. Although uncertified seed continues to be sold, certification is increasingly used for marketing of native species, providing greater assurance of seed origin and providing practitioners with a tool for improving their ability to purchase seed that will be adapted to the planting site.

Member states of the EU can learn from the United States how the use of wild seed can be embedded into governmental schemes. In Europe, the demand for local ecotypes for restoration and revegetation is currently from idealistic scientists and practitioners or NGOs. Restoration
is small scale and in most cases funding comes from scientific or applied research projects. Hence, restoration goals are often highly ambitious with regard to species-rich site-specific plant community assemblies. However, strategies for large-scale restoration projects are needed. In Europe and in the United States there is a need to evaluate and share effective techniques for reestablishing native vegetation in diverse ecosystems. Use of native hay, for example, has received little use in the United States and could be tested in appropriate areas. In both Europe and the United States, certification of native seed and plants across biogeographic regions, developing new market niches for growers, and providing increased stability in demand are all critical issues to increasing the availability and expanding the use of native materials. Tools to aid in selection of appropriate plant materials for restoration in light of climate change, issues of ex situ and in situ conservation of species and communities and discussions surrounding assisted migration are all critical to the future of ecological restoration on both continents.

In the future, the EU and the United States need to share local, regional, national, and international approaches and policies regarding the development and use of native plants at all levels. In addition, we must improve communication with the public, growers, and users to improve our understanding of native plant materials and healthy native ecosystems.

References


Manipulating Internal System Feedbacks to Accelerate Reed Canarygrass (*Phalaris arundinacea*) Control: From Theory to Practice
Craig A. Annen (Operations Manager/Director of Research, Integrated Restorations, LLC, 228 S. Park St., Belleville, WI 53508, 608/424-6997, annen00@aol.com).

Reed canarygrass (*Phalaris arundinacea*) displaces indigenous species and creates extensive monocultures that frustrate restoration efforts. Restoration gains are typically short-lived at sites heavily impacted by this species, but suppression may be feasible at sites in the early stages of invasion (Annen et al. 2008). However, even under these conditions, reversal of invasion and replacement of reed canarygrass by desired endpoint species may require 5 to 6 consecutive growing seasons of effort (pers. obs.).

State and transition models predict that internal feedbacks maintain vegetation in one state (reed canarygrass monoculture) rather than an alternate state (remnant sedge meadow). Local and landscape-scale disturbances make sites vulnerable to reed canarygrass invasion, while feedbacks maintain the invaded state and resist restoration to a pre-invasion state. In other words, invaded states are internally reinforced by indirect feedbacks involving interactions among disturbances and species characteristics (Zedler 2009). Litter accumulation is one example of a feedback mechanism that maintains reed canarygrass dominance. Senescent reed canarygrass litter has a suppressing effect on competing species. As reed canarygrass increases in abundance and comprises a greater proportion of a site’s standing crop, more litter accumulates each subsequent growing season, which further hinders emergence of competing species. This feedback cycle helps maintain reed canarygrass dominance and must be broken for restoration to be successful. Using fire to disrupt litter feedbacks is relatively easy compared to uncoupling other feedbacks that maintain a reed canarygrass-dominated state (e.g., hydrological disturbance). Nevertheless, Herr-Turoff (2005) documented that sethoxydim herbicide applications were more effective when disturbances were addressed prior to initiating chemical control efforts. Consequently,
successful reversal and restoration of a reed canarygrass-dominated state requires not only properly implementing effective control techniques, but also disrupting feedbacks that maintain the invaded state. Regrettably, control efforts for reed canarygrass are rarely applied in conjunction with removal of disturbances and manipulation of the feedbacks indirectly responsible for maintaining a system in a degraded condition.

The 186-ha Swamplovers Nature Preserve, located in southwestern Wisconsin, USA, includes a 10.5-ha sedge meadow remnant. When the property was acquired as a nature preserve, this sedge meadow remnant was on a trajectory toward reed canarygrass dominance. Ten hectares immediately north of the sedge meadow had been planted to row crops for several decades. To make the area more suitable for agricultural production, a drainage ditch and drain tiling system had been installed in the sedge meadow, disconnecting it from its original hydrology. Nitrogen levels were low (10.7 ppm NH₄-N and 9.2 ppm NO₃-N), but available phosphorus was high (57 ppm) when measured in 2007. Long-term absence of fire encouraged successive progression to shrub-carr/lowland forest dominated by fire-intolerant shrub and tree species. This change in vegetation composition exacerbated hydrological losses, as these species have high evapotranspiration rates. Three and one-half hectares of the sedge meadow remnant were dominated by reed canarygrass, with additional outliers of reed canarygrass expanding into canopy gaps in relic populations of sedge meadow species. An additional 2.8 ha existed in the wet meadow condition, dominated by a matrix of reed canarygrass intermixed with aggressive perennial forbs such as Canada goldenrod (Solidago canadensis) and sawtooth sunflower (Helianthus grosseserratus).

Restoration began in 1998, when a wet-mesic prairie buffer was planted into the former cropland bordering the remnant (50 species were planted). The drain tile system was destroyed with a backhoe in 1999 to partially restore the site's hydrology. Hydrological restoration was completed in 2007 when the agricultural drainage ditch was filled and 4 small scrape ponds were created. Contouring and scrape pond construction created 0.6 ha of bare ground, a condition that facilitates subsequent reed canarygrass invasion unless a closed vegetation canopy is established. Bareground space was seeded with 60 native species at a rate of 11.9 kg/ha following recommendations of Wisconsin's Reed Canarygrass Working Group (2009). Contractors also planted plugs or bare root tubers of an additional 15 sedge meadow and aquatic vascular plant species. The next phase of the restoration was to remove fire-intolerant trees and shrubs and reintroduce wildfire to the site, which was accomplished in 2009 and 2010.

At this point, with the major disturbances addressed, reed canarygrass suppression efforts were initiated. As expected, reed canarygrass quickly reestablished in the bareground space adjacent to the scrape ponds. The initial response was to apply a 4% glyphosate (Credit Extra, NuFarm Products, Burr Ridge IL) solution to re-emerging reed canarygrass. Thereafter, spring applications of grass-selective herbicides were employed to enable planted and plugged species to survive and establish. A 2.25% (a.i.) solution of sethoxydim (Sethoxydim G Pro, Etigra Manufacturing, Cary NC) and 1.0% (v/v) nonionic surfactant/methylated seed oil blend (Dyne-Amic, Helena Chemical, Memphis TN) was applied to reed canarygrass in May 2008. A 0.5% (a.i.) solution of clethodim (Intensity, Loveland Products, Greeley CO) and 1.0% NIS/MSO was applied to reed canarygrass in April 2009 and 2010.

In the remainder of the sedge meadow remnant, late spring grass-selective herbicide applications were used for reed canarygrass suppression. In 2008, high water levels resulting from snow melt and abnormally high spring rains delayed sethoxydim applications until June, and the onset of panicle emergence in mid-June quickly ended suppression efforts that year. In 2009, reed canarygrass was treated with clethodim from May through June. In 2010, reed canarygrass was again treated with clethodim from April through May. To close canopy gaps created by herbicide applications and provide competition for reed canarygrass, seed from 31 indigenous species (14 graminoids and 17 forbs) were collected from the remnant sedge meadow and interspersed at high rates (the approximate equivalent of 11 kg/ha) into areas denuded by herbicide application.

In the scrape planting, 44 of 60 planted species and 14 of 15 plugged species were observed in July 2010. Reed canarygrass was still present in the scrape planting but comprised less than 5% of the canopy. In the remainder of the remnant, the area covered by reed canarygrass had decreased by 68% following 3 consecutive years of treatment. It is interesting to compare this reduction to the 5 or 6 growing seasons typically required to affect a similar change in vegetation composition when grass-selective herbicides are used in the absence of mitigating underlying disturbances and disrupting feedbacks. There were substantial decreases in the effort and cost required for reed canarygrass suppression as this project progressed. In 2010, treatments required 33% less labor and 52% lower herbicide volume for coverage of the target area compared to the 2009 treatments. Where it was still present, reed canarygrass was intermixed with a diverse variety of native species dominated by the matrix clonal sedges, tussock sedge (Carex stricta) and hairyfruit sedge (C. trichocarpa), and cool-season grass, bluejoint (Calamagrostis canadensis). These species were present prior to reed canarygrass abatement and expanded rapidly in area following litter removal by burning and reed canarygrass suppression with selective herbicides. An indigenous population of Wisconsin-threatened groovestem Indian plantain (Arnoglossum plantagineum) more than doubled in abundance during this time period, with the majority of new individuals arising in areas that were formerly dominated by reed canarygrass.
Intriguingly, soil sampling in 2010 revealed that available phosphorus was 36% lower than in 2007. A more detailed study would reveal whether phosphorus mining can be achieved by annually burning sedge meadows.

While the reed canarygrass has not been completely eradicated from this site, the pace of progress achieved demonstrates how an integrated vegetation management strategy based upon a state and transition framework can enhance and accelerate progress over single-method (e.g., herbicide only) approaches. This approach involves mitigating disturbances (removal of hydrological disturbances), disrupting facilitating feedbacks that reinforce invasions (litter removal), strengthening feedbacks that augment community recovery and invasion resistance (reseeding after herbicide applications), and reestablishing natural disturbance (i.e., fire) regimes. Although the specific management actions described here were site-specific and not appropriate for all abatements, this case study highlights the importance of correcting the underlying causes of invasions in invasive species management.

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