Table 1. Mean percent cover of understory species in control (C) and seeded (S) plots within a ponderosa pine restoration site in Grand Canyon-Parashant National Monument, Arizona. Measurements in 2005 were before the seeding treatment.

	2005		2006		2007	
Species	C	S	C	S	C	S
grasses:						
blue grama (*Bouteloua gracilis*)	0.001	0.005	0.004	0.003	0.005	0.04
Indian ricegrass (*Achnatherum hymenoides*)	0	0	0	0.01	0.001	0.07
western wheatgrass (*Pascopyrum smithii*)	0	0	0.005	0.21	0.03	0.79
forbs:						
desert globemallow (*Sphaeralcea ambigua*)	0	0	0	0	0	0
Lewis flax (*Linum lewisii*)	0	0	0	0.001	0	0
silvery lupine (*Lupinus argenteus*)	0.56	0.17	1.31	0.16	0.41	0.03
all seeded species	0.56	0.18	1.32	0.38	0.45	0.93
all species	3.98	3.61	6.12	4.04	6.73	5.53

project may be larger in the future. With these encouraging results, gathered during a series of drier than average years, we recommend that managers consider seeding with native species as part of ecological restoration or postfire rehabilitation treatments in southwestern ponderosa pine forests to enhance recovery of the understory community. We further recommend that the timing of seeding should depend on the particular climatic patterns at the site. The Mount Trumbull area lies at the northern edge of the region affected by the southwestern monsoon, and thus seeding in the fall probably worked better for us than if we had attempted to seed earlier and relied on summer rains.

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References

Springer, J.D. and D.C. Laughlin. 2004. Seeding with natives increases species richness in a dry ponderosa pine forest (Arizona). *Ecological Restoration* 22:220–221.

Springer, J.D., A.E.M. Waltz, P.Z. Fulé, M.M. Moore and W.W. Covington. 2001. Seeding versus natural regeneration: A comparison of vegetation change following thinning and burning in ponderosa pine. Pages 66–73 *in* USDA Forest Service Proceedings RMRS-P-22.

Western Regional Climate Center (WRCC). 2008. Nixon Flats Arizona. www.wrcc.dri.edu/cgi-bin/rawMAIN.pl?azANIX

Reed Canarygrass Invasions Alter Succession Patterns and May Reduce Habitat Quality in Wet Meadows

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Wet meadows are among the most endangered plant communities in the upper Midwest, with less than 1% of their former extent remaining. Conversions to agriculture, wildfire suppression, hydrological disturbance, and encroachment by invasive species have all contributed to the loss of acreage. Wet meadows may be important to wildlife species of greatest conservation need, such as grassland birds, because of their structural similarity to tallgrass prairie and ecotonal position between wetland and upland habitat, and may function as refugia for these species during drought years. Reed canarygrass (*Phalaris arundinacea*, hereafter RCG) is extensively distributed throughout North American wetlands, yet few data are available to assist land managers in prioritizing resources for RCG suppression programs. The purposes of this study were to characterize factors associated with the distribution and abundance of RCG and related wet meadow vegetation, and to apply the state and transition concept of plant succession to RCG invasions in order to identify where RCG suppression and native species recovery are more likely to be successful and cost effective.

State and transition models of plant community dynamics are used in range management to predict responses to disturbances and management activities (Stringham et al. 2003). The basic underlying premise is that vegetation assemblages persist in one state (e.g., wet meadow) until a disturbance, species invasion, or any combination of perturbations results in a transition to an alternative successional state. For example, wildfire suppression results in sedge meadow converting to shrub carr. These models predict that state transitions occur after a threshold boundary in an ecological process is crossed, though threshold values can be difficult to pinpoint quantitatively. Species distribution patterns can be monitored and descriptively quantified to detect state transitions and separate them from phase transitions (e.g., a wet meadow dominated by sedges and native cool-season grasses compared to a wet meadow dominated by forbs; both phases are temporary variations of the same successional stage). Friedel (1991) predicted that reversing a state transition after a threshold has been breached would require substantial inputs of financial resources and energy and may not be possible on a time scale practical to management. However, experimental research has not thoroughly evaluated many of the model predictions, and reversals might be possible in some types of sites, such as RCG monocultures possessing intact native seed banks or where chronic background disturbances can be affordably abated.

In 2002, we conducted a descriptive study of 13 wet meadow sites in southwestern Wisconsin and southeastern Minnesota, which characterized a gradient of RCG dominance. Sites ranged in area from 4.5 to 16.5 ha and RCG cover ranged from 2.5% to 90%. We measured a total of 26 environmental variables (10 vegetation descriptors and 16 soil nutrients). Vegetation was sampled within 12.6-m^2 circular plots placed at 50-m intervals along transects within each site. We visually estimated percent cover of all species with a modified relevé technique, then calculated species density (richness), percent forbs, percent shrubs, percent trees, percent litter, and a RCG dominance index (the percent cover of RCG divided by the sum of percent covers of the five most dominant species). We also measured vegetation height, litter depth, and number of potential singing perches for grassland birds (sturdy stems $\geq$ 10 cm taller than average vegetation height). Soils were sampled at three to nine randomly selected sample points per site, depending on site size, representing a 1.5-ha sampling interval. Soil plugs were 6-cm in diameter and taken to a depth of 10 cm. We analyzed soil samples for total nitrogen, ammonium-N, nitrate-N, total phosphorous, available phosphorous, potassium, calcium, magnesium, sulfur, boron, zinc, manganese, iron, copper, aluminum, and sodium. Vegetation and soil nutrient data were analyzed with nonmetric multidimensional scaling (NMDS), a multivariate technique used to explore relationships between plant community composition and environmental gradients (Schiffman et al. 1981), using PC-ORD, version 2.0.

The first two axes of the NMDS ordination explained 94.4% of the variation in the data set (Figure 1). The 13 wet meadow sites formed two distinct clusters along axis 1, "native" and RCG-dominated sites. Mean RCG coverage was 22% in the former and 78% in the latter. Several common wet meadow taxa (*Carex, Aster, Eupatorium*) were negatively correlated with axis 1, indicating their abundance was inversely related to RCG abundance. Of the 146 species included in the ordination matrix, 113 had higher percent coverage in native sites, while the remaining 33 species were more abundant in RCG-dominated sites.

Revealing a well-established pattern for this invasive grass, RCG percent cover was negatively correlated with species density (Pearson's r = -0.91, $p < 0.001$) and positively correlated with mean vegetation height ($r = 0.70$, $p = 0.007$), indicating that RCG invasions can alter vegetation structure in addition to species composition. Sites classified as native wet meadows offered 128% more potential bird singing perches than RCG-dominated sites. Canopy factors and competition for light (Lindig-Cisneros and Zedler 2002) may have influenced RCG distribution patterns: reed canarygrass cover was negatively correlated with percent cover of forbs ($r = -0.92$, $p < 0.001$) and percent cover of shrubs ($r = -0.63$, $p = 0.02$). Similar correlation patterns were observed when the RCG dominance index was substituted for RCG percent cover during analysis, so only the latter are reported here.

Reed canarygrass distribution patterns were positively correlated with soil copper ($r = 0.71$, $p = 0.007$), zinc ($r = 0.57$, $p = 0.04$), and aluminum ($r = 0.62$, $p = 0.02$). These metals are associated with automotive break pad wear and are carried into wetlands by rainwater runoff and soil concentrations of the first two can be used to predict RCG expansions in sedge meadows (Stiles et al. 2008). RCG is more tolerant of these metals than most other wetland species, and is sometimes used for phytoremediation of mine tailings (Hansel et al. 2001). Reed canarygrass distribution was also positively correlated with soil sodium ($r = 0.77$, $p = 0.002$). Widespread use of road salt has been implicated in the spread and dominance of several other invasive taxa, although the present study was not designed to confirm this effect on RCG invasions (but see Stiles et al. 2008).

Reed canarygrass distribution patterns were weakly correlated with soil exchangeable potassium ($r = 0.28$, $p = 0.06$), but not with total nitrogen, nitrate-N, ammonium-N, available phosphorus, or total phosphorus. However, levels of these soil nutrients were generally high within all sites, in contrast to an earlier study that manipulated soil nutrient levels to create gradients and demonstrated strong correlations between RCG abundance and nutrient enrichment (Green and Galatowitsch 2002).

NMDS ordination showed two discrete clusters rather than continuous separation of wet meadow sites along the gradient of RCG abundance, indicating that RCG invasion may result in a successional state transition from a pre-transitional "native" wet meadow to a post-transitional RCG-dominated meadow (Figure 1), whereas a continuous distribution of sites would suggest more of a "phase" shift in species composition. While we advocate caution when inferring management implications from these results, we propose prioritizing reed canarygrass management

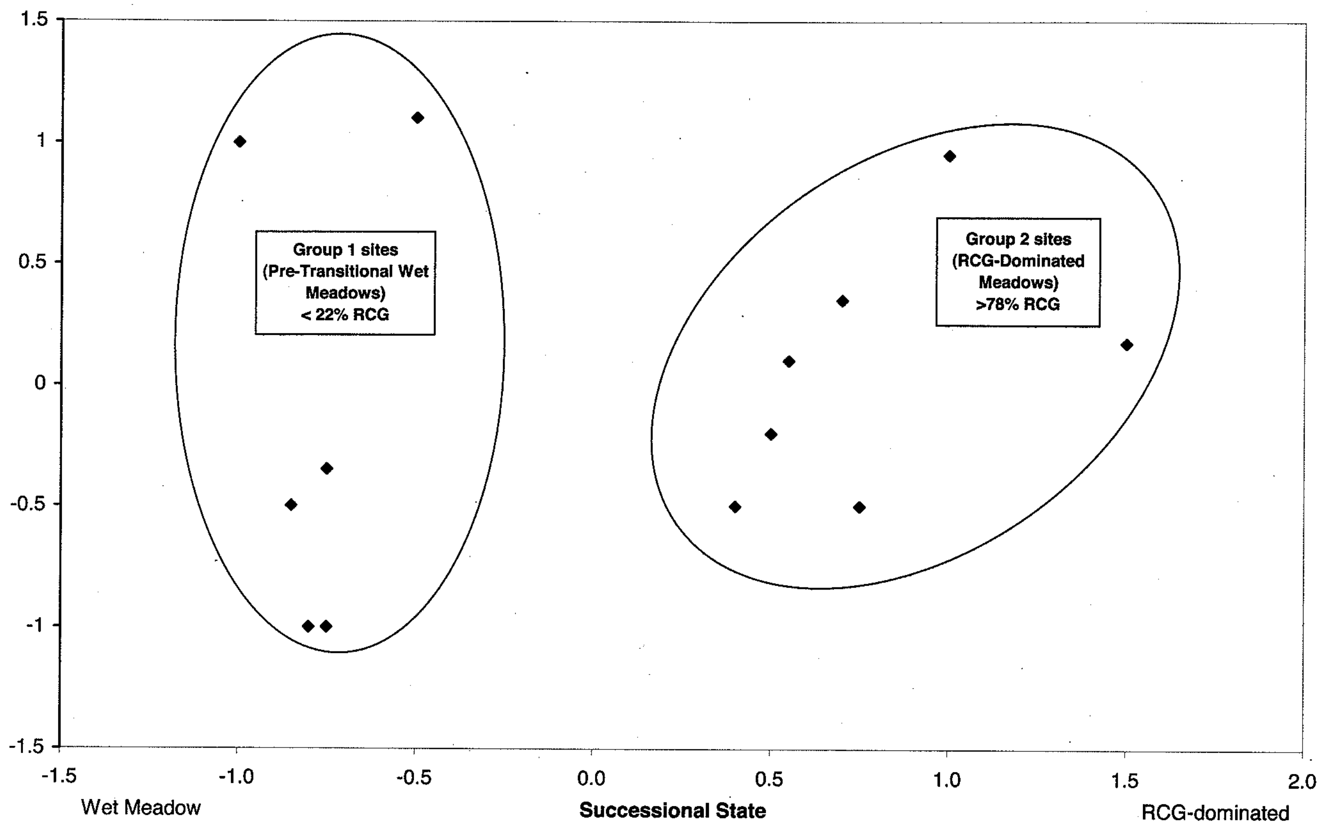

Figure 1. NMDS ordination of 13 wet meadows in Wisconsin and Minnesota. Reed canarygrass (RCG) percent cover is positively correlated with axis 1 (Eingenvalue *R* = 0.93). Group 1 (*n* = 6) consists of "native" wet meadow sites; Group 2 (*n* = 7) consists of RCG-dominated wet meadow sites.

efforts on pretransitional wet meadows (i.e., sites with ≤ 25% RCG cover) until cost-effective and reliable suppression and revegetation tactics can be developed for RCG-dominated wet meadows.

We found that RCG had a tendency to be less abundant where shrubs and forbs were abundant. This finding supports the recommendations of Maurer et al. (2003), who advised coupling RCG suppression tactics with the establishment of a closed native species canopy to provide shade against RCG resurgence.

Although our results indicate that fewer potential grassland bird singing perches accompany RCG invasion and dominance, Kirsch et al. (2007) studied these same sites and reported that common wet meadow breeding songbirds did not avoid sites with high RCG cover. Furthermore, ongoing research at Mankato State University suggests that RCG does not affect nesting success of red-winged blackbirds (*Agelaius phoeniceus*) (Emily Hutchins, graduate student, pers. comm.). However, habitat specialist species may be more adversely affected: Kapfer (in prep.) found that Butler's Garter snake (*Thamnophis butleri*) tended to avoid RCG-dominated wetlands, whereas common garter snake (*Thamnophis sirtalis*) was regularly abundant in a variety of wetland successional states. Clearly, additional research is required to thoroughly evaluate the effect of RCG on habitat quality.

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References

Friedel, M.H. 1991. Range condition assessment and the concept of thresholds: A viewpoint. *Journal of Range Management* 44:422–426.

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.

Hansel, C.M., S. Fendorf, S. Sutton and M. Newville. 2001. Characterization of Fe plaque and associated metals on the roots of mine-waste impacted aquatic plants. *Environmental Science and Technology* 35:3863–3868.

Kirsch, E.M., B.R. Gray, T.S. Fox and W.E. Thogmartin. 2007. Breeding bird territory placement in riparian wet meadows in relation to invasive reed canary grass (*Phalaris arundinacea* L.). *Wetlands* 27:755–766.

Lindig-Cisneros, R. and J.B. Zedler. 2002. Relationships between canopy complexity and germination microsites for *Phalaris arundinacea* L. *Oecologia* 133:159–167.

Maurer, D.A., R. Lindig-Cisneros, K.J. Werner, S. Kercher, B. Miller and J.B. Zedler. 2003. The replacement of wetland vegetation by reed canarygrass (*Phalaris arundinacea*). *Ecological Restoration* 21:116–119.

Schiffman, S.S., M.L. Reynolds and F.W. Young. 1981. *Introduction to Multidimensional Scaling: Theory, Methods and Application.* New York: Academic Press.

Stiles, C.A., B. Bemis and J.B. Zedler. 2008. Evaluating edaphic conditions favoring reed canary grass invasion in a restored native prairie. *Ecological Restoration* 26:61–70.

Stringham, T.K., W.C. Krueger and P.L. Shaver. 2003. State and transition modeling: An ecological process approach. *Journal of Range Management* 56:106–113.

Impacts of Non-native Plant Removal on Vertebrates along the Middle Rio Grande (New Mexico)

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The Middle Rio Grande and its riparian forest in central New Mexico are the focus of restoration activities to reverse or lessen negative anthropogenic impacts. The riparian forest is the largest gallery cottonwood (*Populus deltoides*) forest in the Southwest (Hink and Ohmart 1984). Historically, the river was free to meander across the floodplain, creating a dynamic system in which riparian vegetation establishment on riverbanks alternated with periods of scouring floods (Crawford et al. 1993). The establishment of non-native invasive plants has compounded the impacts of an altered hydrology, which has been changed through channelization and water diversion. Non-native saltcedar (*Tamarix* spp.) and Russian olive (*Elaeagnus angustifolia*) and the accumulation of woody debris along the river have increased the risk of wildfire and suppressed native seed germination (Howe and Knopf 1991, Busch and Smith 1995).

Fires in the riparian forest increase the risk of losing native habitat that has not historically experienced fire. Land managers have a suite of techniques available to reduce the risk of fire by removing non-native plants and fuels in the riparian forest. Biologists are interested in understanding how these management activities can impact other components of the Middle Rio Grande riparian ecosystem, such as native flora and fauna.

In 2000 the USDA Forest Service Rocky Mountain Research Station began monitoring amphibians, reptiles, birds, and bats in riparian forests dominated by Rio Grande cottonwood (*Populus deltoides wislizenii*) and a non-native plant midstory (Figure 1a) to investigate the impacts of restoration treatments on vertebrate species. We established three control and nine experimental 20-ha sites along 140 km of riparian forest. Treatments to remove non-native plants and fuels began in 2003 and were completed in 2005 (Figure 1b). In all three treatments, non-native plants were removed by chainsaw and Garlon (triclopyr) herbicide was applied to stumps. In the second treatment, burning slash piles followed plant removal. In the third treatment, planting native tree and shrub seedlings (247 per hectare) followed plant removal. The cottonwood overstory remained unchanged after treatments, whereas midcanopy cover decreased in experimental sites when non-native plants were removed.

For seven years at each site we gathered data on herpetofauna, birds, and bats. We utilized arrays of drift fences and pitfall and funnel traps to capture and release amphibians and reptiles (Bateman et al. 2008). We followed nomenclature from Crother (2008) for herpetofauna names. We recorded bird species, and distance from observer, at point count stations and monitored nests during the breeding season (Finch and Hawksworth 2006). Ultrasonic detection systems (Anabat detectors, Titley Corporation, Australia) recorded bat echolocation activity once a week during the summer (Chung-MacCoubrey and Bateman 2006).

Restoration treatments appear to be beneficial or non-damaging to lizard species and may have a positive effect on bat foraging. Treatments appear to negatively affect densities of some bird species that nest in midstory vegetation. We did not detect changes in response to restoration treatments in relative abundances or densities of other taxa, such as toads (*Anaxyrus* spp.), snakes, or other avian nesting guilds.

Two lizard species, New Mexico whiptails (*Aspidoscelis neomexicana*) and prairie lizards (*Sceloporus consobrinus*), increased after experimental treatments. Chihuahuan spotted whiptails (*A. exsanguis*), desert grassland whiptails (*A. uniparens*), and side-blotched lizards (*Uta stansburiana*) were either positively associated with the posttreatment forest or negatively associated with the pretreatment forest (Bateman et al. 2008). Lizard abundance may have increased after treatments because the more open midstory may provide more basking opportunities as greater light penetrates to the understory.

Birds nesting in the lower two-thirds of vegetation decreased after treatments for the midstory nesters, black-chinned hummingbird (*Archilochus alexandri*), mourning dove (*Zenaida macroura*), and black-headed grosbeak (*Pheucticus melanocephalus*), and one ground shrub nester,