REVEGETATION OF REED CANARYGRASS INFESTED RIPARIAN AREAS: PERFORMANCE OF PRE-VEGETATED COIR AFTER 3 to 6 YEARS[1]

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Abstract: Reed Canarygrass (RCG) is an exceptionally aggressive invader of wetlands and streambanks throughout much of North America and a serious challenge in many riparian restoration projects. RCG generally degrades habitat quality, can outcompete most native grasses and forbs, and can sometimes invade stream channels. Over the last six years, we have used pre-vegetated coir mats for rapid bank stabilization and native plant establishment and to thwart reinvasion by RCG. In this paper, we use field experiments, case studies, and literature to evaluate whether aggressive revegetation can defeat RCG. Our results demonstrate that: (1) successful revegetation of infested sites is possible; (2) although RCG can be excluded for several years, even sites with very successful revegetation are not immune to reinvasion and may require periodic management; and (3) major hurdles include incomplete initial RCG control and poorly understood or challenging hydrology. Factors that favor RCG over native herbaceous plants include rapid spread by seed and rhizomes, high competitive ability, strong response to nutrients and disturbance, broad hydrologic tolerances, resistance to herbicides and other control practices, presence of non-native genotypes, and historical or continuing use as a forage or for bank stabilization. Complete control and replacement of RCG is unrealistic. Decisions about goals and effort for RCG control will depend on RCG abundance in the landscape, project purpose, project sponsors' values, regulatory context, and mandated performance criteria. Depending on the location and agency, RCG may be treated as a regulated noxious weed, an undesirable invasive plant with low value in rating biological integrity, not a concern, or a plant suitable for pastures and other uses. Where minimizing RCG is required, time and resources should be allocated to pre-project RCG control, analysis of site hydrology, intensive revegetation practices, large woody plant material, follow-up RCG control, and monitoring for at least 3 years.

Additional Key Words: invasive plants, streambank bioengineering, riparian restoration, hydrology.

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Introduction

Reed canarygrass (*Phalaris arundinacea*) embodies many of the challenges, complexities, and contradictions of invasive species management in riparian restoration. With remarkable ecological versatility, aggressive spread, resistance to control practices, and stubborn persistence, it is an extremely difficult weed to suppress during and after revegetation and compromises many restoration projects. In addition to its own weedy traits, RCG may be favored by landscape and watershed-scale factors that are beyond the control of restorationists. Furthermore, divergent attitudes towards this species complicate management decisions: in different cases, reed canarygrass may be native or introduced, valued or reviled, planted or attacked, and regulated, ignored, or promoted.

The purpose of this paper is to synthesize available information on reed canarygrass and explore realistic strategies for its management in the context of riparian restoration. In four main sections, we:

1. review the RCG ecology literature with emphasis on factors that influence invasiveness and ecological impacts;
2. summarize previous research on RCG control and revegetation of RCG-infested sites with emphasis on identifying practices with best potential for successful restoration, as well as the serious shortcomings of most conventional practices;
3. report original riparian restoration research and project experience that demonstrates the ability and limitations of some aggressive revegetation practices to suppress and replace RCG;
4. summarize considerations that affect the desirability and feasibility of controlling RCG in restoration work as a basis for project selection.

Restoration of RCG-infested sites appears to be feasible but difficult, protracted, and unlikely to be 100% successful. Therefore, it is probably advisable to pursue RCG replacement only where legally mandated and/or strongly desired by a landowner and supported with adequate funding, human resources, long-term commitment, and favorable site conditions. In other cases, it may be better to adopt more modest goals for habitat enhancement and plant community restoration that accept coexistence with RCG at some level.

Reed Canarygrass Ecology and Impacts

The extreme challenge presented by RCG reflects its ecology, response to human land use, and historical and continuing use in agriculture and other arenas. Realistic restoration strategies and goals require appreciation of the many factors that contribute to RCG's success and the limited range of practices that are effective in controlling, replacing, and combating it. (The information summarized here and in the following section build on several previous reviews that discuss some topics in greater detail: Antieau, 1998; Stannard and Crowder, 2001; Reinhardt and Galatowitsch, 2004; Tu 2004).

Plant attributes and environmental factors contributing to invasion

Reed canarygrass is a tall, fast-growing, perennial, rhizomatous, cool-season grass of wetlands, riparian areas, and moist meadows and pastures. High competitive ability and accumulation of a thick litter layer contribute to its tendency to form dense stands in which it is the only dominant species, often in near monoculture (Figure 1). The ability to dominate across a broad range of environments adds to its threat.

Figure 1. Reed canarygrass at Silver Creek Preserve, Idaho. Plants are 1-2 m tall and form mono-dominant stands extending up to 30 m from the channel. Invasion of this site may have been aided by: use of RCG for bank revegetation in the area; agricultural fertilizer use in the watershed; construction and partial breaching of a farm pond dam.

The extraordinary competitive ability of RCG against a wide range of species and over a wide range of conditions is well documented (Lavergne and Molofsky, 2004; Reinhardt and Galatowitsch, 2004; Reinhardt Adams and Galatowitsch, 2006). It is highly competitive even by the standards of aggressive, rhizomatous weeds, sometimes out-competing quackgrass (*Elytrigia* *repens*) (Harrison et al., 1996). As an adult, reed canarygrass is well adapted to expand vegetatively from existing populations into areas not conducive to establishment from seed. Parent clones apparently subsidize growth of tillers into shaded microsites (Maurer and Zedler, 2002), with subsequent growth allowing RCG to overtop existing herbaceous vegetation. Massive expansion from small initial populations can occur over one or two decades (Barnes, 1999; Mulhouse and Galatowitsch, 2003; Lavergne and Molofsky, 2004).

Establishment from seed is also a problem on disturbed sites such as newly planted restoration projects. In a three-year-long restoration experiment, *Carex stricta* seedlings were able to outgrow all species recruited from the seedbank except RCG (Budelsky and Galatowitsch, 2004). Rapid initial shoot growth relative to other wetland species may be important to RCG's competitive advantage (Fraser and Karnezis, 2005; Reinhardt Adams and Galatowitsch, 2005). RCG seeds germinate better on exposed soil than flooded sites and seedlings survive and grow faster under exposed conditions (Coops and van der Velde, 1995; Fraser and Karnezis, 2005). RCG and the ecologically similar common reed, *Phragmites australis*, invade riverine marshes more quickly during or following low water years (Hudon et al., 2005; Lavoie 2005).

RCG's positive responses to environmental factors associated with human land and water use contribute to its invasiveness; that is, humans have made many ecosystems more vulnerable to RCG invasion. High nutrient availability appears to favor RCG invasion, especially where sites are disturbed (Wetzel and van der Valk, 1998; Green and Galatowitsch, 2001; Maurer et al., 2003; Kercher et al., 2006), whereas nutrient-poor environments appear to retain native wetland plants (Perry and Galatowitsch, 2002). RCG responds more than native sedges to nutrient additions (Wetzel and van der Valk, 1998; Gusewell, 2005). A native sedge that can out-compete RCG under nitrogen-poor conditions loses this advantage with even modest increases in N (Perry et al., 2004), and relative abundance of RCG and another native sedge shifts with phosphorus and base cation levels (Schröder et al., 2005). High nutrient levels favor RCG vegetative expansion by increasing biomass allocation to shoot growth and enhancing growth of tillers into shaded areas (Maurer and Zedler, 2002).

Reed canarygrass thrives and is highly competitive under a wide range of hydrologic environments including some upland sites (Harrison et al., 1996; Hoag et al., 2001; Kercher and Zedler, 2004; Magee and Kentula, 2005; Mahaney et al., 2005; Miller and Zedler, 2003).

Suitable water levels are narrower for initial establishment, but temporary periods of low water are sufficient to facilitate colonization, which is followed by population growth and persistence under wetter or more variable conditions. Anthropogenic changes in hydrology may facilitate RCG invasion (Herrick and Wolf, 2005; Weddell, 2002); contributing factors may be altered surface water level regimes (shallower, more variable), stream channel downcutting, and lowering of water tables in wet meadows.

Landscape-scale analyses further support the idea that human impacts such as nutrient enrichment, sediment deposition, hydrologic alteration, creation of weed dispersal corridors, physical site disturbance, and removal of native plant populations play an important role in RCG invasion. A reconstruction of the history of RCG in Quebec during the twentieth century suggested that expansion into new sites was due to increased road construction, nitrogen pollution, and periods of drought and low water levels (Lavoie et al., 2005). An analysis of wetlands in an Oregon watershed found that urbanization accounted for much of the variation in RCG abundance, likely due to increased runoff from impervious surfaces and water quality degradation (Inahara, 2003). An evaluation of 41 prairie pothole restorations that relied on passive recolonization found high levels of RCG and other weedy species (Mulhouse and Galatowitsch, 2003); wetlands isolated within an agricultural landscape were cut off from native seed sources and affected by historical and ongoing changes in hydrology, sediment deposition, and nutrients. Such historical and landscape-scale influences present serious challenges to restoration projects that rely entirely on on-site efforts and they argue against fertilizer use in restoration projects.

Ecological Impacts

As a result of its dense growth and dominance, RCG is a threat to native plant communities, aquatic habitat quality, and stream restoration projects. It can reduce native floristic diversity drastically (Barnes, 1999; Maurer et al., 2003; Kercher et al., 2004; Schooler et al., 2006). Houlahan and Findlay (2004) found that lower native plant species richness in communities strongly dominated by RCG was mainly due to exclusion of rare native species. In some cases endangered species or culturally significant Native American food plant populations are reduced (Lesica, 1997; Weddell, 2002). Diversity can be lowered even where invasion by RCG replaces a strongly-dominant native such as a sedge or cattail (Werner and Zedler, 2002).

Aquatic ecosystems can also be affected. According to Ringold et al. (2007), RCG is "associated with reduced biotic condition of fauna in western US streams". They sampled stream vertebrate and macroinvertebrate communities in more than 1000 stream reaches and recorded presence of riparian weeds. In some regions, indices of biotic condition were significantly lower where reed canarygrass or other invasive plants were present.

RCG is typically limited to streambanks and channel margins that are exposed periodically during low water periods. However, it can sometimes affect instream physical conditions directly. Dense stands in stream channels or ditches can retard water flow, reduce channel definition, promote sediment accumulation, and impede fish migration (Antieau, 1998; Carrasco, 2000; August et al., 2006).

Distribution, Introduction, Uses and Invasive Status

A history of deliberate introduction and practical use complicates RCG's management as an invader. RCG is native to Europe and western Asia and possibly North America but is now widely distributed globally. In North America, it occurs north of Mexico except in Texas, some southeastern states, and the far northeast of Canada (USDA Plants Database, http://plants.usda.gov, accessed January 2009). RCG is considered an invasive weed in many countries including parts of the United States.

Populations in North America are believed to be a mixture of native and introduced genotypes (Harrison et al., 1996; Merigliano and Lesica, 1998). RCG was present in the inland Pacific Northwest before settlement by European peoples (Merigliano and Lesica, 1998). Introduced, weedy genotypes or native-introduced hybrids may be responsible for RCG's huge increase in abundance, but this remains speculative. Invasion by exotic genotypes of native species has been documented for the common reed *Phragmites australis* (Lavoie et al., 2005). There is substantial genotypic diversity within and between wild and pasture populations, and genotypes vary in early survival, growth, and ability to compete with existing vegetation (Gifford et al., 2002; Morrison and Molofsky, 1998; Molofsky et al., 1999).

Agricultural use of RCG in the U.S. dates to at least the 1830s; agronomic research began before 1920, and named varieties were released in the 1940s (Stannard and Crowder, 2001 and 2002; Tu, 2004). Development of varieties more suitable for livestock grazing has contributed to widespread use of RCG in pastures. The USDA and land grant state universities have had breeding programs and actively promoted RCG as forage; this continues in some states. Seed is commercially available from many sources. North American RCG populations now include native genotypes, cultivars that have been developed by deliberate selection using native or introduced genotypes, and crosses between native, introduced, and cultivated genotypes (Lyons, 1998; Casler, 2003). There are no known morphological traits that distinguish among native and non-native or hybrid types (Lyons, 1998).

Widespread and effective use for ditch and streambank stabilization has also contributed to the spread of non-native (or at least non-local) genotypes (Stannard and Crowder, 2002; Casler, 2003; Pick et al., 2004). Some agencies, including in the Rocky Mountain region, continue to recommend use of reed canarygrass in bank stabilization and applications such as stormwater filtration, wastewater treatment wetlands, riparian revegetation, and even wildlife habitat improvement (e.g. Ogle and Hoag, 2000; Hoag et al., 2001). Concerns about invasiveness and non-native status are not always mentioned. Potential use of RCG for bioenergy production could result in increased planting of non-native RCG varieties.

Official policy towards promotion or control RCG is inconsistent. It is a prohibited, listed weed only in Washington and Massachusetts (USDA Plants Database, http://plants.usda.gov). Many government natural resource agencies involved in ecological conservation and restoration regard it as an invader, while agencies with a more pragmatic soil-and-water-conservation focus recognize its utility and threats (Stannard and Crowder, 2002; Pick et al., 2004). Conservation organizations and biologists generally view RCG as a pernicious weed and a serious threat to riparian and wetland ecosystems (e.g. Lyons, 1998). The situation in Minnesota illustrates the conflicted status of RCG particularly well: the University of Minnesota is home to a RCG breeding program (in cooperation with the USDA) and a restoration ecology program that has conducted some of the best RCG control research (in cooperation with the state's Departments of Natural Resources and Transportation). Given the diversity of revegetation goals in land rehabilitation – from channel stabilization, stormwater management, and livestock grazing to recreating native ecosystems – conflicting attitudes towards RCG are bound to persist.

Control and Revegetation

Control

Research and management experience supports two general conclusions about reed canarygrass control: (1) regardless of control methods used, repeated treatments over several to many years are required; (2) re-establishment of reed canarygrass (though not necessarily dominance) should be treated as a risk in any restoration project where reed canarygrass is initially present or occurs in adjacent areas (Antieau, 1998 and 1999; Lyons, 1998; Reinhardt and Galatowitsch, 2004; Tu, 2004; Reinhardt Adams and Galatowitsch, 2006).

Reinhardt and Galatowitsch (2004; Reinhardt Adams and Galatowitsch 2006) recently published results of the most comprehensive, well designed field test of reed canarygrass control yet reported. Late-August and September glyphosate applications were more effective (>90% control) than mid-May application (75% control). Late-season efficacy was associated with seasonal transport of carbohydrates to rhizomes. Spring burning alone did not control reed canarygrass, and herbicide efficacy was not enhanced by a spring burn. However, burning did reduce the reed canarygrass seed bank, probably by inducing germination before herbicide application, and this might aid restoration by limiting recolonization from seed.

Selective herbicides show some promise for reed canarygrass control, but effective use has not been demonstrated. Annen et al. (2005) tested the grass-selective, systemic herbicide sethoxydim ("Vantage") on a reed canarygrass-dominated community in Wisconsin. Reed canarygrass biomass was reduced by 50% compared to controls in the year of application, and seed head density was reduced by over 90%. However, these effects did not persist into the next year and non-target species including native sedges did not increase significantly. Repeat applications over several years could be more effective and should be tested.

Practices that attempt to draw down plants' carbohydrate reserves by damaging them physically (mowing, grazing, disking, burning), shading them (synthetic mulches such as weed barriers), or flooding them deeply may require several years to reduce reed canarygrass and may not eliminate it (Antieau, 1998). These practices are more effective if combined with herbicide. Flooding may need to be deep and sustained over 2-3 years or more to eliminate stands and thus requires artificial hydrologic control. Short-term flooding at > 0.85 m depth caused modest decreases in RCG, with better results in areas of regenerating willow forest (Jenkins et al., 2008).

Depletion of reed canarygrass in the seedbank is also desirable and may be attempted by repeated disking, herbicide application, and fallowing over several growing seasons (Antieau, 1998). This may also help reduce other weeds that can replace reed canarygrass after short-term herbicide control (NRCS, 2005).

Revegetation after reed canarygrass control

Good initial control of reed canarygrass typically does not translate into longer-term success. Where control with glyphosate is incomplete, shoots may grow back at even greater density (Kilbride and Paveglio, 1999). Reed canarygrass can produce abundant seed, creating a large seedbank (Galatowitsch and van der Valk, 1996) that may allow rapid reestablishment after existing plants are killed. Surviving rhizome fragments may also allow reestablishment. In spite of apparently effective control in Reinhardt and Galatowitsch's (2004) experiment, reed canarygrass recolonized rapidly and prevented establishment of native species. Extremely high rates of seeding with native species (15,000 seeds/m^2) did not suppress recruitment of reed canarygrass from seed even when reed canarygrass seed density was very low (10 seeds/m^2). Because shading inhibits reed canarygrass germination, rapid establishment of other plants using vegetative plant materials might be able to suppress its establishment from seed (Lindig-Cisneros and Zedler, 2001, 2002a, 2000b). Mahaney et al. (2005) concluded that where restoration relies on seed, reed canarygrass is likely to dominate a wide range of sites regardless of hydrologic conditions, sedimentation, or nutrient enrichment. Passive revegetation has led to high levels of reed canarygrass in restoration projects in disturbed agricultural landscapes where native plants are scarce (Mulhouse and Galatowitsch, 2003).

Foster and Wetzel (2005) tried to establish native plants from seed and transplants after a single herbicide application or burning. Herbicide resulted in better initial native plant establishment than burning, but reed canarygrass increased rapidly while planted natives declined over the two year study period. The authors concluded that native species "needed more than two growing seasons to become established enough to compete with *P. arundinacea* sprouting from the seed bank or surviving rhizomes." Seeding and exclusion of deer resulted in at least short-term (3 year) increases in species richness after "non-catastrophic" disturbances that did not reduce reed canarygrass abundance (Kellogg and Bridgham, 2004); communities were still dominated by reed canarygrass. Tests of a cover crop strategy were not promising because both reed canarygrass and native sedges were suppressed (Perry and Galatowitsch, 2003).

Mark Stannard and colleagues at the USDA's Pullman, Washington, Plant Materials Center are testing several control and revegetation strategies. Results have not been published but include (M. Stannard, personal communication; NRCS 2005):

- Poor revegetation success with grasses, legumes and forbs, with recolonization by reed canarygrass after a few years.
- Some success with rooted woody plants, but limited success with cuttings.
- Success with woody plants has required use of a weed barrier, in addition to initial herbicide application, because control by legal herbicides does not persist long enough. Without a weed barrier reed canarygrass recolonizes or other weeds such as Canada thistle take over. However, installing weed barrier properly is very labor intensive.

Tim Miller and Craig MacConnell of Washington State University Cooperative Extension tested methods to control reed canarygrass and establish riparian trees in Western Washington (unpublished results summarized by Dobrowolski and Miller, 2004). They achieved good tree establishment after chemical control of reed canarygrass if follow-up maintenance was done to limit regrowth of reed canarygrass and broadleaf weeds. Follow-up weed treatments included spot application of glyphosate, which had the best results, clipping, which was effective but used 30% more labor, and mulching with wood chips, which only worked if initial reed canarygrass control was complete. Tree protectors aided survival and growth. Other restorationists have also reported promising results with tree planting or transplanting (Moore et al., 2000; Naglich, 2000).

A trial using willow cuttings after reed canarygrass control showed promising results for a high intensity approach (Kim et al., 2006). During one year, test plots were mowed twice, sprayed with glyphosate three times, and covered with 10-15 cm of woodchip mulch. The next year 3-ft long willow cuttings were planted at high density (spaced 2, 3, or 4 ft apart). Willows grew well and shaded reed canarygrass, reducing its biomass by 45-68% within two years.

Intermountain Aquatics' Revegetation Research and Experience

We have been testing riparian revegetation with pre-vegetated coconut fiber mats since 1999, first with small-scale plot experiments, then in full-scale restoration projects. Our main objectives in using pre-vegetated mats, generally in combination with other practices, have been rapid native plant establishment for bank stabilization, revegetation under challenging hydrologic conditions, weed suppression, native plant community restoration, habitat improvement, and aesthetics. We have previously presented guidelines for planning and implementing projects with pre-vegetated coir (Hook and Klausmann, 2006) and a case study (Hook, 2006). Here we present highlights of research and project observations related to suppression and replacement of

RCG. The central question we have asked is whether pre-vegetated coir can accomplish what no other approach has been able to do reliably: reestablish native herbaceous vegetation on RCG-infested riparian sites.

The pre-vegetated mats we use have been planted with bare-root or container-grown graminoids at a density of approximately 20 plants/m^2 (20-cm spacing) and grown to ≥50% cover in our nursery (Hook and Klausmann, 2006). They have well-established vegetation thoroughly rooted into the mat and are marketed under the name "Wetland Sod". This material and similar products grown by several other U.S. nurseries should not be confused with coir mats that are planted immediately before field installation; those mats, which are sometimes called "pre-planted", are used by some practitioners. The plants we normally use are sedge, rush, spikerush, bulrush, and grass species.

Initial small-scale demonstration of weed suppression

In 1999-2000, we compared seven methods for revegetating sedge-dominated wetlands in northwest Wyoming using a randomized, replicated experiment with five 9-m^2 (10-yd^2) plots per treatment (Klausmann and Hook, 2001). The seven methods spanned a wide range of cost and effort, the most intensive being pre-vegetated, nursery-grown coir mats; the others were passive revegetation (unplanted control), broadcast seeding, salvaged marsh surface (SMS), greenhouse-propagated bareroot plants, greenhouse-propagated containerized plants ("tubelings"), and wild-collected transplants. We planted Nebraska sedge (*Carex nebrascensis*), northern territory sedge (*C. utriculata*, previously reported as *C. rostrata*) and hardstem bulrush (*Schoenoplectus acutus*) in each treatment except for control and SMS. Saturated to shallowly flooded conditions were maintained during the growing season.

All of the vegetative methods provided superior establishment of planted species compared to passive revegetation, broadcast seeding, and salvaged marsh surface. However, the most striking result was the uniquely effective suppression of weeds by pre-vegetated mats compared to other treatments (Fig. 2). Vegetated mats had ≤0.5% cover of exotic or invasive species at the end of the second growing season. This was likely due to both the mulching effect of the coir mat and the rapid establishment of planted species.

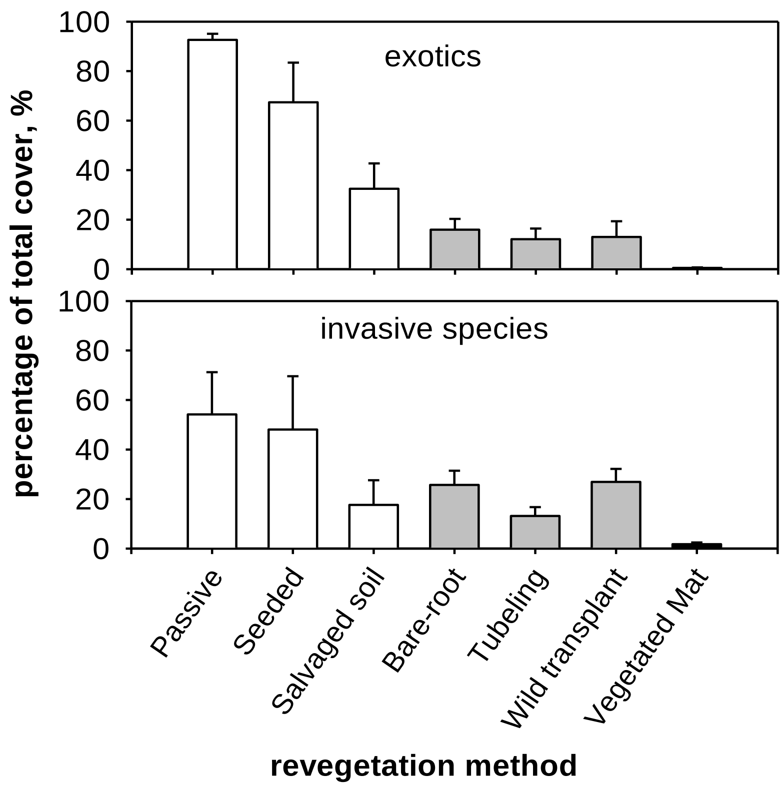

Figure 2. Fraction of plant cover in exotic, and invasive species (mean + SE) in plots revegetated with seven different methods. Unfilled bars are non-vegetative methods. Gray bars are vegetative methods using individual plants. Black bars (far right) are pre-vegetated coir.

Floodplain Revegetation After RCG Control

In 2003, The Nature Conservancy (TNC) initiated a project to enhance stream habitat and restore RCG-dominated riparian areas at the Silver Creek Preserve in central Idaho. The preserve includes 883 acres and several spring-fed creeks that support extraordinary densities of stream insects and a regionally significant trout fishery. Supporting aquatic habitat quality and the fishery are top priorities. RCG is a problem on many of the preserve’s streambanks and floodplains. Although protected and managed for ecological resources, streams are affected by nutrients from agriculture in the watershed, small impoundments, and residual effects of prior land use on the property.

Methods. In the first of two experiments at Silver Creek Preserve (Hook and Klausmann, 2008), we tested several revegetation methods with the objective of replacing RCG-infested riparian vegetation with a higher quality native wetland community. Because research and management literature indicated that revegetation by seeding after RCG control was likely to fail, we focused on higher intensity revegetation methods. Four revegetation treatments, listed in order of decreasing intensity, were used at six replicate floodplain study sites:

1. Wetland sod (pre-vegetated coir mats, "SOD")
2. High density plugs (10 cubic inch containers 8.25 inches deep) planted on 20 cm (8 inch) centers in hydro-mulched plots ("HDP")
3. Medium density plugs (10 cubic inch containers 8.25 inches deep) planted on 60 cm (2 ft) centers without hydromulch ("MDP")
4. Unplanted Control (passive revegetation, "CON");

The first two active revegetation methods were much more aggressive than conventional practice both in initial plant material and mulching to suppress RCG and other weeds. The third method is within the range of conventional practice, though still relatively intensive.

At each site, five 5 x 5 m (16 x16 ft) plots with 2 m (6 ft) buffers were established, and the four treatments plus an untreated plot representing background conditions (BAK) were assigned randomly to plots. Plots other than the untreated ones were mowed in May, 2005, and sprayed with a non-selective herbicide (glyphosate) in June and again in August. Herbicide efficacy was evaluated in September, 2005, using ocular cover estimates. Prior to treatment, all study sites were strongly dominated by RCG (70-85% of total plant cover) and had dense RCG litter (0.8 - 2.6 kg/m^2, 2- 8 cm deep [0.2 - 0.5 lb/ft^2, 1 - 3 in deep]).

Spraying with glyphosate quickly killed the RCG and other plants. Four weeks after the second application, only trace amounts of RCG with green leaves remained. The following spring, immediately before planting, total plant density averaged 1.4 plants/m^2 (0.13 plants/ft^2). The most abundant of the remaining plants were RCG, arctic rush, field sowthistle, Kentucky bluegrass and Canada thistle. Subsequent results suggested that low amounts of RCG rhizomes had probably survived the herbicide.

In May of 2006, plots were prepared for planting as follows: SOD plots were raked to remove litter, expose soil, and to create a relatively even surface; HDP plots were hydro-mulched with EcoAegis® at 3500 lb/acre (80 lb/1000 ft^2); MDP plots had no additional preparation and litter was left in place. Plots were planted May 31-June 2, 2006. All three active revegetation treatments used one-third Nebraska sedge (*Carex nebrascensis*), one-third Northwest Territory sedge (*Carex utriculata*), and one-third arctic rush (*Juncus arcticus*), with individuals of the three species alternating within the planting grid. Plots were watered daily during the first few weeks and then every 2-3 days. Cover was estimated ocularly on September 27, 2006, and August 29, 2007, to evaluate initial revegetation success and abundance of RCG; treatment effects were evaluated with one-way, blocked ANOVA with alpha = 0.1. Sites represented the range of hydrologic conditions over which RCG occurred, from intermittently flooded to deeply sub-irrigated (water at 2-3 ft depth, surface dry) based on groundwater well monitoring in 2005-2006.

<u>Results</u>. At the end of the first growing season (2006), percent cover of the planted species and RCG differed among treatments ($p < 0.0001$ and 0.027, respectively; Figure 3). Planted species cover was generally higher in HDP and SOD plots than MDP plots and only at trace levels in unplanted controls. Percent RCG cover was low in SOD plots, intermediate in the HDP and control plots, and highest in MDP plots. Though low compared to planted species, RCG cover in SOD and HDP plots was high enough to be a concern after just four months. In SOD plots most of the visible RCG plants were in small gaps between the coir mats. Elevated RCG in MDP compared to HDP plots was likely due to the combination of watering, no hydromulch, and relatively low density planting. Wetter site conditions generally enhanced growth of both RCG and planted species (for regression of cover against mean depth to water $R^2 = 0.18\text{-}0.95$ for RCG depending on revegetation treatment, $R^2 = 0.29\text{-}0.87$ for planted species).

At the end of the second growing season, percent cover of the planted species and RCG again differed among treatments ($p < 0.001$ and 0.056). Differences in RCG followed the same pattern as the previous year but cover had increased by 15-25% in all treatments, reaching 20-50% (Fig. 3). RCG reestablishment was strongly favored by wetter conditions in control, MPD, and HDP plots ($R^2 = 0.71$ for cover versus depth to water). RCG cover in SOD plots was not related to water table depth and was generally lower than other treatments for a given site suggesting some continuing RCG suppression. Measured as either cover of planted species or the ratio of planted species to RCG, performance generally improved with intensity of revegetation (Fig. 3 and 4). Controls were worse than untreated plots (BAK), and MPD plots were no better than untreated plots. Although planted species were twice as high in HDP and SOD plots as untreated plots, which suggests a degree of short-term success in reestablishing natives, the already high RCG levels do not bode well for long-term success.

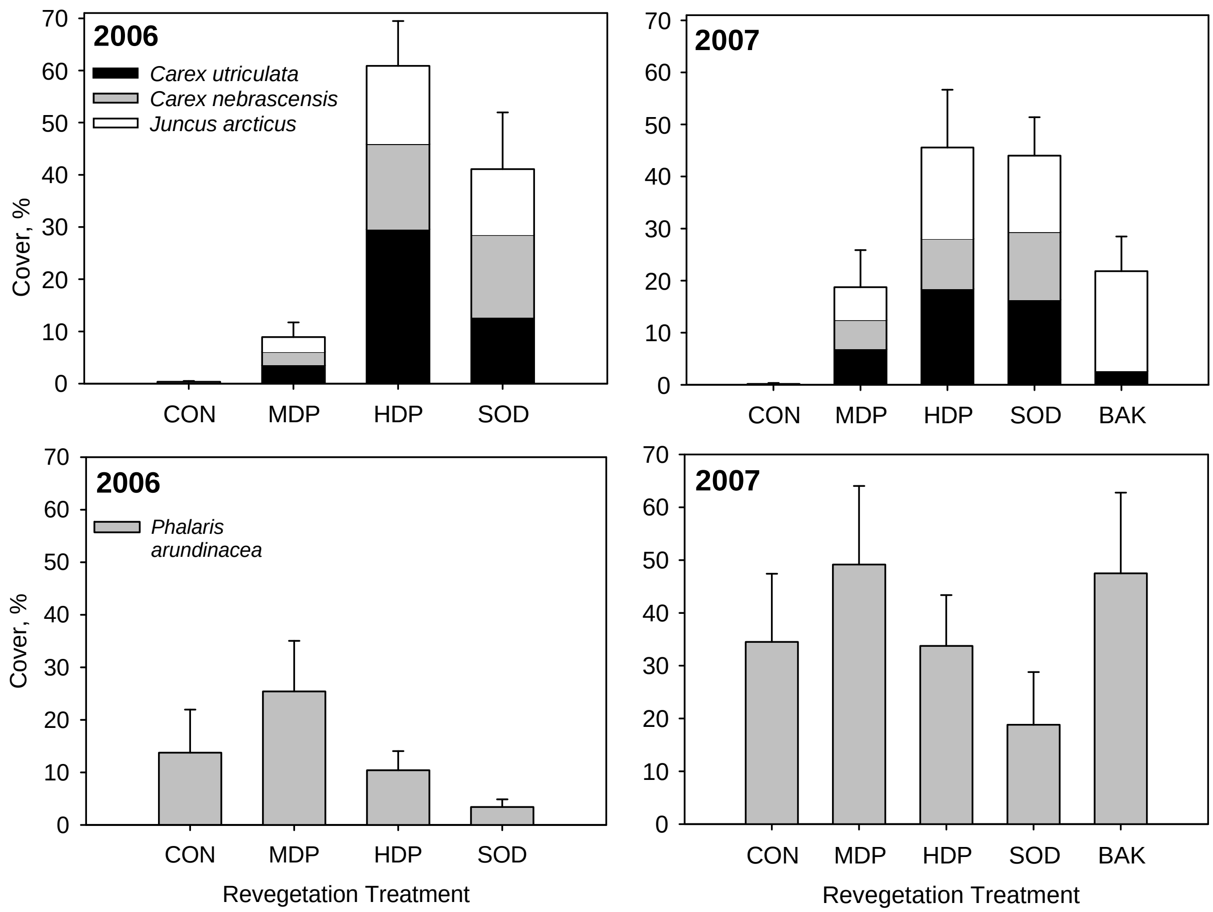

Figure 3. Cover of three planted species and reed canarygrass in plots at Silver Creek Preserve, Idaho, at the end of the first and second growing season after revegetation. Values are means of six replicate sites. Error bars are one standard error for the sum of all planted species or reed canarygrass. Untreated plots, which represent the background RCG-infested condition, were not sampled in 2006. See text for explanation of treatments and abbreviations.

Performance of pre-vegetated coir mats in this experiment was inferior to our experience in many stream and pond projects. This may have resulted from two obstacles. First, soils were not removed from study sites, leaving a significant residual RCG population after mowing and herbicide. Banks are often reconstructed during riparian restoration and bank stabilization, which removes roots and rhizomes. Without soil removal, repeat herbicide application over several years probably would have been needed for adequate RCG control. Second, there was an extreme drought during the first growing season; rainfall totaled only 1.0 cm (0.4 inches) in

June-August, and <2.5 cm (1 inch) in September. Even with watering and litter removal, many plants in the pre-vegetated coir died or failed to grow due to drought stress exacerbated by poor initial root contact with the soil. Plots with the most consistently wet conditions did the best, and sites that were initially dryer and then wet later in the growing season were more susceptible to colonization by RCG. Water levels on most of the floodplain sites diverged strongly from the adjacent stream and ponds making planning based on site elevation inappropriate.

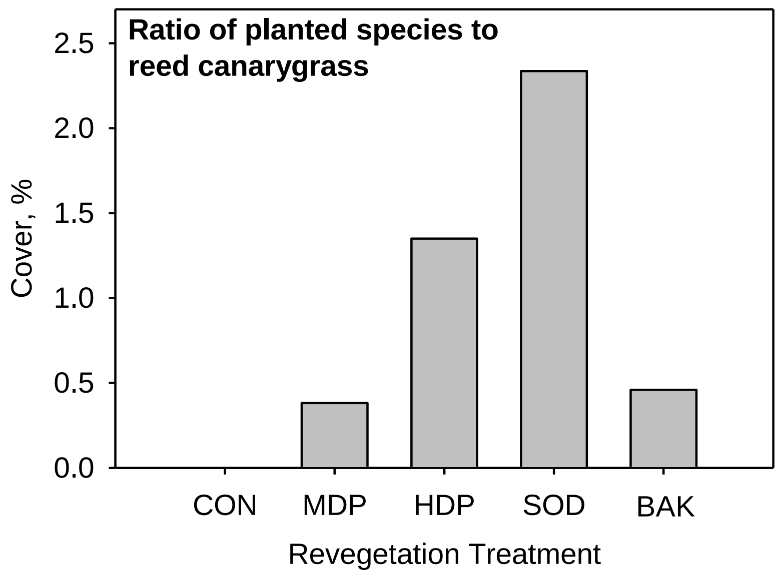

Figure 4. Ratio of total planted species cover to reed canarygrass cover at the end of the second growing season after revegetation.

Overall, these results confirm that conventional revegetation methods are totally inadequate for restoring native herbaceous vegetation to RCG-infested sites without major eradication efforts (well beyond normal site preparation and weed control). Even very high intensity practices such as pre-vegetated coir mats may not succeed without such control measures and good characterization of site hydrology.

Spring Creek Restoration Trial

Methods. In the second experiment at Silver Creek Preserve, we tested the use of pre-vegetated coir mats for riparian revegetation after channel narrowing. In the fall of 2003, TNC installed a series of channel constrictors in Stalker Creek, one of the main tributaries to Silver Creek, to increase in-stream habitat diversity. Coir logs ("biologs") were used to create a narrower, more sinuous channel then backfilled with dredged streambed material; fill depth decreased downstream and was less than the design called for in much of the area. The constrictors were installed within the existing channel and the original streambanks, which supported moderate to very dense RCG, remained unaltered. Reaches upstream of the restoration site are also dominated by RCG. Average channel width was 19.5 m (64 ft) before and 12.5 m (41 ft) after installing constrictors.

In 2004, five channel constrictors (660 m^2) were revegetated using closely spaced (15 cm) herbaceous wetland plants. Plant survival was poor, primarily due to inadequate depth of fill behind the biolog forms and subsequent prolonged inundation during the growing season. In June, 2005, pre-vegetated coir mats were installed on the constrictors, with no additional fill, to test their ability to establish under these difficult conditions. Pre-vegetated coir mats contain plants that are larger and more tolerant to flooding than the individual plants originally used. Sixty percent of the mats contained a mixture of Nebraska sedge, Northwest Territory sedge, and arctic rush; this was installed on the higher elevation areas. The other 40% of mats contained hardstem bulrush (*Schoenoplectus acutus*) and was installed in lower areas.

Planting success was measured using estimates of cover in belt transects for two years and with photo points and informal observation subsequently. Water levels were monitored at a stream gauge in the study reach and used in combination with a detailed elevation survey to estimate hydrologic regimes within each revegetated area. Abundance of submerged aquatic vegetation (SAV) was also monitored to evaluate possible effects on hydrology.

Results. At the end of one growing season in September, 2005, pre-vegetated mats were fully rooted and thriving and plant cover averaged 67% (Fig. 5). Although this was an impressive success in overcoming the barriers to survival that plagued the initial revegetation attempt, two potential problems were observed (Fig. 6). First, significant areas of grazing by muskrats were seen, especially in the deepest areas of the sedge/rush mixture. Second, there were early signs of RCG invasion that appeared to originate from floating plant fragments or from the neighboring RCG stands that had not been controlled. Nine small RCG plants were found on the biologs that formed the channel edge; they appeared to have floated and lodged on the biologs. Fragments of RCG rhizomes with green shoots were seen floating downstream attached to aquatic plant detritus. Four more RCG plants were found within a planted area. All RCG plants had been dug out and removed from the constrictors before planting, so these RCG plants presumably floated in, too. Additionally, RCG rhizomes had extended a short distance into filled areas from existing stands on the original bank, mainly where fill was not covered with pre-vegetated coir.

Figure 5. Revegetation of a channel constrictor on Stalker Creek using pre-vegetated coir mats (wetland sod). This site had the highest elevation of the areas revegetated (most fill) and had the most successful establishment of sedge/rush native vegetation and the least muskrat damage. Top left: Unsuccessful establishment of sedge plugs in 2004; mortality increased through spring, 2005. Top right: The same site after installing wetland sod in spring, 2005, at low water. Bottom left: Fall 2005, at high water; nearly full native cover. Bottom right: Fall 2007; native cover remains intact, but reed canarygrass is encroaching from the channel edge after three growing seasons.

Figure 6. Top left: Muskrat damage at a mid-elevation revegetation site with inadequate fill. Top right: Individual reed canarygrass plant rooting on a biolog at channel edge at end of first growing season in 2005. Bottom left: The same site at the end of the 2007 growing season; RCG has expanded along the biolog but not into the deeper area with very shallow fill that was planted with bulrush (left of RCG), or the channel. Bottom right: Three year old willow grown from cutting planted into a square of weed barrier and surrounded by RCG.

By the end of the second growing season, planted areas were a mosaic of healthy vegetation and open areas resulting from muskrat herbivory combined with prolonged inundation. Native sedge/rush vegetation was dense and robust only in locations that were originally constructed with adequate fill and subject to lower depth and duration of flooding. Openings without plants expanded substantially in lower lying sedge/rush areas. Bulrush matured in deeper areas. RCG plants mapped a year earlier persisted in most but not all locations and one new plant was found. Some RCG plants had expanded from a few stems in 2005 to clumps 15-45 cm (0.5-1.5 ft) across, while others appeared stressed and several had died.

These trends continued in 2007. At the end of August, RCG had extended along significant lengths of biologs and more plants had established in the interior of revegetated areas, especially at intermediate depths where muskrat damage opened up the sedge/rush canopy but inundation was less than in the bulrush stands. Performance differences resulted from relatively small discrepancies in fill; average elevations of the different constrictors only varied by 25 cm (0.8 ft) and specific microsites varied by up to 40 cm (1.4 ft). RCG also increased gradually on both the landward and channel sides of areas that had achieved complete cover of sedges and rushes. No other weeds had invaded in noticeable amounts.

These results suggest that on sites with suitable hydrology, wetland sod can provide excellent revegetation in the short-term but will not exclude RCG indefinitely. Wetland sod is not an effective revegetation method on sites that are too deep to sustain sedges and rushes in the presence of muskrats but not deep enough to prevent RCG establishment. Unusual hydrology on Stalker Creek exaggerated this challenge. Seasonal growth of SAV raised water levels by 1.5-2.1 ft between early May and mid-August in both 2005 and 2006 by retarding flow. Stream stage did not track discharge but were correlated with changes in abundance of SAV ($r = 0.77$), resulting in protracted inundation during the peak growing season on sites with inadequate fill. With this hydrology, building channel constrictors with a clearer demarcation between deep bulrush backwaters and emergent sedge/rush benches would have been more effective at limiting RCG invasion of the channel and delaying or reducing invasion of the riparian zone. This challenge may be specific to shallow, low-gradient streams lacking strong seasonal variations in flow; Stalker Creeks' slope was 0.02-0.05% in the study reach, while flow measured downstream was steady at 25-50 CFS except during one major runoff event.

Willow cuttings were also planted in some biologs and streambank areas during the initial revegetation effort. Survival was poor on biologs but fair on streambanks. Growth of streambank willows, which were planted into patches of weed barrier, was slow; surviving plants typically had several live, branching shoots 0.6-1.2 m (2-4 ft) tall after 4 years (Fig. 6).

<u>Restoration project experiences</u>

<u>Teton River</u>. Beginning in 2002, we completed several projects on the Teton River in Idaho. Streambanks have been degraded by grazing and erosion in many locations and do not recover readily after grazing is stopped. Muskrat burrowing, ice scouring, and continuing erosion of the base of banks help maintain degraded conditions. RCG is abundant along the Teton and is a

threat to any restoration project on the river. Because it has such wide hydrologic amplitude, it can be a problem from lower to upper banks.

The sequence of photographs in Fig. 7 shows changes in vegetation following bank reshaping and planting at one site. In the fall of 2003 we regraded the bank from nearly vertical, established a new slope with two lifts of soil enclosed within erosion control fabric, and planted willow cuttings. After high water in 2004, we installed two layers of pre-vegetated coir mats. The two layers used different species mixes to fit the hydrologic gradient from lower to upper bank. The erosion control fabric and staked coir mats provided initial erosion control.

Herbaceous vegetation dominated by sedges developed rapidly in the pre-vegetated mats, providing full cover during the first growing season. The lower bank had observable sediment deposition the following year. Willows grew in gradually but approached mature height within 5 years. For the first 4 years, RCG did not grow into the planted sedge zone in significant amounts although it was present in the adjacent stream reach and was starting to become established immediately upstream from the restored bank. No RCG control has been necessary so far. Willows are large enough to escape shading by RCG if it reinvades in the future. From a fisheries perspective, the inundated sedge zone now provides excellent juvenile trout habitat.

Specific conditions and results vary among sites. In some places, RCG has reinvaded upper slopes and outcompeted willow cuttings and seeded plants. Even in these situations, however, RCG has not increased markedly in the zone planted with pre-vegetated coir. Reducing slopes of over-steepened banks, ideally to 5:1 or gentler, appears to allow sedges to outcompete RCG in a zone with favorable hydrology. Well-characterized hydrology and good reference sites have helped us place pre-vegetated coir optimally on the Teton River.

These results suggest a general strategy: Pre-vegetated coir mats can be used to limit RCG in the short-term, as well to stabilize banks and reestablish native graminoids. Several years with low RCG pressure will allow establishment of shrubs that will be immune to RCG competition and may partially suppress RCG by shading. If sustained and relatively complete control of RCG is desired, it may require periodic control, such as selective wick application of glyphosate. Even if some RCG is tolerated, a project can achieve channel rehabilitation, partial restoration of the natural plant community, and enhancement of aquatic and riparian habitat. In the best case, native sedges and other riparian graminoids may continue to dominate the shoreline.

Figure 7. Streambank restoration on the Teton River, Idaho. Top left: Pre-project conditions in October, 2003, featured a steep, eroding bank. The upper bank was dominated by introduced pasture grasses, while the lower bank was barren in most places. In the photograph, a clump of reed canarygrass covers the bank below the building; additional RCG occurs upstream in the background. Top right: The bank was regraded to a gentle slope in October, 2003, and reconstructed with two lifts of soil enclosed in erosion control fabric. Bundles of willow cuttings were planted in the upper bank. Bottom left: Two rows of pre-vegetated coir mats were installed on top of the ECF-wrapped soil lifts and staked into place after high water in July, 2004. This photograph was taken immediately after bank revegetation. In addition to erosion control, the pre-vegetated coir mats provided rapid establishment of native vegetation that could resist re-invasion by reed canarygrass. Bottom right: In July, 2008, native herbaceous vegetation was well established. Occasional RCG plants had started to grow into the lower bank but were very sparse and did not require control. Willows were tall enough to overtop herbaceous potential weeds including RCG.

Wilson Creek. While working at Silver Creek Preserve, we also surveyed a nearby spring creek that had been dredged and reshaped in the early 1990s to remove sediment, reestablish desirable channel form, and restore aquatic and riparian habitat (August et al., 2006). Whereas RCG was abundant on the banks of Stalker Creek, it was thick both on the banks and in the channel of Wilson Creek, occupying most of its 20 m (65 ft) width and greatly reducing the area of open channel (Fig. 8). RCG appeared to have grown out onto mats of aquatic vegetation in Wilson Creek, increased in density over time, and stabilized the underlying silt and muck deposits. RCG apparently prevented washout of the aquatic vegetation and sediment deposits that would normally occur during high flows or when aquatic vegetation becomes too thick or dies back. The infestation in Wilson Creek has promoted sediment accumulation and a wide, shallow, poorly-defined, grass-choked channel, undoing an expensive stream restoration project within less than 15 years. In-channel RCG was statistically associated with relatively shallow water (<2-ft deep), dense aquatic vegetation mats, and deposits of silt or organic muck. Shallow, low-velocity channels that support dense aquatic vegetation and do not experience strong seasonal variation in water level and flow may be particularly vulnerable to invasion.

Summary and Recommendations

A number of straightforward conclusions can be drawn from published and unpublished research and our work:

- Effective reed canarygrass control requires extraordinary measures such as repeat use of herbicide over several years (with or without other practices), topsoil removal, or prolonged inundation. Lesser efforts will leave seedbanks or viable rhizomes.
- Establishment of herbaceous native vegetation after incomplete RCG control is unlikely to succeed using passive revegetation, seeding, or individual plants, though combining vegetative plantings with ongoing RCG control and/or water level management may succeed.
- Use of pre-vegetated coir mats can suppress RCG for several years but generally will not prevent RCG reinvasion indefinitely. Using pre-vegetated coir may enable establishment of robust native plant populations that can coexist with RCG on suitable sites, but this outcome is speculative.

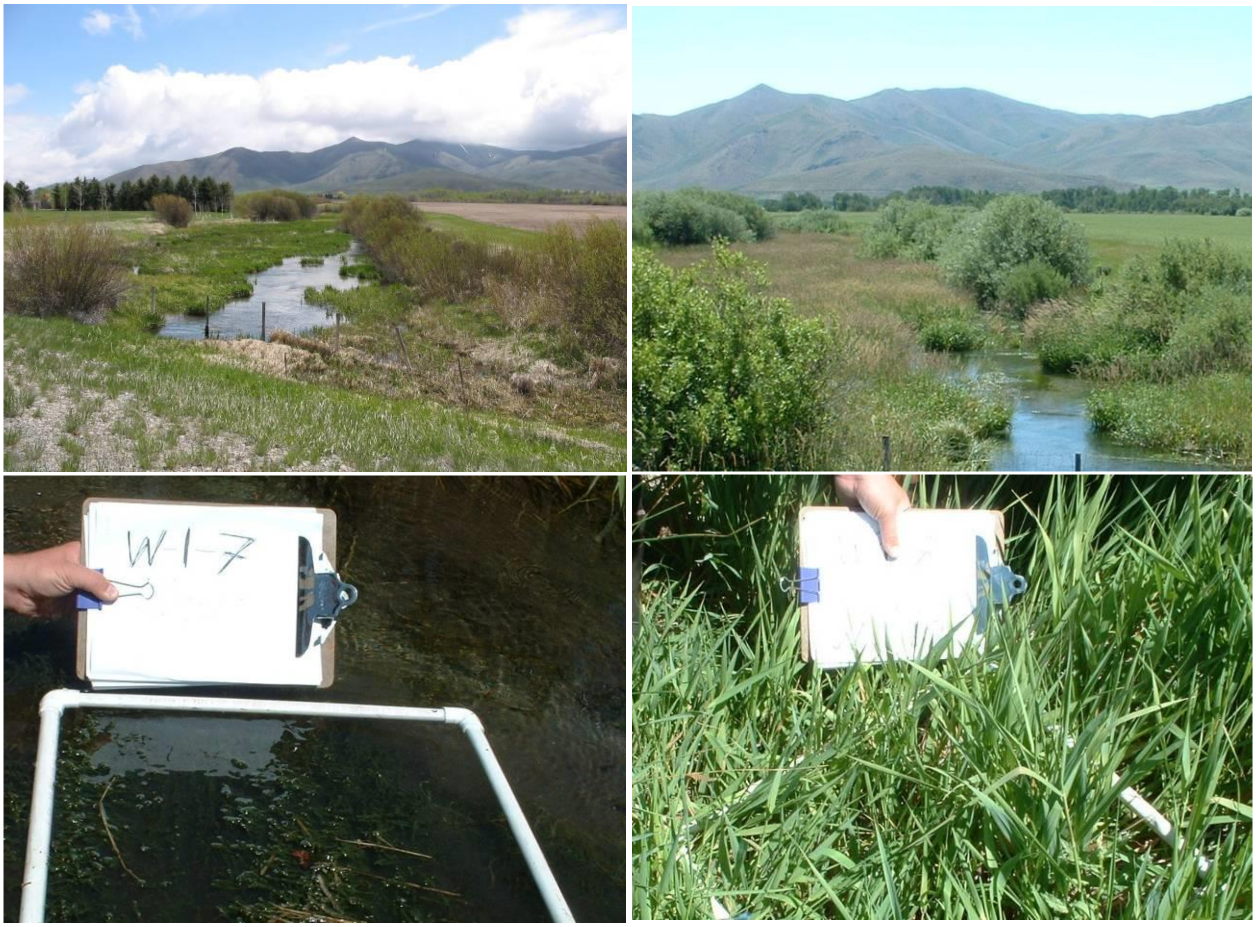

Figure 8. Reed Canarygrass has nearly filled the channel of Wilson Creek here. Top left: May 4, 2005. Top right: Same reach on July 22. Bottom left: aquatic plant mat; beds of *Chara vulgaris* and other species are abundant in area spring creeks. Bottom right: RCG growing on top of an aquatic plant mat; it is not rooted in the channel bed itself. Presence of RCG clumps on top of *Chara* mats was first recorded here by Idaho State University biologists in 1985.

- Native trees and large shrubs are likely to persist in the presence of RCG and can provide habitat benefits in RCG-infested areas. Investment in larger or more mature, rooted plant material will improve woody plant establishment.
- Up-front investment in site preparation, RCG control, and hydrologic information will improve odds of success regardless of revegetation methods and should always be made.
- Many landscape, watershed, and historical factors can work against complete and lasting RCG control and replacement using on-site measures: agricultural fertilizer use; certain hydrological alterations; roading and other landscape disturbance; possible introduction

of more invasive genotypes; past or ongoing RCG use for bank stabilization or pasture; extensive occurrence in many landscapes; and ease of transport by water.

In summary, at least partial restoration of RCG-infested sites is feasible with aggressive and sustained control efforts. The most promising approach with wide application appears to be a "bridging" strategy that combines (a) intensive and/or sustained control measures (repeat herbicide applications, flooding, weed barriers, deep mulching, topsoil removal), (b) aggressive herbaceous revegetation practices that can delay or possibly prevent RCG reinvasion (e.g. pre-vegetated coir mats), and (c) establishment of shrubs or trees during this bridging period. Where successful, this will most likely lead to coexistence of a somewhat degraded native plant community with reduced levels of RCG, the levels depending on site-specific conditions and use or discontinuation of RCG management efforts. European research on common reed (*Phragmites australis*) found that many reed populations that had invaded fens were fluctuating from year to year rather than increasing over time (Gusewell et al., 2000), lending credence to the possibility of long-term coexistence of reed canarygrass and native plant species in some communities.

Considering these challenges, prioritization and realistic expectations are necessary. Commitment of landowners and managers to significant and sustained efforts is critical. At a minimum, this entails relatively intensive and expensive initial work – RCG control, site preparation, and aggressive revegetation – and several years of monitoring and, usually, follow-up RCG control. Such commitment may result from legal or regulatory requirements or the conservation goals of the landowner or manager. Most states have no legal requirements to control RCG. Some agencies may require use of native plants in stream and wetland projects they fund or permit. However, the philosophical commitment of a conservation focused landowner such as a land trust, conservancy, or dedicated individual is likely be the motivation in many cases.

Even with commitment in principle, the difficulty of restoration calls for "ecological triage" in which projects are selected based on both need and potential for success. Examples of high need are to protect critical aquatic habitat or threatened plant communities. Factors that improve or reduce odds of success should be considered. Many of these factors, listed above, amount to the degree of human impact to the landscape as a whole; the best chance of success is in protected areas within relatively less disturbed watersheds. Other factors include the current levels of RCG in the landscape, whether the desired vegetation type includes a significant shrub or tree component, ability to obtain adequate hydrologic data for planning, and future continuity of ownership and management. Lacking appropriate conditions, some areas dominated by RCG may be sacrificed as impractical to restore and, therefore, a poor use of resources.

Finally, appropriate criteria for success need to be defined based on challenges as well as objectives and regulatory obligations. In some cases, partial control of RCG may be enough to maintain reduced populations of desired plant species, establish more shrubs and trees, or enhance riparian and aquatic habitat. In other cases, more complete control of RCG and reestablishment of pre-invasion stream conditions may be sought.

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Literature Cited

Annen, C.A., R.W. Tyser and E.M. Kirsch. 2005. Effects of a selective herbicide, Sethoxydim, on reed canarygrass. Ecological Restoration 23:99-102. http://dx.doi.org/10.3368/er.23.2.99.

Antieau, C.J. 1998. Biology and management of reed canarygrass, and implications for ecological restoration. Washington State Department of Transportation, Seattle, Washington. http://www.ser.org/sernw/pdf/RCG_BIO_MGT.pdf

Antieau, C.J. 1999. Emerging themes in reed canarygrass management. Washington State Department of Transportation Wetland Mitigation Technical Group Meeting, December 9, 1999.

August, E., P. Hook, and K. Salsbury. 2006. *Reed Canarygrass Distribution at the Nature Conservancy's Silver Creek Preserve*. Prepared for Idaho Field Office, The Nature Conservancy, Hailey, ID, by Intermountain Aquatics, Inc., Driggs, ID.

Barnes, W.J. 1999. The rapid growth of a population of reed canarygrass (Phalaris arundinacea L.) and its impact on some riverbottom herbs. Journal of the Torrey Botanical Society 126:133-138. http://dx.doi.org/10.2307/2997289.

Budelsky, R. A. and S. M. Galatowitsch. 2004. Establishment of Carex stricta Lam. seedlings in experimental wetlands with implications for restoration. Plant Ecology 175:91-105. http://dx.doi.org/10.1023/B:VEGE.0000048102.72101.50.

Carrasco, K. 2000. Coho pre-spawn mortalities in a flooded reed canarygrass habitat. Reed Canarygrass Working Group Conference, Washington State Department of Transportation, USDA Natural Resources Conservation Service, and Society for Ecological Restoration –Northwest Chapter, March 15, 2000, Olympia, Washington.

Casler, M.D. 2003. Reed canary grass: friend and foe. 30th Natural Areas Conference, September 24-27, 2003, Madison, Wisconsin.

Coops, H. and G. van der Velde. 1995. Seed dispersal, germination and seedling growth of six helophyte species in relation to water-level zonation. Freshwater Biology 34:13-20. http://dx.doi.org/10.1111/j.1365-2427.1995.tb00418.x.

Dobrowolski, J.P.` and T.W. Miller. 2004. Reed canarygrass – a formidable foe for Washington's riparian areas. Sustaining the Pacific Northwest: Food, Farm, and Natural Resource Systems 2(3):11-13.

Foster, R.D. and P.R. Wetzel. 2005. Invading monotypic stands of Phalaris arundinacea: a test of fire, herbicide, and woody and herbaceous native plant groups. Restoration Ecology 13:318-324. http://dx.doi.org/10.1111/j.1526-100X.2005.00040.x.

Fraser, L.H. and J .P. Karnezis. 2005. A comparative assessment of seedling survival and biomass accumulation for fourteen wetland plant species grown under minor water-depth differences. Wetlands 25:520-530. http://dx.doi.org/10.1672/0277-5212(2005)025[0520:ACAOSS]2.0.CO;2.

Galatowitsch, S.M. and A.G. van der Valk. 1996. The vegetation of restored and natural prairie wetlands. Ecological Applications 6:102-112. http://dx.doi.org/10.2307/2269557.

Gifford, A.L., J.B. Ferdy, and J. Molofsky. 2002. Genetic composition and morphological variation among populations of the invasive grass, Phalaris arundinacea. Canadian Journal of Botany 80:879-785. http://dx.doi.org/10.1139/b02-063.

Green, E.K. and S.M. Galatowitsch. 2001. Difference in wetland plant community establishment with additions of nitrate-N and invasive species (Phalaris arundinacea and Typha x glauca). Canadian Journal of Botany 79:170-178. http://dx.doi.org/10.1139/b00-15.

Green, E.K. and S.M. Galatowitsch. 2002. Effects of Phalaris arundinacea and nitrate-N addition on the establishment of wetland plant communities. Journal of Applied Ecology 39:134-144. http://dx.doi.org/10.1046/j.1365-2664.2002.00702.x.

Güsewell, S., C. Le N´edic, and A. Buttler. 2000. Dynamics of common reed (Phragmites australis Trin.) in Swiss fens with different management. Wetlands Ecology and Management 8:375-389. http://dx.doi.org/10.1023/A:1026553700571.

Güsewell, S. 2005. responses of wetland graminoids to the relative supply of nitrogen and phosphorus. Plant Ecology 176:35-55. http://dx.doi.org/10.1007/s11258-004-0010-8.

Harrison, R.D., N.J. Chatterton, R.J. Page, M. Curto, K.H. Asay, K.B. Jensen, and W.H. Horton. 1996. *Competition, Biodiversity, Invasion, and Wildlife use of Selected Introduced Grasses in the Columbia and Great Basin*. Research Report 155, Utah Agricultural Experiment Station, Logan, Utah.

Herrick, B.M. and A.T. Wolf. 2005. Invasive plant species in diked vs. undiked Great Lakes wetlands. Journal of Great Lakes Research 31:277-287. http://dx.doi.org/10.1016/S0380-1330(05)70259-8.

Hoag, J.C, S.K. Wyman, G. Bentrup, L. Holzworth, D.G. Ogle, J. Carleton, F. Berg, and B. Leinard. 2001. *Users Guide to Description, Propagation and Establishment of Wetland Plant Species and Grasses for Riparian Areas in the Intermountain West*. Plant Materials Technical Note No. WY-5. United States Department of Agriculture, Natural Resources Conservation Service, Interagency Riparian/Wetland Project USDA-NRCS Plant Materials Center Aberdeen, ID.

Hook, P.B. 2006. Pre-vegetated coir achieves rapid results in reconstructed stormwater channel. Land and Water 50 (1):7-11.

Hook, P.B. and J.K. Klausmann. 2006. Dos and don’ts of successful revegetation with pre-planted coir blankets. In *Proceedings of the 2006 Billings Land Reclamation Symposium, June 5-8, 2006, Billings MT*. BLRS and ASMR, Lexington, KY.

Hook, P.B. and J.K. Klausmann. 2008. *Streamside Revegetation and Reed Canarygrass Suppression at Silver Creek Preserve*, Project Completion Report Prepared for The Nature Conservancy, Idaho Field Office by Intermountain Aquatics, Inc., Driggs, ID

Houlahan, J.E. and C.S. Findlay. 2004. Effect of invasive plant species on temperate wetland plant diversity. Conservation Biology 18:1132-1138. https://doi.org/10.1111/j.1523-1739.2004.00391.x

Houlahan, J.E. and C.S. Findlay. 2004. Effect of invasive plant species on temperate wetland plant diversity. Conservation Biology 18:1132-1138. http://dx.doi.org/10.1111/j.1523-1739.2004.00391.x.

Inahara, S. 2003. *Assessing cumulative influences of watershed-scale landuses on reed canarygrass (Phalaris arundinacea L.) abundance*. M.S. Thesis, Oregon State University, Corvallis, OR.

Jenkins, N.J., J.A. Yeakley, and E.M. Stewart. 2008. First-year responses to managed flooding of lower Columbia river bottomland vegetation dominated by Phalaris arundinacea. Wetlands 28 (4):1018-1027. http://dx.doi.org/10.1672/06-145.1.

Kellogg, C.H. and S.D. Bridgham .2004. Disturbance, herbivory, and propagule dispersal control dominance of an invasive grass. Biological Invasions 6:319-329. http://dx.doi.org/10.1023/B:BINV.0000034606.84830.d5.

Kercher, S.M. and J.B. Zedler. 2004. Flood tolerance in wetland angiosperms: a comparison of invasive and noninvasive species. Aquatic Botany 80:89-102. http://dx.doi.org/10.1016/j.aquabot.2004.08.00302.

Kercher, S.M., A. Herr-Turoff, and J.B. Zedler. 2006. Understanding invasion as a process: the case of Phalaris arundinacea in wet prairies. Biological Invasions 9:657-665. http://dx.doi.org/10.1007/s10530-006-9066-9.

Kercher, S.M., A. Herr-Turoff, and J.B. Zedler. 2006. Understanding invasion as a process: the case of Phalaris arundinacea in wet prairies. Biological Invasions 9:657-665. http://dx.doi.org/10.1007/s10530-006-9066-9.

Kilbride, K.M. and F.L. Paveglio. 1999. Integrated pest management to control reed canarygrass in seasonal wetlands of southwestern Washington. Wildlife Society Bulletin 27:292-297.

Kim, K.D. K. Ewing, and D.E. Giblin. 2006. Controlling Phalaris arundinacea (reed canarygrass) with live willow stakes: A density-dependent response. Ecological Engineering 27:219-227. http://dx.doi.org/10.1016/j.ecoleng.2006.02.007

Klausmann, J.K. and P.B. Hook. 2001. *Comparison of Seven Methods for Revegetating Sedge-Dominated Rocky Mountain Wetlands*. Project completion report for USEPA Region 8 104(b)(3) Wetlands Protection Grant to Teton County, WY. Intermountain Aquatics, Inc., Driggs, ID.

Lavergne, S. and J. Molofsky. 2004. Reed canary grass (Phalaris arundinacea) as a biological model in the study of plant invasions. Critical Reviews in Plant Sciences 23:1-15. http://dx.doi.org/10.1080/07352680490505934.

Lavoie C., C. Dufresne, and F. Delisle. 2005. The spread of reed canarygrass (Phalaris arundinacea) in Québec: A spatio-temporal perspective. Ecoscience 12:366-375. http://dx.doi.org/10.2980/i1195-6860-12-3-366.1.

Lesica, P. 1997. Spread of *Phalaris arundinacea* adversely impacts the endangered plant *Howellia aquatilis*. Great Basin Naturalist 57:366-368.

Lindig-Cisneros, R.A. and J.B. Zedler. 2001. Effects of light on Phalaris arundinacea L. germination. Plant Ecology 155:75-78. http://dx.doi.org/10.1023/A:1013224514980.

Lindig-Cisneros, R.A. and J.B. Zedler. 2002a. Species-rich canopies limit the germination microsites for Phalaris arundinacea L. Oecologia 133:159-167. http://dx.doi.org/10.1007/s00442-002-1020-7.

Lindig-Cisneros, R.A. and J.B. Zedler. 2002b. Phalaris arundinacea seedling establishment: Effects of canopy complexity in fen, microcosm, and restoration experiments. Canadian Journal of Botany 80:617-624. http://dx.doi.org/10.1139/b02-042.

Lyons, K.E. 1998. *Element Stewardship Abstract for Phalaris arundinacea L. Reed canarygrass*. The Nature Conservancy Wildland Invasive Species Team, University of California, Davis.

Magee, T.K. and M.E. Kentula. 2005. Response of wetland plant species to hydrologic conditions. Wetlands Ecology and Management 13:163-181. http://dx.doi.org/10.1007/s11273-004-6258-x181.

Mahaney, W., D. Wardrop, and R. Brooks. 2005. Impacts of sedimentation and nitrogen enrichment on wetland plant community development. Vegetatio 175:227-243. http://dx.doi.org/10.1007/s11258-005-0011-2.

Maurer, D.A., R. Lindig-Cisneros, K.J. Werner, S. Kercher, R. Miller, and J.B. Zedler. 2003. The replacement of wetland vegetation by reed canarygrass (Phalaris arundinacea). Ecological Restoration 21:116-119. http://dx.doi.org/10.3368/er.21.2.116.

Maurer, D.A. and J.B. Zedler. 2002. Differential invasion of a wetland grass explained by tests of nutrients and light availability on establishment and clonal growth. Oecologia 131:279-288. http://dx.doi.org/10.1007/s00442-002-0886-8.

Merigliano, M.F. and P. Lesica. 1998. The native status of reed canarygrass (*Phalaris arundinacea* L.) in the inland Northwest, U.S.A. Natural Areas Journal 18:223-230.

Miller, R.C., and , J.B. Zedler. 2003. Responses of native and invasive wetland plants to hydroperiod and water depth. Plant Ecology 167:57-69. http://dx.doi.org/10.1023/A:1023918619073.

Molofsky, J., S.L. Morrison, and C.J. Goodnight. 1999. Genetic and environmental controls on the establishment of the invasive grass Phalaris arundinacea. Biological Invasions 1:181-188. http://dx.doi.org/10.1023/A:1010091227793.

Moore, S., D. Ward, and B. Aldrich. 2000. Transplanting large trees for reed canarygrass control. Reed Canarygrass Working Group Conference, Washington State Department of Transportation, USDA Natural Resources Conservation Service, and Society for Ecological Restoration – Northwest Chapter, March 15, 2000, Olympia, Washington.

Morrison, S.L. and J. Molofsky. 1998. Effects of genotypes, soil moisture, and competition on the growth of an invasive grass, Phalaris Arundinacea (reed canary grass). Canadian Journal of Botany 76:1939-1946. http://dx.doi.org/10.1139/b98-157.

Mulhouse, J.M. and S.M. Galatowitsch. 2003. Revegetation of prairie pothole wetlands in the mid-continental US: twelve years post-reflooding. Vegetatio 169:143-159.

Naglich, F. 2000. Using excavation and conifer establishment in managing reed canarygrass. Reed Canarygrass Working Group Conference, Washington State Department of Transportation, USDA Natural Resources Conservation Service, and Society for Ecological Restoration – Northwest Chapter, March 15, 2000, Olympia, Washington.

NRCS. 2005. *Pullman Plant Materials Center Progress Report of Activities-2004*. USDA, NRCS, Pullman Plant Materials Center, Pullman, Washington.

Ogle, D.G. and J.C. Hoag. 2000. *Stormwater Plant Materials: a Resource Guide (Guidelines for Plant Selection, Establishment, and Maintenance for Storm Water Best Management Practices)*. Prepared By USDA-Natural Resources Conservation Service, Aberdeen Plant Materials Center. City of Boise Public Works Department, Boise, Idaho.

Perry, L.G. and S.M. Galatowitsch. 2002. Lowering nitrogen availability may control reed canarygrass in restored prairie pothole wetlands (Minnesota). Ecological Restoration 20:60-61.

Perry, L.G. and S.M. Galatowitsch. 2003. A test of two annual cover crops for controlling Phalaris arundinacea invasion in restored sedge meadow wetlands. Restoration Ecology 11:297-307. http://dx.doi.org/10.1046/j.1526-100X.2003.00174.x.

Perry, L.G., S.M. Galatowitsch, and C.J. Rosen. 2004. Competitive control of invasive vegetation: a native wetland sedge suppresses *Phalaris arundinacea* in carbon-enriched soil. Journal of Applied Ecology 41:151-162.
https://doi.org/10.1111/j.1365-2664.2004.00871.x

Perry, L.G., and S.M. Galatowitsch. 2004. The influence of light availability on competition between Phalaris arundinacea and a native wetland sedge. Vegetatio 170:73-81. http://dx.doi.org/10.1023/b:vege.0000019019.78909.f0.

Pick, T., P. Husby, W. Kellogg, B. Leinard, and R. Apfelbeck. 2004. *Riparian Assessment Using the NRCS Riparian Assessment Method*, Environment Technical Note Number MT-2, USDA NRCS, Bozeman, MT.

Reinhardt, C. and S.M. Galatowitsch. 2004. *Best Management Practices for the Invasive Phalaris arundinacea L. (Reed Canary Grass) in Wetland Restorations*. University of Minnesota Report, published by Minnesota Department of Transportation, Research Services Section.

Reinhardt Adams, C. and S.M. Galatowitsch. 2005. Phalaris arundinacea (reed canary grass): Rapid growth and growth pattern in conditions approximating newly restored wetlands. Ecoscience 12:569-573. http://dx.doi.org/10.2980/i1195-6860-12-4-569.1

Reinhardt Adams, C. and S. M. Galatowitsch. 2006. Increasing the effectiveness of Phalaris arundinacea L. (reed canarygrass) control in wet meadow restorations. Restoration Ecology 14(3):440-450. http://dx.doi.org/10.1111/j.1526-100X.2006.00152.x.

Ringold, P.L., K.M. McNyset, T. Magee, and J. Van Sickle1. 2007. Are invasive riparian plants associated with reduced biotic condition of fauna in western US streams? Abstracts, Ecological Society of America/Society for Ecological Restoration Joint Meeting, San Jose, California, August 5-10, 2007.

Schooler, S.S., P.B. McEvoy, and E.M. Coombs. 2006. Negative per capita effects of purple loosestrife and reed canary grass on plant diversity of wetland communities. Diversity and Distributions 12(4):351-363. http://dx.doi.org/10.1111/j.1366-9516.2006.00227.x

Schröder, H.K., H.E. Andersen, and K. Kiehl. 2005. Rejecting the mean: estimating the response of fen plant species to environmental factors by non-linear quantile regression. Journal of Vegetation Science 16:373-382. http://dx.doi.org/10.1111/j.1654-1103.2005.tb02376.x.

Stannard, M. and W. Crowder. 2001. *Biology, History, and Suppression of Reed Canarygrass (Phalaris arundinacea)*. USDA, NRCS, Plant Materials Technical Note 43, Pullman Plant Materials Center, Pullman, Washington.

Stannard, M. and W. Crowder. 2002. *Plant Guide: Reed canarygrass (Phalaris arundinacea L.)*. USDA, NRCS, Pullman Plant Materials Center, Pullman, Washington.

Tu, M. 2004. *Reed Canarygrass (Phalaris arundinacea L.) Control and Management in the Pacific Northwest*. The Nature Conservancy Wildland Invasive Species Team, The Nature Conservancy, Oregon Field Office.

Weddell, B.J. 2002. The causes and consequences of loss of a culturally significant resource: seasonally moist meadows in the Palouse region. Appendix 4 in G. Servheen, P. Morgan, B. Weddell, P. Gessler, and P. McDaniel. *Wetlands of the Palouse Prairie: Historical Extent and Plant Composition*. Unpublished report funded by the USDI Environmental Protection Agency.

Werner, K.J. and J.B. Zedler. 2002. How sedge meadow soils, microtopography, and vegetation respond to sedimentation. Wetlands 22(3):451-466. http://dx.doi.org/10.1672/0277-5212(2002)022[0451:HSMSMA]2.0.CO;2.

Wetzel, P.R. and A.G. van der Valk. 1998. Effects of nutrient and soil moisture on competition between Carex stricta, Phalaris arundinacea, and Typha latifolia. Plant Ecology 138:179-190. http://dx.doi.org/10.1023/A:1009751703827.