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SUPPRESSION OF REED CANARYGRASS FOR LARGE-SCALE FLOODPLAIN FOREST RESTORATION ACROSS FOUR SITES IN SOUTHEAST MINNESOTA, USA

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SUPPRESSION OF REED CANARYGRASS FOR LARGE-SCALE FLOODPLAIN FOREST RESTORATION ACROSS FOUR SITES IN SOUTHEAST MINNESOTA, USA

By William R. Kiser

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ABSTRACT

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Experiments can provide insight on whether invasive plant dominance is caused by superior competitive ability (driver) or by environmental changes that facilitate plant invasion (passenger). Reed canarygrass (*Phalaris arundinacea*, hereafter RCG) displaces native plants and forms near-monocultures in North American wetlands. In the Upper Mississippi River (UMR) system, floodplain forests are negatively impacted by RCG invasion. We tested two RCG control techniques on a reforestation project at four sites in SE Minnesota. Treatments consisted of (1) applying glyphosate (Rodeo) herbicide and (2) mulching followed by applying sulfometuron methyl (Oust XP) herbicide. Treatments were applied in Fall 2016. We monitored herbaceous plant response and RCG performance over the 2017 and 2018 growing seasons. We also calculated the number of days flooded in each plot for the 2017 growing season using linear interpolation of river gauge data. Both treatment methods significantly reduced RCG performance relative to controls during the 2017 growing season; however, RCG performance in treatment plots was similar to control plots during the 2018 growing season. Herbaceous plants increased in species richness and cover relative to control plots, although volunteer plant diversity varied among sites. These results indicate that follow-up herbicide applications are necessary to control RCG to facilitate tree establishment. Further, flooding and lack of native propagules may be factors in RCG invasions. Our results indicate that RCG may behave as a driver in some parts of the UMR and as a passenger in parts of the UMR that are more affected by hydrologic alteration.

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INTRODUCTION

Biological invasion from non-indigenous plants is a prominent threat to ecological communities. Most invasions are characterized by a non-indigenous plant negatively affecting native species through competitive exclusion (Catford et al., 2012; Maurer et al., 2003; Simberloff & Von Holle, 1999). Negative effects on native plants are often associated with loss of biodiversity as native species richness and cover are reduced (Catford et al., 2012; Fink & Wilson, 2011). Ecological restoration is a management tool to undo the damage of invasive plants. In such cases, management efforts focus on removing the invader (Lindenmeyer et al., 2015), restoring natural ecosystem processes (e.g. fire), reintroducing native plant propagules (MacDougall & Turkington, 2005, 2007), or a combination of these approaches (Fink & Wilson, 2011; Foster & Wetzel, 2005; Thomsen, Brownell, Groshek, & Kirsch, 2012).

Invasive plants characterized as “drivers” dictate community structure through superior competitive ability, whereas “passenger” species invade a site in response to environmental changes that negatively affect the community (Bauer, 2012; Catford, 2016; Lindenmeyer et al., 2015; MacDougall & Turkington, 2005). While the drivers versus passengers conceptual framework is a useful starting point, the dynamics of plant invasion likely occur over a continuum between these two concepts (Bauer, 2012). For example, an invasive plant may initially enter a system as the result of altered environmental conditions but then may further affect conditions in a way that increases its own fitness (Suding, Gross, & Houseman, 2004; Zedler, 2009). In such a scenario, an

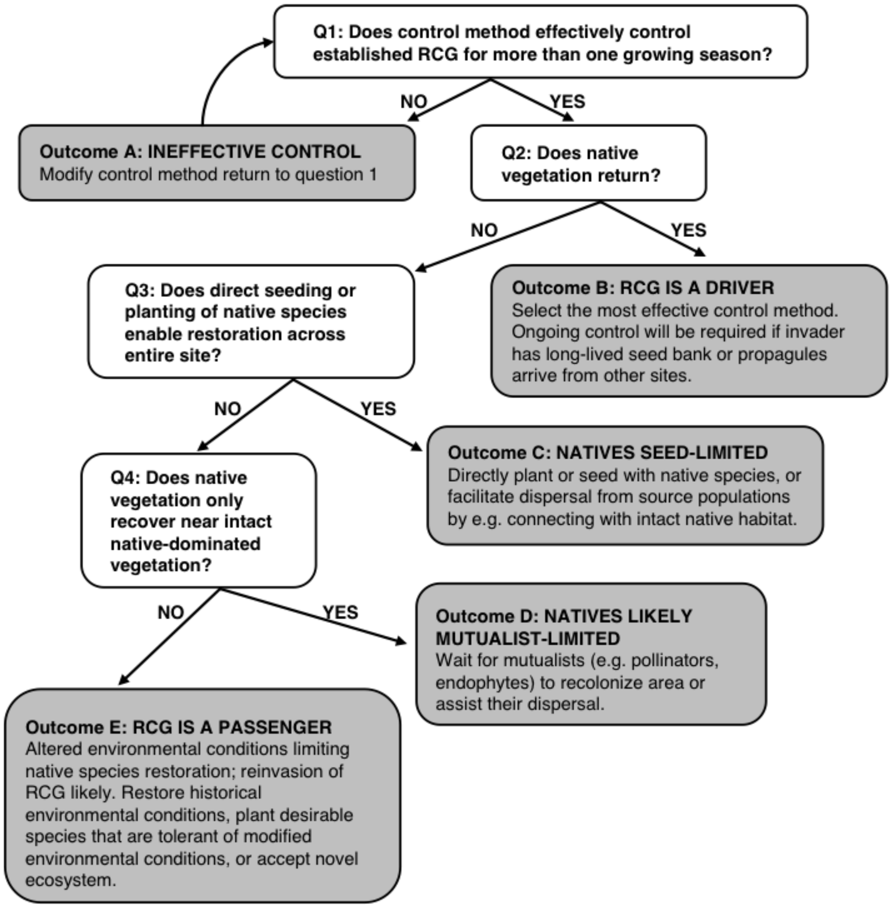

Figure 1. Process flow chart for evaluating whether reed canarygrass (*Phalaris arundinacea*) is a passenger or driver. Adapted from Catford (2016) and Lindenmeyer et al. (2015).

invader is behaving first as a passenger and then as a driver (Bauer, 2012; Lindenmeyer et al., 2015). Studies on the restoration of communities impacted by invasive plants can provide insights into the underlying dynamics of the invasion (Bauer, 2012; Catford, 2016; Lindenmeyer et al., 2015; MacDougall & Turkington, 2005). By following the investigative process depicted in Figure 1 (Catford, 2016), studies on the restoration of sites impacted by invasive plants can help determine whether the invader is the cause of ecological change or an outcome of it.

Our study system is the Upper Mississippi River (UMR) floodplain forest, which is negatively impacted by the invasive grass *Phalaris arundinacea* (reed canarygrass, hereafter RCG). While RCG can grow in a wide range of habitats, RCG invasion threatens a wide variety of North American wetlands due to its ability to displace native plants and form near-monocultures (Barnes, 1999; Lavergne & Molofsky, 2004, 2006; Price, Spyreas, & Matthews, 2018; Zedler & Kercher, 2004). RCG is usually 1-2 m in height, long-lived, and can maintain photosynthetic ability when roots are submerged (Lavergne & Molofsky, 2004). Early-season growth through rapid height gain and horizontal spread results in RCG's ability to shade out competitors (Lavergne & Molofsky, 2004). RCG reproduces both through vigorous annual seed production as well as vegetatively, via underground rhizomes (Lavergne & Molofsky, 2006). The accumulation of litter in areas dominated by RCG can delay the spring emergence of wetland plants by slowing the warming of soil; further, RCG has been shown to have vigorous growth, despite delayed spring emergence, where there are greater amounts of litter, suggesting a positive feedback mechanism (Kaproth, Eppinga, & Molofsky, 2013; Zedler, 2009). Although it is accepted that RCG is native to North America (Jakubowski, Casler, & Jackson, 2013), repeated human introductions of Eurasian genotypes and self-incompatibility have resulted in the loss of native genotypes; as such, North American populations of RCG are genetically similar to European populations and exhibit a high degree of within-population genetic diversity (Jakubowski et al., 2013; Lavergne & Molofsky, 2007; Nelson & Anderson, 2015; Nelson, Anderson, Casler, & Jakubowski, 2014). High genetic diversity makes RCG adaptable to a wide range of habitat types and contributes to its invasiveness (Brodersen, Lavergne, & Molofsky, 2008; Nelson & Anderson, 2015).

Physical, chemical, and biological control methods have been tested for their ability to suppress RCG in restoration settings (reviewed in Lavergne & Molofsky, 2006). Combinations of tillage, herbicide, fire, and seeding with native species have been demonstrated as potentially effective; however, results have been mixed, thus more research is needed to develop integrated approaches to suppress RCG for restorations (Foster & Wetzel, 2005; Hovick & Reinartz, 2007; Lavergne & Molofsky, 2006; Maurer et al., 2003; Miller-Adamany, Baumann, & Thomsen, *in press*; Thomsen et al., 2012). Herbicide is often used because it can be applied over large areas (reviewed in Lavergne & Molofsky, 2006); Reinhardt Adams and Galatowitsch (2006) showed that late-August to late-September herbicide applications on RCG are nearly twice as effective as spring applications. Negative effects on desirable plants can be minimized by using selective herbicides or applying chemical during periods when other species are dormant and RCG is actively growing (Hovick & Reinartz, 2007; Miller-Adamany et al., *in press*; Thomsen et al., 2012). The root systems of RCG have both active and dormant rhizomatous buds, and these dormant buds can be activated following a suppression treatment, so multiple control treatments are often needed (Annen, 2010; Bahm, Barnes, & Jensen, 2014). Reducing RCG canopy cover while promoting the growth of shade-producing native plants is a promising method to restore native communities (Hovick & Reinartz, 2007; Reinhardt Adams, Kauth, & Sorenson, 2011; Thomsen et al., 2012). Miller-Adamany et al. (*in press*) had positive results in restoring floodplain forest using an initial October herbicide application, followed by planting floodplain forest trees and shrubs, with a second late-fall herbicide application after planted stock had senesced. By disrupting RCG growth long enough to allow tree and shrub establishment, reforestation is a potential restoration opportunity for RCG-dominated areas (Annen, Kirsch, & Tyser, 2008; Foster & Wetzel, 2005; Hovick & Reinartz, 2007; Merriman, 2014; Miller-Adamany et al., *in press*; Thomsen et al., 2012).

RCG has increasingly become a major issue in UMR floodplain forest (Romano, 2010); despite little mention by Curtis (1959), RCG is now widespread in the UMR and is common in the understory of existing floodplain forest (De Jager, Rohweder, & Hoy, 2017; Guyon & Battaglia, 2018; Johnson, Mudrak, & Waller, 2014). Canopy gaps created by wind-throw or upheaval during flooding are often invaded by RCG (Romano, 2010; Thomsen et al., 2012). RCG outcompetes young trees; tree seedling density has been shown to be inversely proportional to RCG cover (Cogger, De Jager, Thomsen, & Reinhardt Adams, 2014; Romano, 2010), and tree seedlings do not survive in untreated areas of RCG (Reinhardt Adams et al., 2011; Thomsen et al., 2012). As such, RCG has been characterized as "likely" the most damaging of all invasive plant species in the UMR floodplain forest system (Guyon, Deutsch, Lundh, & Urich, 2012). Floodplain forest losses in the UMR are estimated to be 50-75% due to conversion to agricultural lands, urbanization, and permanent inundation due to the lock and dam system (De Jager & Rohweder, 2017; Knutson & Klaas, 1998), making them a high priority for restoration among land managers (Guyon et al., 2012).

As part of a multi-partner research project to restore RCG-dominated floodplains of the UMR, our project focuses on the herbaceous plant and volunteer tree seedling response to two methods of RCG suppression (Questions 1 & 2, Fig. 1). While the majority of restoration research projects are carried out over smaller scales (e.g. plot sizes less than 30 m^2), using the results from these studies to scale up restoration efforts may be difficult (Kettenring & Reinhardt Adams, 2011; Lamb, 2018). Few studies have tried to apply small-scale results to larger scales (Kettenring & Reinhardt Adams, 2011); as such, one of our major objectives is to test the demonstrated methods of Miller-Adamany et al. (*in press*) at the restoration-scale using larger plot sizes (2023 m^2 versus 36 m^2) on multiple sites in the UMR region. Furthermore, for successful restoration, understanding the target setting for restoration efforts is necessary (Miller et al., 2017), and yearly climactic variations during restoration can have long-term influence on community composition and restoration success (Groves & Brudvig, 2018; Healy, Rojas, & Zedler, 2015; Kettenring & Reinhardt Adams, 2011). Understanding the short- and long-term influence of flooding on UMR floodplains is important to future restoration, so we completed a spatial analysis that examined the potential effect of flooding on the four research sites using GIS data and historical water elevation gauge data from the Mississippi River and applicable tributaries. Flood effects were analyzed for both the 2017 growing season as well as long-term flooding patterns. We designed this field experiment to investigate the following questions:

a. How effective are Rodeo™ (glyphosate) and Oust™ herbicides at controlling RCG in UMR floodplains for more than one growing season?
b. What is the practicality of using demonstrated small-scale RCG suppression techniques on large-scale restorations?
c. What are the capacities of natural community assemblages to regenerate passively following RCG suppression in the UMR?

d. What are the effects of 2017 (year effects) versus long-term average flooding (landscape position effects) on plant response to RCG suppression in UMR floodplains?

We studied these questions to determine whether RCG acts as a passenger or driver in the UMR using the framework presented in Figure 1. Questions ‘a’ and ‘b’ relate to the effectiveness of the treatments in the passengers versus drivers framework (Question 1, Fig. 1). Questions ‘c’ and ‘d’ relate to whether native vegetation returns following RCG control (Question 2, Fig. 1). Flooding is an important influence on wetland plant establishment (Kercher & Zedler, 2004) and vegetation dynamics in the UMR (De Jager, Cogger, & Thomsen, 2013; De Jager, Swanson, Strauss, Thomsen, & Yin, 2015; De Jager, Thomsen, & Yin, 2012). In regard to question ‘d,’ we used this analysis to determine if any observed variation in volunteer plant response across plots or sites was due to flooding rather than the capacity of other plants to revegetate passively following RCG suppression.

METHODS

UMR Floodplain Forests

Since the 1850s, the Upper Mississippi River (UMR) has been altered for transportation, flood control, shipping, and water supply; the current lock and dam system was completed in the 1940s to maintain a 2.74-meter (9-feet) channel depth for shipping (Romano, 2010). Despite UMR forest losses due to conversion to agricultural lands, urbanization, and permanent inundation due to the lock and dam system (Knutson & Klaas, 1998), floodplain forest in the UMR is relatively well-represented when compared to other Upper Midwest community types (Curtis, 1959; De Jager & Rohweder, 2017; Johnson & Waller, 2013). Primary overstory tree species in the UMR floodplain include *Acer saccharinum* (silver maple), *Fraxinus pennsylvanica* (green ash), and *Ulmus americana* (American elm); additionally, *Betula nigra* (river birch), *Populus deltoides* (Eastern cottonwood), *Quercus bicolor* (swamp white oak), *Celtis occidentallis* (common hackberry), and *Salix nigra* (black willow) are common (Curtis, 1959; De Jager et al., 2012; Guyon & Battaglia, 2018; MN DNR, 2018b; WI DNR, 2018). Common herbaceous plants include *Laportea canadensis* (Canada nettle), *Boehmeria cylindrica* (smallspike false nettle), *Helenium autumnale* (common sneezeweed), *Urtica dioica* (common nettle), *Carex lupulina* (hop sedge), and *Elymus virginicus* (Virginia wildrye) (Guyon & Battaglia, 2018; WI DNR, 2018).

Historically, flooding was the primary disturbance to the system, and bottomland forests frequently received additions of silt during flood events (Curtis, 1959). Flood tolerance is a major driver of floodplain forest composition (De Jager et al., 2012; Knutson & Klaas, 1998; Romano, 2010; Turner, Gergel, Dixon, & Miller, 2004). Hydrologic alteration of large river systems in the Upper Midwest has changed floodplain forest successional trajectories (Dixon & Turner, 2006) and has resulted in decreased basal area, decreased among-stand diversity, and shifts in tree species composition (Johnson & Waller, 2013). Indeed, forest composition in the UMR has shifted since the late-1800s, resulting in lower tree density, increased dominance of silver maple, decreased representation of early successional species (e.g. sandbar willow, river birch, and swamp white oak), and lower overall diversity (Guyon et al., 2012; Knutson & Klaas, 1998; Romano, 2010).

Floodplain forest systems are important to the UMR for their role in hydrologic cycling, providing a carbon source for river nutrient processing, and supporting wildlife habitat (MN DNR, 2018b; WI DNR, 2018). Due to their longitudinal distribution along river systems, these forests help to provide an important migration corridor for wildlife such as migratory songbirds (Kirsch, Heglund, Gray, & McKann, 2013); as such, land managers seek to restore forest cover to increase wildlife habitat area and connectivity (Guyon et al., 2012; Romano, 2010). Priority wildlife benefitting from restoration include neotropical migratory songbirds, waterfowl, heron and egrets (which nest in forested rookeries), as well as federally-endangered Eastern Massasauga Rattlesnake (*Sistrurus catenatus catenatus*) and Indiana Bat (*Myotis sodalis*; Guyon et al., 2012; MN DNR, 2018b; Romano, 2010; WI DNR, 2018).

Table 1. Location and ownership of four research sites in SE Minnesota (MN).

Site Name	Ownership	Latitude/Longitude	
		°N Lat.	°W Long.
Collischan	MN Department of Natural Resources	44° 35’ 22.5”	92° 36’ 53.2”
Root River	U.S. Fish and Wildlife Service	43° 46’ 59.6”	91° 18’ 59.6”
Whalen	U.S. Fish and Wildlife Service	43° 30’ 25.1”	91° 16’ 42.9”
Whitewater	MN Department of Natural Resources	44° 8’ 58.5”	92° 18’ 7.9”

Study Sites

This study is part of a reforestation project using a randomized-block design of 68 plots across four sites in southeast Minnesota (Table 1; Fig. 2). All plots were established in areas with average cover greater than 95% RCG prior to treatment; these were laid out in 44.8-m by 44.8-m squares (2023 m^2) in a grid configuration, with the exception of Collischan. Plots here were irregularly shaped and boundaries were marked with GPS. We marked all plot corners with 2.1-m lengths of PVC painted at the top to identify plot type.

Treatments were randomly assigned to plots at each site; treatments consisted of Control, application of Rodeo herbicide (active ingredient glyphosate, Dow AgroSciences, Indianapolis, IN), and application of Oust XP herbicide (hereafter Oust, active ingredient sulfometuron methyl, Bayer, Research Triangle Park, NC). Because Oust has a soil-residual mode of action, all Oust plots were mulched prior to herbicide application. Except for Collischan, each site consisted of four Control plots, eight Rodeo plots and eight Oust plots. The Collischan site consisted of two Control, three Rodeo, and

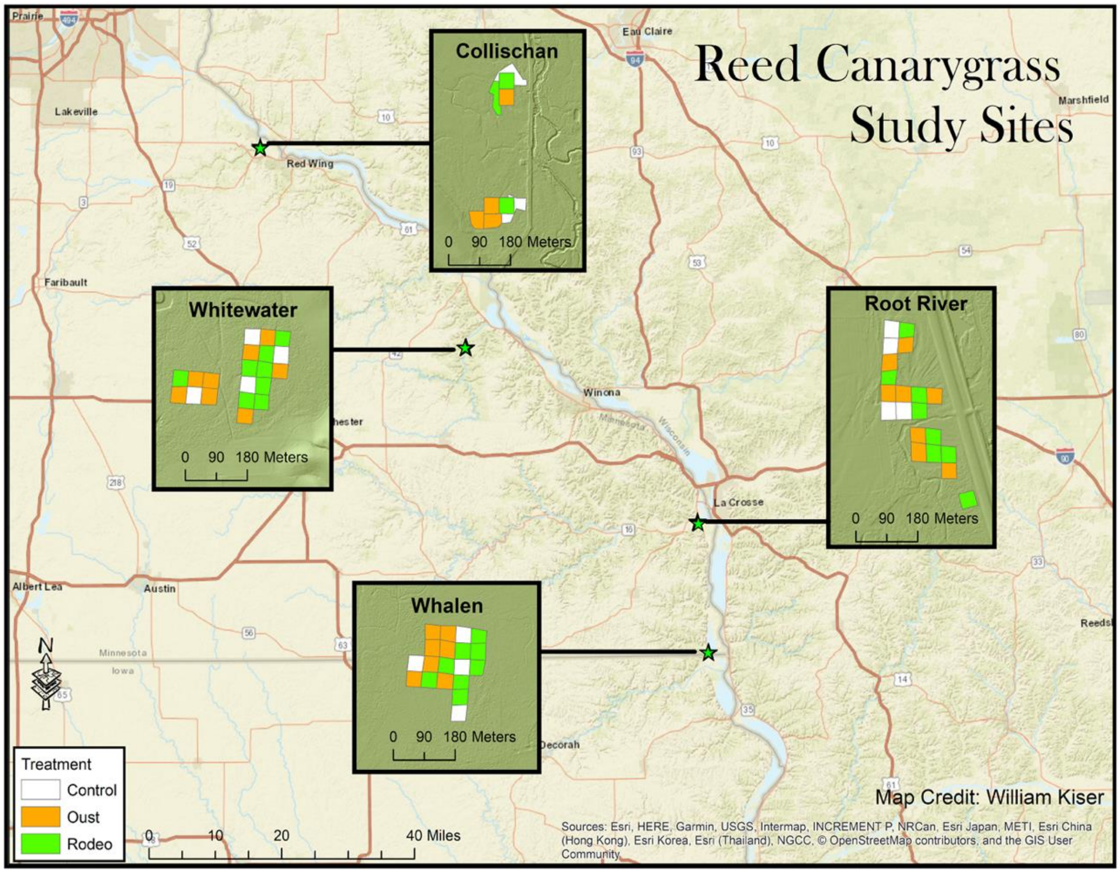

Figure 2. Map of the four research sites in SE Minnesota. Treatments were randomly assigned to plots at each site, and are color coded for Control (white), Rodeo (green), and Oust (orange) plots.

four Oust plots. Initial RCG suppression treatments were applied in Fall 2016. A certified pesticide applicator completed Rodeo treatments October 24 – 28, 2016. Rodeo herbicide solution consisted of 11.4 L (384 ounces) of Rodeo, 0.71 L (24 ounces) of Methylated Seed Oil (MSO), and 272.55 L (72 gallons) of water. The applicator used a boomless sprayer (Warne Chemical and Equipment Co., Rapid City, SD) mounted in a 4-wheel-drive Utility Task Vehicle (Polaris Ranger, hereafter UTV, Polaris, Medina, MN). Plots were sprayed at a rate of approximately 22.7 L (6 gallons) of herbicide solution per plot, which equates to an active ingredient application rate of approximately 4.45 L per hectare (61 fluid-ounces per acre, label-recommended rate is 48 – 72 fluid-ounces per acre). A contractor mulched Oust plots in early November 2016 using a Fecon forestry mulcher (Fecon Incorporated, Lebanon, OH) mounted on a tracked skid steer (ASV Holdings, Inc., Grand Rapids, MN). Oust herbicide solution consisted of 88.6 grams (3.125 ounces) Oust and 283.9 L (75 gallons) water. Following mulching, a pesticide applicator applied Oust solution at a rate of approximately 22.7 L per plot, which equates to an active ingredient application rate of approximately 35.0 grams per hectare (0.5 ounces per acre), using the same equipment as for the Rodeo application.

From April 10, 2017 through May 5, 2017, plots were planted with tree species common to UMR floodplain forest using bare root and container stock. Seedlings were planted at a rate of approximately 208 trees per plot, of which 30 were container stock. Trees were planted with 2.6-m x 2.6-m spacing and planted using dibble bars (bare root stock) and shovels (container stock; see DeLaundreau, 2019). We intended to repeat the herbicide applications in Fall 2017 after trees senesced as in Hovick & Reinartz (2007), Thomsen et al. (2012), and Miller-Adamany et al. (*in press*); however, weather conditions prevented the repeat application.

Plant Response Data Collection

We measured plant response to RCG suppression treatments once per month in June through August 2017 and May through August 2018 (total of seven sampling periods); however, we were unable to complete sampling at Collischan and Whalen in June 2018 due to flooding. Plant data were collected using a 1-m^2 quadrat placed at five fixed intervals of 0 m, 8 m, 16 m, 24 m, and 32 m along a transect. These transects ran from south to north and were located along the midline between two rows of planted trees, beginning 3.05 m from the south boundary of each plot. We laid out transects in a similar manner in the irregularly shaped plots at Collischan; however, in cases where the transect length exceeded plot dimensions, we completed the remainder of the transect length in subsequent rows. Sampling transects within each month were randomly assigned and not repeated throughout the study. We relocated transects in a small number of cases when they fell within strips of RCG that were missed in herbicide treatments.

Plants within each quadrat were identified to species using conventions from Chadde (2013) and we used nomenclature as published on the National Wetland Plant List (USACE, 2018a). Percent cover of RCG, other herbaceous plants, tree seedlings, litter, and bare ground were visually estimated and assigned to cover classes consisting of 0 (not present), 1 (>0 – 5%), 2 (>5 – 25%), 3 (>25 – 50%), 4 (>50 – 75%), 5 (>75 – 95%), and 6 (>95%). We transformed cover class values to the midpoint for data analyses. We measured plant height and litter depth in three standardized points within each quadrat; plant heights were measured for RCG and other herbaceous plants. Plant heights were measured by recording the tallest leaf nearest to the meter stick; that is, they were not manually straightened for measurement. We identified and counted volunteer tree seedlings within each quadrat to species, and then recorded the heights of the three tallest individuals. We counted the number of RCG flowering culms within quadrats in June of both years. Measurements that could not be taken due to flooding were recorded as ‘no data’ and were not considered for data analyses. Quadrat values were averaged for each plot within each sampling month for data analyses.

Data Analyses

We analyzed plant response to restoration treatments in RStudio (RStudio Team, 2018). For RCG cover, RCG height, litter cover, and litter depth, we used mixed-effects linear models using maximum likelihood through the ‘lmer’ function in the ‘lme4’ package (Bates, Mächler, Bolker, & Walker, 2015). Within models for each of these response variables, we nested ‘plot’ within ‘site’ and set this as the random effect while ‘treatment’ was set as a fixed effect. We evaluated the assumption of normality of residuals using the Shapiro-Wilk normality test (‘shapiro.test’ function; R Core Team 2018) and by inspecting residual plots; data not meeting this assumption were rank transformed using the ‘rank’ function (R Core Team, 2018). We assessed the significance of the explanatory variables and interaction terms using Type II Wald chi-square tests through the ‘Anova’ function in the ‘car’ package (Fox & Weisberg, 2011). We completed post-hoc tests using Tukey’s honest significant difference (HSD) test through the ‘lsmeans’ function (Lenth, 2016). Because site was used as a blocking factor, we focused our post-hoc analyses on comparisons among treatments within each site, within sampling month. To evaluate flowering culm production and non-RCG species richness, we used a two-way ANOVA through the ‘aov’ function (R Core Team, 2018) with ‘treatment’ and ‘site’ set as factors. To better meet residual normality assumptions, we transformed species richness and 2017 flowering culms data using the ‘transformTukey’ function (Mangiafico, 2018).

To evaluate the effects of flooding on plant response, we calculated the proportion of the growing season that each plot was flooded for both 2017 (April 1 – July 31, 2017) and over long-term (≥ 6 years, 2004 – 2017) trends using the data summarized in Table 2. Mean plot elevations were calculated from 1-meter Digital Elevation Model (DEM) data using the Zonal Statistics tool in ArcGIS 10.1 (Esri, 2018). We visually inspected plot elevations and used the ArcGIS flow direction tool to determine hydrologic influences of

Table 2. Summary of data used to calculate proportion of growing season days flooded for other cover versus reed canarygrass (RCG) cover multivariate regression analysis. Data were sourced from the U. S. Geological Survey (USGS), U. S. Army Corps of Engineers (USACE), and Minnesota Department of Natural Resources (MN DNR).

Dataset	Description	Source
DEM for each site	1-meter Digital Elevation Model data were used for each site to calculate average elevation in each plot, assess influence of Mississippi River and applicable tributaries on flooding at each site and calculate both 2017 and long-term mean flood days at each site	USGS
2017 flood data	2017 flood data for each site were calculated by using linear interpolation of Mississippi River and applicable tributary water elevation gauge data for April 1, 2017 – July 31, 2017	USGS, USACE, MN DNR
Long-term flood data	Long-term growing season flood data for each site were calculated using linear interpolation of Mississippi River and applicable tributary water elevation gauge data. Due to limitations in data availability, long-term means were calculated for April 1, 2007 – September 30, 2016 at Collischan; April 1, 2008 – September 30, 2013 at Whitewater; April 1, 2004 – September 30, 2013 at Root River; and April 1, 2007 – September 30, 2015 at Whalen.	USGS, USACE, MN DNR

applicable tributaries and the Mississippi River for each site. Through this, we determined that Collischan was not directly influenced by the floodwaters from the Cannon River due to a levee system; rather, floodwaters at Collischan were primarily from the Mississippi. Likewise, we determined floodwaters at Whalen to be primarily from the Mississippi. In contrast, we determined that the floodwaters at Whitewater and Root River sites were exclusively from the tributaries (Whitewater River and Root River, respectively).

Growing season flood days for each plot were determined by using linear interpolation of river gauge data similar to De Jager et al. (2012). This was completed by obtaining historical river gauge data from the Mississippi River and applicable tributaries for each site (MN DNR, 2018a; USACE, 2018b; Table 2). We defined the growing season as April 1 – September 30 (183 days). Site water elevation (SWE) was then calculated as follows:

$$SWE = \frac{(\mathrm{El}_{\mathrm{upstream}} - \mathrm{El}_{\mathrm{downstream}}) \times (\mathrm{Mile}_{\mathrm{site}} - \mathrm{Mile}_{\mathrm{downstream}})}{(\mathrm{Mile}_{\mathrm{upstream}} - \mathrm{Mile}_{\mathrm{downstream}})} + \mathrm{El}_{\mathrm{downstream}}$$

Where:

1. $\mathrm{El}_{\mathrm{upstream}}$ = Upstream gauge water elevation
2. $\mathrm{El}_{\mathrm{downstream}}$ = Downstream gauge water elevation
3. $\mathrm{Mile}_{\mathrm{site}}$ = River-mile of site
4. $\mathrm{Mile}_{\mathrm{upstream}}$ = River-mile of upstream gauge
5. $\mathrm{Mile}_{\mathrm{downstream}}$ = River-mile of downstream gauge

Flow in the tributaries was influenced by the Mississippi River, thus Mississippi River water elevation was used for the downstream level for the tributaries at the point of confluence for both the Whitewater and Root River sites. This was due to neither tributary having a river gauge installed downstream of our study sites. Thus, Mississippi River elevation calculations were nested within tributary elevation calculations for the Whitewater and Root River sites. We validated site water elevations by installing trail cameras at each site; these were set to take daily photographs to validate plot flooding as predicted by linear interpolation models. After visually assessing days flooded versus not flooded in trail camera photographs, we adjusted the predicted flood values at each site with a coefficient in the linear interpolation formulas to match flood conditions observed in photographs. Water elevation data for 2017 were only used through July 31 due to plant sampling occurring in early August.

Due to data availability, long-term flood day calculations varied among sites for Collischan (2007 – 2016), Whitewater (2008 – 2013), Root River (2004 – 2013), and Whalen (2007 – 2015). Long-term flood days within a given plot were averaged per growing season over the course of time evaluated. In long-term analyses, missing data were handled by averaging the day prior to and after the missing value. When more than one consecutive day was missing, missing data were calculated using regression based on the existing data for that year. This was done for the Lake City gauge for 5 days in 2009 and 40 days in 2016 (used to estimate water levels at Collischan) as well as the Lansing gauge for 18 days in 2014 (used for Whalen). In all of these cases, R^2 values for the regression equations exceeded 0.90.

We modeled the response of other species percent cover as a function of RCG percent cover while testing the importance of four candidate explanatory variables to determine a best-fit model; these candidate explanatory variables included site, 2017 flood days, long-term flood days, and treatment. We completed this multivariate regression using the 'lm' function (R Core Team, 2018). We transformed other species percent cover using the 'transformTukey' function to better meet residual normality assumptions. We utilized corrected Akaike Information Criterion (AIC_c; 'ICtab' function; Bolker & R Development Core Team, 2017) to determine a best-fit model, where the model with the least AIC_c value was selected as best-fit (Burnham & Anderson, 2002; Symonds & Moussalli, 2011). Our null model was simply other species percent cover (y) as a function of RCG percent cover (x_{RCG}):

Null Model: $y = \varepsilon + \beta_0 + \beta_{RCG} x_{RCG}$

where ε is the error term, β_0 is the intercept, and β_{RCG} is the slope. Next, we added terms to our null model to test the effects of the various candidate explanatory models; we tested both an additive and interactive model for each candidate explanatory variable and selected the model with the lowest $\mathrm{AIC_c}$ for final model comparisons. To test the effects of flooding on other plant percent cover responses, we used additive models for 2017 flooding (x_{2017}) and long-term flooding (x_{LT}):

2017 flooding: $y = \varepsilon + \beta_0 + \beta_{RCG}x_{RCG} + \beta_{2017}x_{2017}$

Long-term flooding: $y = \varepsilon + \beta_0 + \beta_{RCG}x_{RCG} + \beta_{LT}x_{LT}$

where β_{2017} is the coefficient for 2017 flooding and $\beta_{LT}$is the coefficient for long-term flooding. We then tested for a threshold effect of 2017 flooding using a segmented regression model:

2017 flooding threshold model: $y = \begin{cases} \varepsilon + \beta_0 + \beta_{RCG}x_{RCG} + \beta_{2017}x_{2017}\,, & x_{2017} \leq X \\ \varepsilon + \beta_0 + \beta_{RCG}x_{RCG} + \beta_{2017}x_{2017}\,, & x_{2017} > X \end{cases}$

where *X* is the threshold 2017 flood duration. We used the 'segmented' command (Muggeo, 2008) for this segmented regression model. To test whether other species percent cover responded differently among sites, we used an interactive model that included terms for each site:

Site interactive: $y = \varepsilon + \beta_0 + \beta_{RCG}x_{RCG} + \beta_{Site}x_{Site} + \beta_{RCG\times Site}(x_{RCG}x_{Site})$ ...

where β_{Site} is the coefficient for a given site, x_{Site} is a dummy variable for each site (i.e. 1 if yes, 0 if no), $\beta_{RCG\times Site}(x_{RCG}x_{Site})$ is the interaction term for each site, and '…' is the addition of dummy variables and interaction terms for the remaining sites. The final models that we tested included adding 2017 flooding to the site interactive model and a saturated interactive model that included all explanatory variables. We selected a best-fit model using the minimum $\mathrm{AIC_c}$, then we calculated the $\Delta\mathrm{AIC_c}$ for each model by taking the difference between the AIC_c value of the best model and the AIC_c of other models in question. For each model, we evaluated residual normality assumptions by examining residual plots and using the Shapiro-Wilk normality test. Additionally, we used the Breusch-Pagan test (‘bptest’ function; Zeileis & Hothorn, 2002) to evaluate heteroscedasticity.

RESULTS

Reed Canarygrass Performance

For 2017 RCG height, all factors as well as the Month × Treatment and Month × Site × Treatment interactions were significant (Table 3; Fig. 3). In June, mean RCG height in Control plots was approximately two- to five-times greater than in experimental plots across all sites (Fig. 3). RCG height was similar in Rodeo and Oust plots throughout 2017, with the exception of Oust having greater height at Collischan in August (mean 54.8 cm vs. 32.8 cm) and at Whalen in July (mean 61.2 cm vs 42.5 cm; Fig. 3). RCG height in Rodeo and Oust plots increased between June and August whereas height in Control plots declined during this time period. However, significant differences remained between Control and experimental plots in August, with the exception of at Whalen (Fig. 3). In 2018, all RCG height factors and interactions were significant (Table 3). There were no significant differences in the height of RCG between Control and experimental plots in May 2018, with the exception of RCG height in Rodeo plots being greater than Control plots at Root River (68.6 cm vs. 43.3 cm; Fig. 3). Between June and August 2018, height among plot type remained the same at all sites with the exception of Whitewater. Here, RCG was approximately 20 cm taller in Rodeo plots than in Control plots in July and August, while RCG in Oust plots was 20 cm – 30 cm shorter than in Control plots throughout the growing season (Fig. 3).

Table 3. Summary of analysis of deviance P-values and chi-squared test values for response variables in mixed-effects linear models. Significant (P<0.05) factors and interactions are highlighted in bold.

	Factor			Interaction			
Response variable	Month	Site	Treatment	Month × Site	Month × Treat.	Site × Treat.	Month × Site × Treat.
2017 RCG Height	**<0.001** (df=2, X^2=60.5)	**<0.001** (df=3, X^2=45.6)	**<0.001** (df=2, X^2=352.0)	0.12 (df=6, X^2=10.2)	**<0.001** (df=4, X^2=202.7)	0.40 (df=6, X^2=6.3)	**<0.001** (df=12, X^2=38.8)
2018 RCG Height	**<0.001** (df=3, X^2=706.8)	**<0.001** (df=3, X^2=166.3)	**<0.001** (df=2, X^2=85.4)	**<0.001** (df=7, X^2=123.1)	**0.017** (df=6, X^2=15.5)	**<0.001** (df=6, X^2=53.7)	**0.013** (df=14, X^2=28.3)
2017 RCG Cover	**<0.001** (df=2, X^2=322.8)	**<0.001** (df=3, X^2=46.8)	**<0.001** (df=2, X^2=294.5)	**<0.001** (df=6, X^2=31.0)	**<0.001** (df=4, X^2=65.5)	0.081 (df=6, X^2=11.2)	0.562 (df=12, X^2=10.6)
2018 RCG Cover	**<0.001** (df=3, X^2=363.3)	**<0.001** (df=3, X^2=35.7)	0.176 (df=2, X^2=3.5)	**<0.001** (df=9, X^2=229.4)	**0.009** (df=6, X^2=17.2)	0.165 (df=6, X^2=9.2)	0.053 (df=18, X^2=28.6)
2017 Litter Cover	0.056 (df=2, X^2=5.8)	**<0.001** (df=3, X^2=68.4)	**<0.001** (df=2, X^2=201.3)	0.492 (df=6, X^2=5.4)	0.809 (df=4, X^2=1.6)	**<0.001** (df=6, X^2=25.9)	0.115 (df=12, X^2=18.0)
2018 Litter Cover	**<0.001** (df=3, X^2=151.4)	**<0.001** (df=3, X^2=72.9)	0.0648 (df=2, X^2=5.3)	**<0.001** (df=9, X^2=206.6)	0.634 (df=6, X^2=4.3)	**0.029** (df=6, X^2=14.1)	**0.005** (df=18, X^2=37.1)
2017 Litter Depth	**<0.001** (df=2, X^2=115.5)	**<0.001** (df=3, X^2=31.0)	**<0.001** (df=2, X^2=379.8)	0.0632 (df=6, X^2=11.9)	**<0.001** (df=4, X^2=32.0)	0.208 (df=6, X^2=8.4)	0.631 (df=12, X^2=9.8)
2018 Litter Depth	**<0.001** (df=3, X^2=89.9)	**<0.001** (df=3, X^2=134.1)	**<0.001** (df=2, X^2=17.8)	**<0.001** (df=9, X^2=155.6)	0.331 (df=6, X^2=6.9)	0.560 (df=6, X^2=4.9)	0.197 (df=18, X^2=22.8)

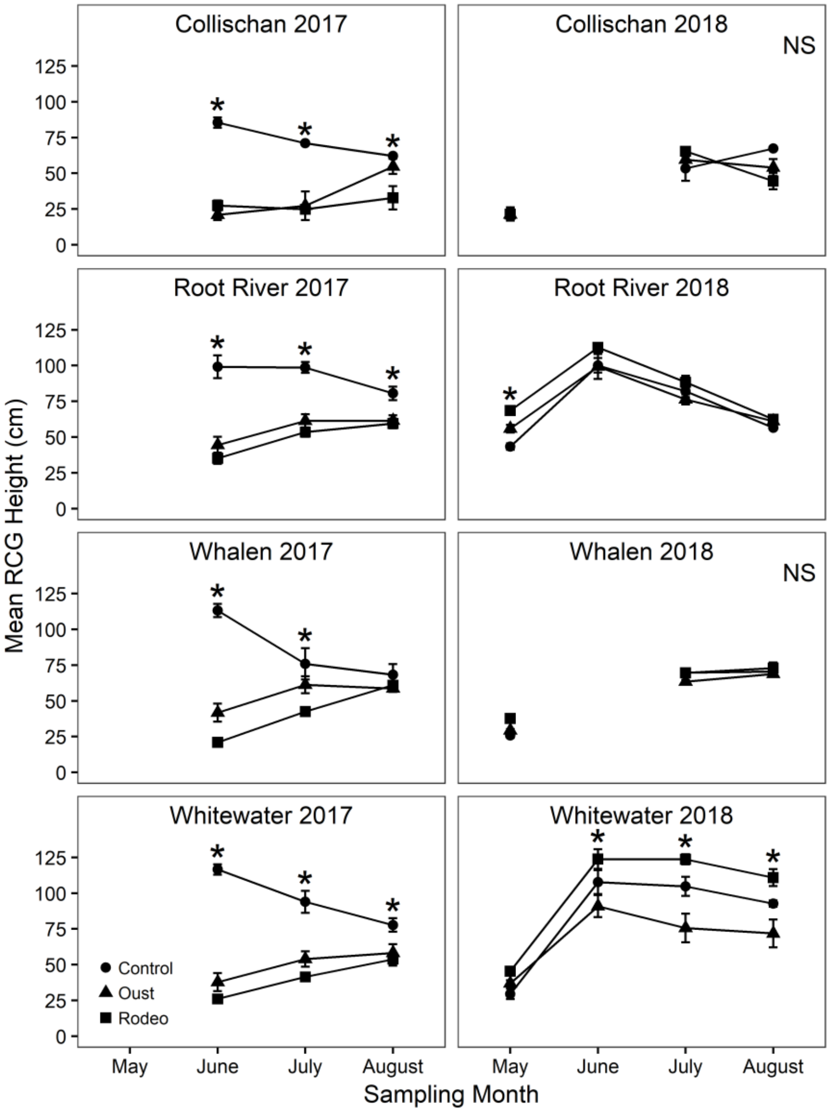

Figure 3. Mean reed canarygrass height by month and treatment at each site. Asterisks indicate within-month pair-wise significant difference (P<0.05) among treatment types; sites with no significant difference among treatment types are indicated with 'NS.' Missing values for Collischan and Whalen in 2018 are due to flooding.

For 2017 RCG flowering culm production, the Treatment factor was significant (Table 4; Fig. 4); RCG produced 0.6% to 25% the number of flowering culms in experimental plots as in Control plots, across all sites. Additionally, flowering culm production in Rodeo plots was approximately 10% of that in Oust plots across all sites, with the exception of at Collischan; here, flowering culm production in Oust and Rodeo

Table 4. Summary of two-way factorial ANOVA F- and P-values for non-RCG species richness and RCG flowering culms. Significant ($P<0.05$) factors and interactions are highlighted in bold.

	Factor and Interaction		
Response variable	Treatment	Site	Treatment × Site
2017 Flowering Culms	**<0.001** (df=2, F=67.5)	0.520 (df=3, F=0.8)	0.348 (df=6, F=1.1)
2018 Flowering Culms	**<0.001** (df=2, F=12.6)	**0.004** (df=1, F=9.4)	0.323 (df=2, F=1.2)
2017 Species richness	**<0.001** (df=2, F=12.9)	**<0.001** (df=3, F=32.7)	0.333 (df=6, F=1.2)
2018 Species richness	**0.048** (df=2, F=3.2)	**<0.001** (df=3, F=12.4)	**0.045** (df=6, F=2.3)

plots were both near zero (means 1.9 culms/m^2 and 0.5 culms/m^2, respectively) versus Control plots (mean 80 culms/m^2; Fig. 4). In 2018, we were unable to count RCG flowering culm production at Collischan and Whalen due to flooding; thus, 2018 flowering culm production only considers Root River and Whitewater. In 2018, RCG flowering culm production was significantly affected by both Site and Treatment (Table 4). At Root River, RCG flowering culm production in Rodeo plots was nearly four times greater than Control plots while flowering culm production in Oust plots was intermediate. At Whitewater, RCG flowering culm production in Rodeo plots was about four times greater than Oust while culm production in Control plots was intermediate (Fig. 4).

For 2017 RCG percent cover, all factors and the Month × Site and Month × Treatment interactions were significant (Table 3; Fig. 5). RCG percent cover in control plots remained near 100% June – August whereas it increased in experimental

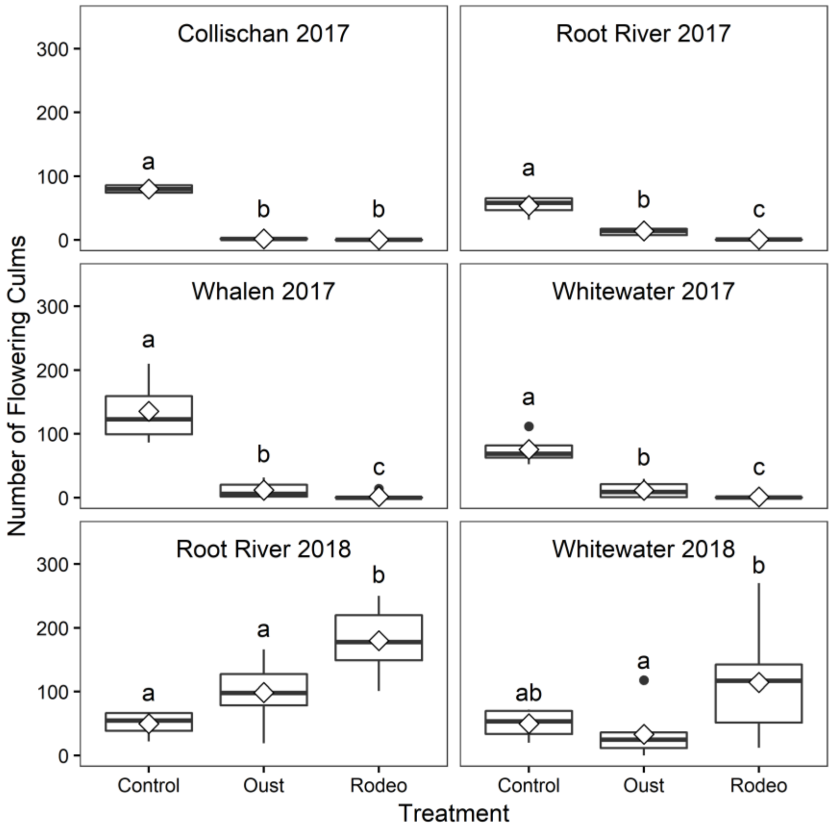

Figure 4. Boxplots of reed canarygrass flowering culm production per m^2 among treatments at each site. Boxes indicate interquartile range, whiskers indicate range of data occurring within 1.5-times the interquartile range, points outside of whiskers are considered outliers, horizontal lines within boxes indicate median, diamonds indicate mean, and letters above boxes indicate within-site post-hoc test statistical groups ($P<0.05$). Flowering culms were counted within five 1-m^2 quadrats per plot and then averaged. We were unable to count flowering culms at Collischan and Whalen in 2018 due to flooding.

plots during this time period at all sites (Fig. 5). RCG percent cover in experimental plots was significantly less than Control plots in all samples with the exception of Oust plots at Root River in July and August and in Oust plots at Whalen in August. Additionally, RCG percent cover in Rodeo plots was 15% to 40% less than Oust plots in all samples with the exceptions of Collischan at all time points and at Whitewater in August (Fig. 5). In 2018, RCG percent cover Month and Site factors as well as Month × Site and Month × Treatment interactions were significant (Table 3); however, there was no

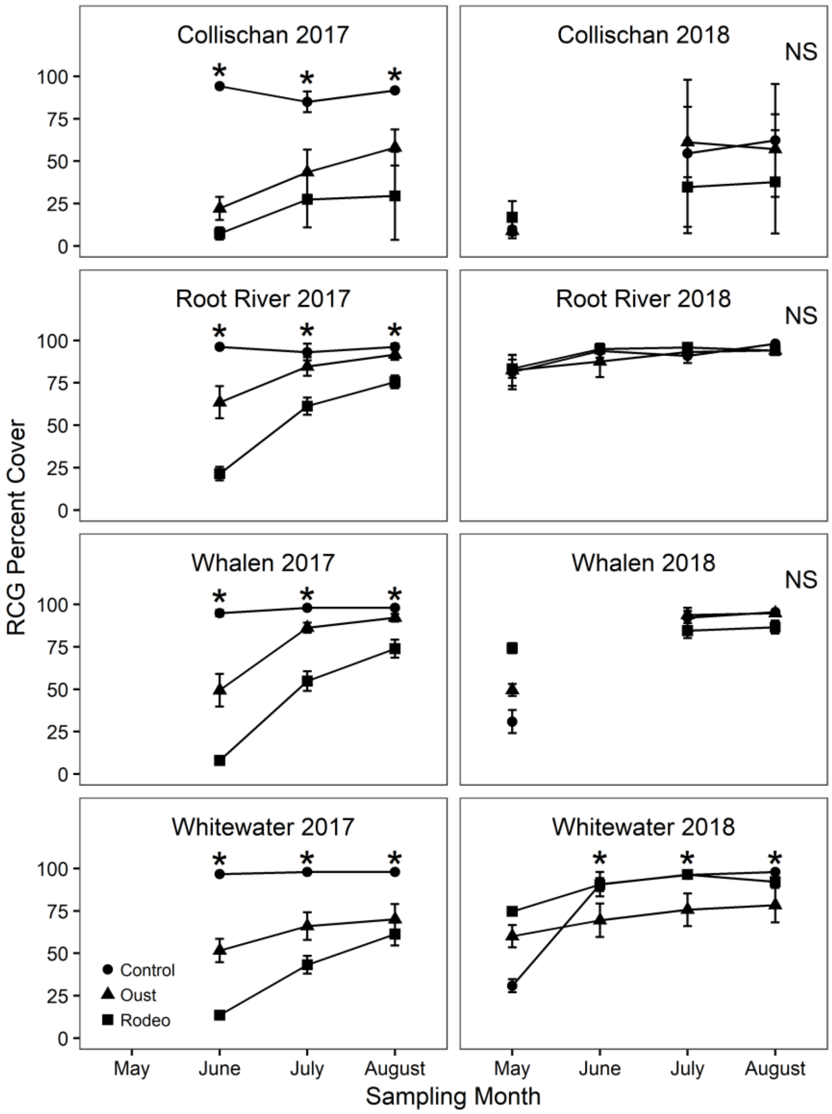

Figure 5. Mean reed canarygrass percent cover by month and treatment at each site. Asterisks indicate within-month pair-wise significant difference (P<0.05) among treatment types; sites with no significant difference among treatment types are indicated with 'NS.' Missing values for Collischan and Whalen in 2018 are due to flooding.

significant difference among treatments at Collischan, Root, and Whalen at any time point (Fig. 5). While there was no significant difference among treatments at Whitewater in May, we observed pair-wise significant differences where RCG cover in Oust plots was approximately 20% less than in Rodeo and Control plots in June through August.

Litter Response

For 2017 litter percent cover, the Site and Treatment factors as well as the Site × Treatment interaction were significant (Table 3; Fig. 6). There were no significant differences between Rodeo and Control within all sites and months except for Whitewater in August; here, Rodeo plots had about 10% lower litter cover than Control plots (Fig. 6). Although litter cover in Oust plots was significantly less than both Control

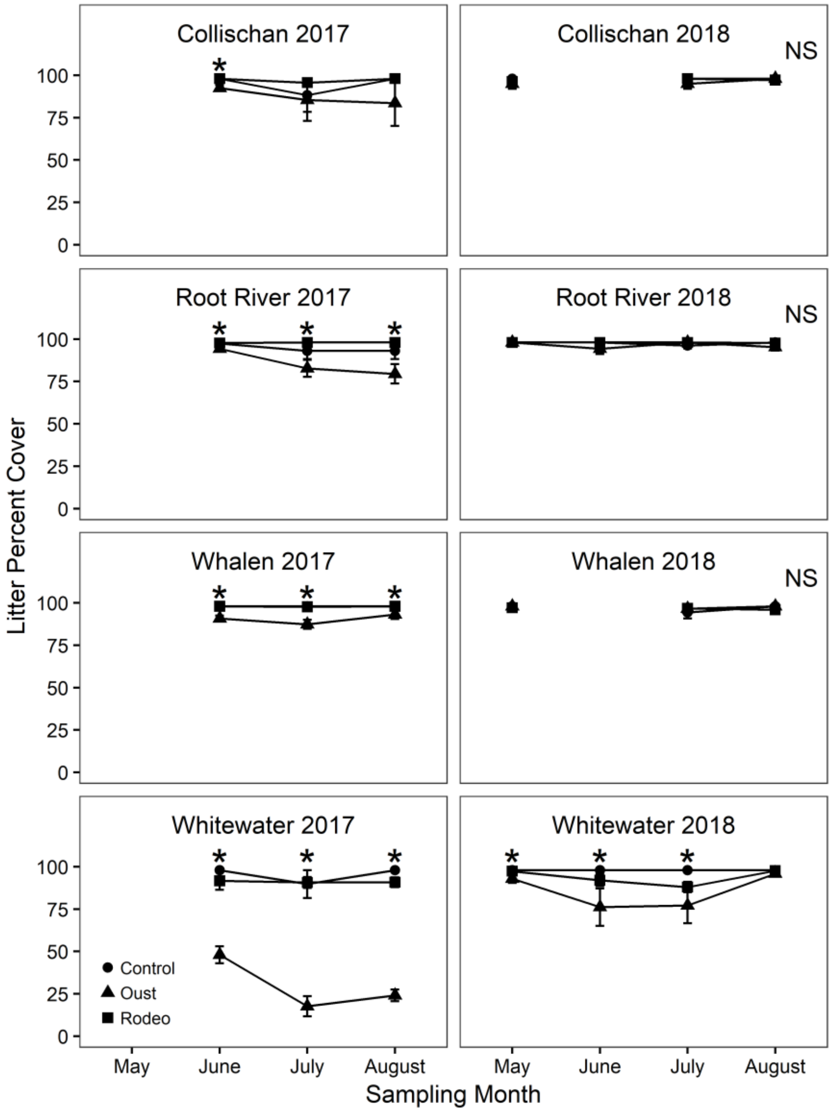

Figure 6. Mean litter percent cover by month and treatment at each site. Asterisks indicate within-month pair-wise significant difference ($P<0.05$) among treatment types; sites with no significant difference among treatment types are indicated with 'NS.' Missing values for Collischan and Whalen in 2018 are due to flooding.

and Rodeo plots for all sites and months (with the exception of Collischan in July and August), the decrease was less than 20% at Collischan, Root River, and Whalen. At Whitewater, litter cover in Oust plots was reduced by at least 50% in June through August (Fig. 6). In 2018, Month and Site as well as interactions Month × Site, Site × Treatment, and Month × Site × Treatment significantly affected litter cover (Table 3). However, post-hoc tests indicated no difference between treatments with the exception of pair-wise differences at Whitewater in May (Oust 5% less than Control), June (Oust 22% less than Control; Rodeo 6% less than Control), and July (Oust 21% less than Control; Rodeo 10% less than Control; Oust 11% less than Rodeo; Fig. 6).

All 2017 litter depth factors as well as the Month × Treatment interaction were significant (Table 3; Fig. 7). Mean litter depths in Oust plots ranged from 0.3 cm to 3.6 cm and were significantly less than that of Control plots for all sites and months. In June, litter depths in Rodeo plots were similar to those in Control plots at all sites; however, litter depths in Rodeo plots were significantly less than that of Control plots by August at Root River (5.1 cm vs 8.1 cm) and Whalen (3.4 cm vs 8.3 cm). By comparison, litter depths in Oust plots were approximately 56% to 78% less than that of Rodeo plots at all sites in June; however, litter depths in Oust plots were similar to those of Rodeo plots by August at Collischan and Whalen (Fig. 7). For 2018 litter depth, all factors as well as the Month × Site interaction were significant (Table 3). However, treatments were not significantly different for all samples, with the exceptions of pair-wise differences at Whitewater in May (Rodeo 50% less than Control and Oust), Collischan in August (Rodeo 45% less than Control), and Root River in August (Oust 70% less than Control).

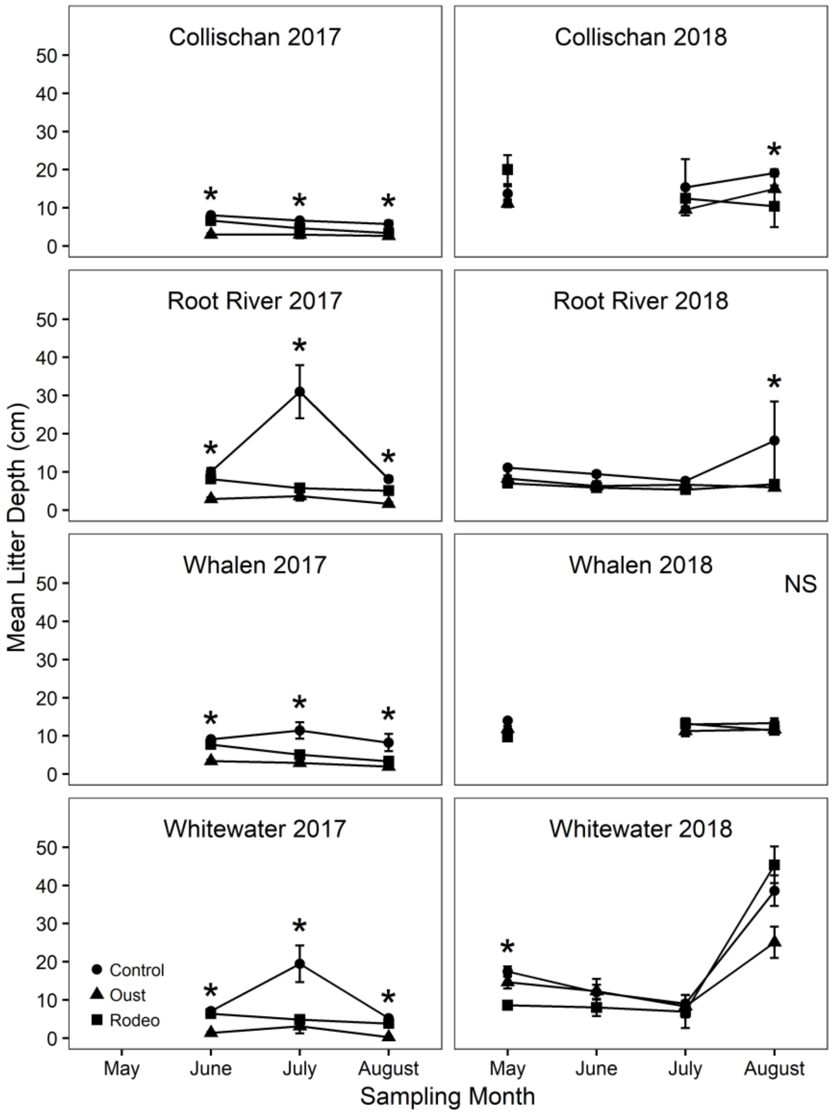

Figure 7. Mean litter depth by month and treatment at each site. Asterisks indicate within month pair wise significant difference ($P<0.05$) among treatment types; sites with no significant difference among treatment types are indicated with 'NS.' Missing values for Collischan and Whalen in 2018 are due to flooding.

Non-RCG Plant Response

Volunteer plants were represented by 26 families with 55 species as well as unknown plants identified to 12 genera (Appendix A). In 2017, we counted 87 volunteer tree seedlings across the 68 plots sampled at multiple time points (*Acer negundo, A. saccharinum*, *Fraxinus pennsylvanica*, *Populus deltoides*, *Celtis occidentalis,* and *Salix* spp.). In the test of 2017 non-RCG species richness, the Treatment and Site factors were significant (Table 4, Fig. 8). Non-RCG species richness in control plots was generally low (maximum of 2.0 species) and significantly less than Oust and Rodeo plots at all sites except for Root River. Non-RCG species richness even in treated plots was quite low at Whitewater and Whalen (approximately 2.25 species), whereas at Collischan we

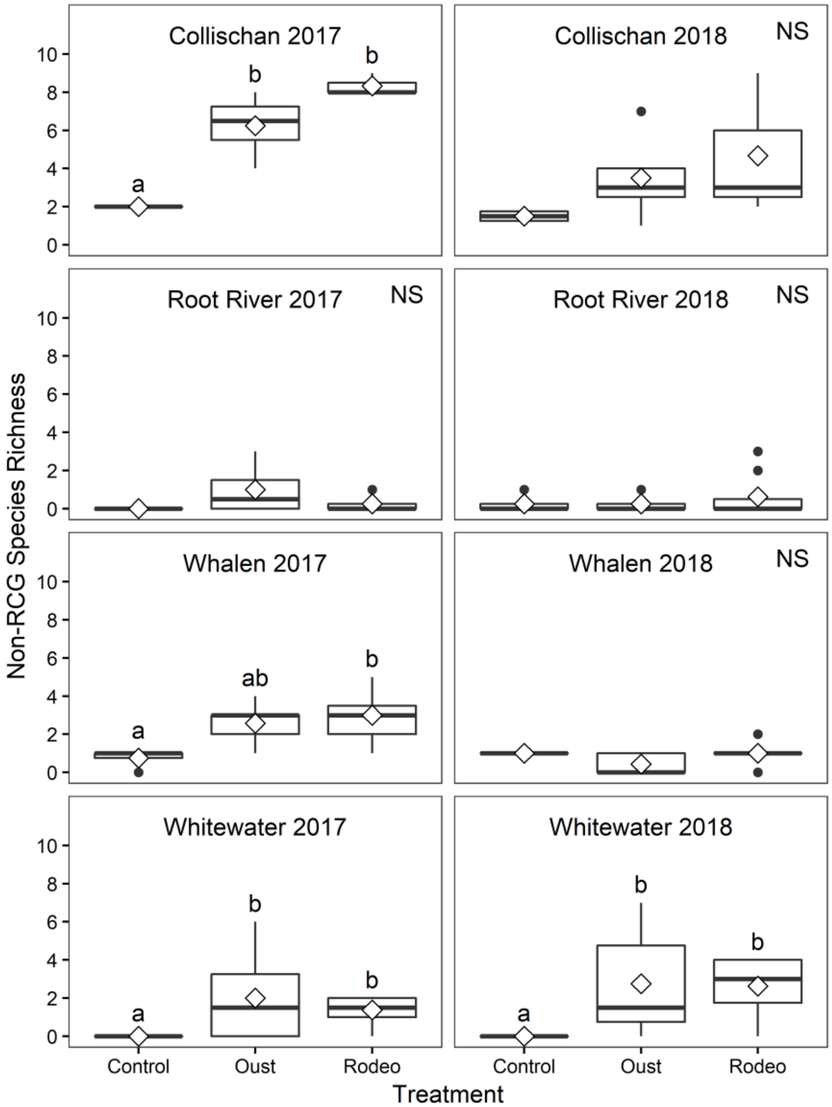

Figure 8. Boxplots of non-RCG species richness among treatments at each site. Boxes indicate interquartile range, whiskers indicate range of data occurring within 1.5-times the interquartile range, points outside of whiskers are considered outliers, horizontal lines within boxes indicate median, diamonds indicate mean, and letters above boxes indicate within-site post-hoc test statistical groups (P<0.05). Species richness data were collected in August 2017 and 2018.

observed 6.3 and 8.3 species in Oust and Rodeo plots, respectively. In 2018, non-RCG species richness Treatment and Site factors as well as the Treatment × Site interaction were significant (Table 4, Fig. 8). However, there were no significant differences among treatments at Collischan, Root River, or Whalen. Non-RCG species richness in Whitewater Rodeo and Oust plots averaged 2.7 while none were observed in Control plots.

In modeling Other plant percent cover, our best model based on AIC_c values was the interactive model using RCG cover and Site as explanatory variables; further, no other models had a $\Delta AIC_c < 2$ (Table 5). Other plant response versus RCG cover varied

Table 5. Summary of Akaike Information Criterion (AIC) results for the multivariate linear regression analysis; values are maximized log-likelihood ($\log(\mathcal{L})$), the number of parameters (K), corrected-AIC value (AIC_c), the difference between AIC_c values of the best-fit model and the model in question (ΔAIC_c), the Akaike weights (ω_i), and adjusted-R^2. Other species percent cover was the dependent variable as a function of RCG percent cover; candidate models included Site, 2017 flooding, Long-term flooding, and Treatment as candidate explanatory variables. Operators for model explanatory variables indicate additive (+) versus interaction (×) models. Segmented (Seg()) models tested threshold models as a function of 2017 flooding. Bold indicates best-fit model.

Model Explanatory Variables	$\log(\mathcal{L})$	K	AIC_c	ΔAIC_c	ω_i	R^2
RCG cover × Site	**-55.5**	**9**	**132.2**	**0.0**	**0.8**	**0.67**
RCG cover × Site + 2017 flooding	-55.5	10	134.9	2.8	0.2	0.67
RCG cover + 2017 flooding	-83.0	4	174.6	42.4	<0.001	0.32
Seg(RCG cover + 2017 flooding)	-81.2	6	175.9	43.7	<0.001	0.34
Null Model (RCG cover)	-92.1	3	190.7	58.5	<0.001	0.13
RCG cover + Long-term flooding	-92.1	4	192.8	60.6	<0.001	0.11
Saturated Model (RCG cover × Site × 2017 Flooding × Treatment)	-32.8	41	280.0	147.9	<0.001	0.64

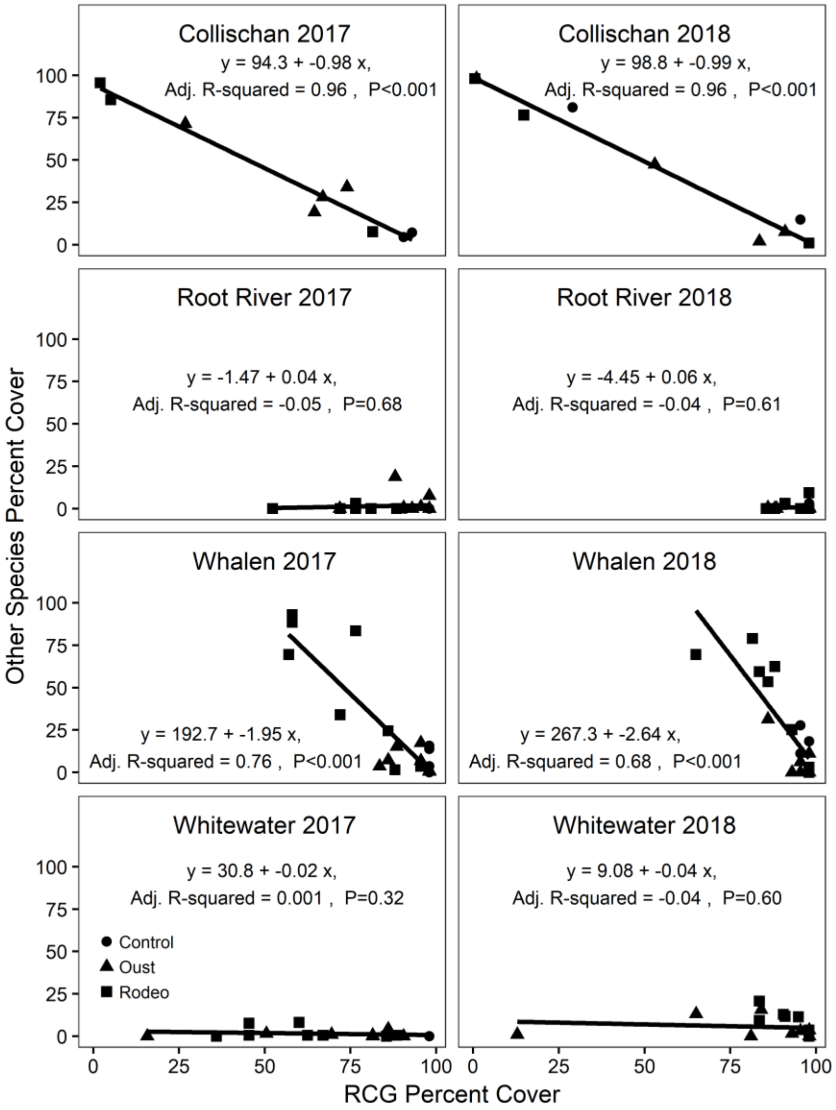

Figure 9. Regression analysis of Other species percent cover (y-axis) versus RCG percent cover (x-axis) by site and year. Only August percent cover data were considered for this analysis.

across sites in both 2017 and 2018; here, we observed inverse relationships between Other plant cover and RCG cover at both Collischan and Whalen whereas there was no relationship at Root River and Whitewater (Fig. 9). The relative importance of Site as an explanatory variable was 1.0 (sum of Akaike weights for models containing this variable). Flooding in 2017 appeared to vary both among sites and from long-term trends (Fig. 10); when modeling Other plant percent cover as a function of flooding and RCG

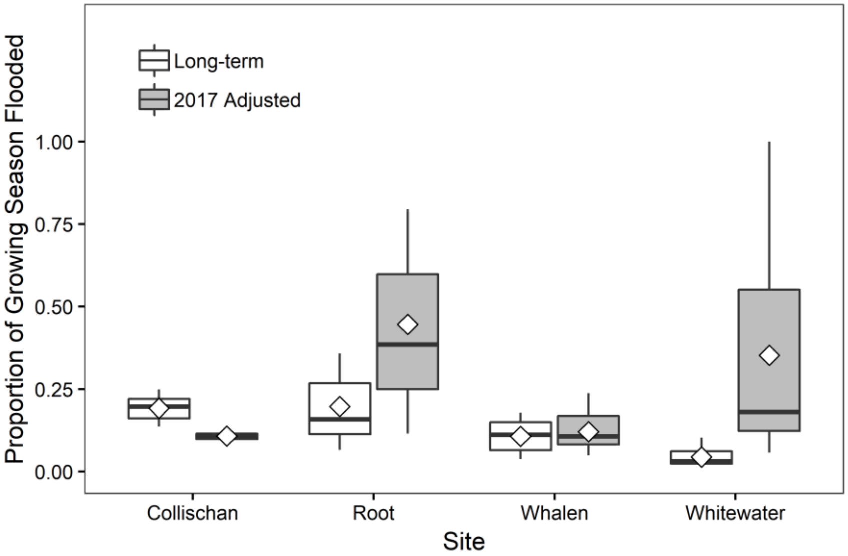

Figure 10. Boxplots of growing season 2017 adjusted flood and long-term (≥ 6 years, 2004 – 2017) flood data by site. Boxes indicate interquartile range, whiskers indicate range of data occurring within 1.5-times the interquartile range, points outside of whiskers are considered outliers, horizontal lines within boxes indicate median, and diamonds indicate mean. Growing season flooding for each plot was estimated by linear interpolation of Mississippi River (Collischan and Whalen), Whitewater River (Whitewater site), and Root River (Root River site) river gauge data and digital elevation model data. 2017 flooding was calculated using April 1 – July 31, 2017 gauge data.

percent cover, additive models that included 2017 flooding performed better than the null model (RCG percent cover only) whereas the long-term flooding additive model performed worse than the null model (Table 5). The test of a threshold effect for 2017 flooding used a segmented regression with a 2017 flooding break point of 0.149 (approximately 27 growing season days); however, this model did not perform as well as the 2017 flooding additive model. The relative importance of 2017 flooding as an explanatory variable was 0.2; however, the model that included RCG percent cover, Site, and 2017 flooding as explanatory variables did not perform as well as the RCG cover and Site interactive model ($\Delta AIC_c = 2.8$; Table 5). Our saturated model (interactive model including all candidate explanatory variables) included Treatment; while this term marginally increased the strength of some models, we dropped this term from further consideration because its inclusion did not markedly increase the strength of any model in the set ($\Delta AIC_c < 2$; Burnham & Anderson, 2002).

DISCUSSION

Effectiveness of RCG Suppression

Our first objective was to evaluate the effectiveness of our treatments in controlling established RCG for more than one growing season (Question 1, Fig. 1). Initial Fall 2016 treatments reduced RCG performance for much of the 2017 growing season. Although RCG height was comparable between Rodeo and Oust plots, Rodeo seemed to perform slightly better than Oust overall, particularly as the growing season progressed. The effectiveness of Oust may have been better had it been applied at the full labeled rate or if we had completed the forestry mulching in late-Summer 2016, allowed the RCG to re-sprout, then applied Oust to actively growing RCG in October 2016. Oust is also labeled for ‘hardwood release,’ where the herbicide is applied in the dormant season prior to bud-break of planted trees. This, along with our results, indicates that Oust may be effective in providing follow-up RCG control after planting trees for restoration.

While our initial results in 2017 were encouraging, RCG in experimental plots recovered throughout the 2017 growing season and RCG growth characteristics were back to that of Control plots by our May 2018 sampling. Similar to Reinhardt Adams & Galatowitsch (2006), the recolonization of RCG that we observed appeared to be from dormant rhizomes. Thus, our results underscore the importance of multiple herbicide treatments for effective control of RCG to fully control established plants as well as RCG germinating from the soil seed bank. Schulz, Vaughan, and Remelius (2009) demonstrated the utility of a physiologically based approach for controlling invasive *Lonicera* spp. shrubs, with an initial mechanical treatment followed up by an herbicide application approximately eight growing-season weeks later, to interrupt replenishing of non-structural carbohydrate root reserves. A similar approach could be utilized for follow-up treatments on RCG; there were approximately 60 growing-season days between our initial herbicide treatments and our first sampling period in June 2017. The reduced RCG flowering, percent cover, and height in experimental plots, followed by the increase in RCG height and percent cover between the June and July sampling, gives some insight into the optimum time needed between herbicide applications. It is likely that a follow-up herbicide application during this time period would provide more complete control by disrupting RCG recovery and interrupting the replenishing of root carbohydrate reserves; however, site access for herbicide applications during this time period can be complicated by flooding in the UMR.

Scaling Up RCG Suppression Techniques

Our study sought to test the demonstrated techniques of Miller-Adamany et al. (*in press*) on a large-scale restoration, since some techniques evaluated by small-scale studies are not practical to be implemented at the larger scale (Kettenring & Reinhardt Adams, 2011). Weather-related year effects were important in determining restoration outcomes; in contrast to Miller-Adamany et al. (*in press*), we were unable to complete our scheduled follow-up herbicide applications due to warm weather in Fall 2017, snow in mid-April 2018, and flooding in May – June 2018. In the case of the Fall 2017 herbicide application, our original intent was to apply herbicide after planted trees (see DeLaundreau, 2019) senesced and RCG was still actively growing; however, unseasonably warm weather in Fall 2017 caused planted trees to not senesce until well into November, at which time the RCG was no longer actively growing. The size of our study required that herbicide applications be made by mechanized equipment and these multiple weather issues as well as need to avoid harming planted trees prevented our planned follow-up herbicide applications; at the smaller scale, it is likely we could have completed these applications with a more targeted approach (e.g. spraying herbicide on foot with a backpack sprayer). We need to develop restoration strategies using more thorough control prior to planting desirable plants, as well as having less-narrow parameters for follow-up treatments by developing an 'adaptive management' strategy of RCG control to respond to annual conditions (Almquist, Wirt, Adams, & Lym, 2015; Healy et al., 2015). Understanding setbacks allows for a better understanding of the study system and should improve future restoration outcomes (Catford, 2016; Neeson, Smith, Allan, & McIntyre, 2016); in the UMR, it is necessary to have a flexible control program to mitigate undesirable outcomes from year effects. Further, landscape position effects are important for prioritizing areas to restore, such as targeting areas less likely to have prolonged inundation, where flooding may prevent access, or where prolonged flooding can have negative effects on plant recruitment for the target vegetative community (Dixon & Turner, 2006; Kabrick, Dey, Van Sambeek, Coggeshall, & Jacobs, 2012).

Non-RCG Plant Response

Our second objective was to evaluate the capacities of natural plant community assemblages to recolonize following RCG suppression in the UMR. Our analyses showed that volunteer plant responses were substantially different across sites in ways which are consistent with the idea that native seed limitation is an issue in our system (Fig. 1). In our model of non-RCG plant cover, we determined that RCG percent cover and site were the most important determinants of volunteer plant response. While models that included 2017 flooding performed better than the null model, variation in flooding did not explain the differences in non-RCG plant response that we observed across sites. Flooding in 2017 appeared to explain some of the plant response at the plot level (e.g. plots with extended inundation); however, there are variations in attributes among our four sites not accounted for. While litter feedbacks by RCG have been shown to be important in stimulating RCG invasion (Gaertner et al., 2014; Kaproth et al., 2013; Zedler, 2009), litter did not have a significant impact on non-RCG plants in our study; rather, it was the percent cover of actively growing RCG that negatively impacted other plants. Our observations show that RCG litter breaks down through the growing season; thus, controlling RCG growth and recolonization will reduce impacts of litter over time, as suggested by Miller-Adamany et al. (*in press*).

Ecological communities affected by plant invasion are often in highly modified landscapes where feedbacks among abiotic and biotic factors result in alternative states; past land-use affects native plant propagule availability and degraded areas often have limited amounts of native plant seed (Hobbs & Suding, 2009; Morimoto, Shibata, Shida, & Nakamura, 2017; Suding et al., 2004) or high amounts of invasive plant seed (Dalton, Carpenter, Boutin, Allison, & Hölzel, 2017). Our results suggest that a lack of native propagule availability may explain the variation in non-RCG plant response among sites; Collischan and Whalen had both increased cover and species richness in experimental plots whereas Root River and Whitewater had little non-RCG plant cover, even though treatments affected RCG similarly in the first year across the four sites. While Whitewater did have increased species richness at the site level, this was driven by the presence of seed-producing trees within a limited number of plots (*personal observation*); thus, propagule supply was not uniform over the site. Following invasive plant removal, native propagules must be present or able to colonize from surrounding areas in order for desirable plant succession to occur (Flory, Bauer, Phillips, & Clay, 2017; Morimoto et al., 2017). We only documented 87 volunteer trees across the entire study area by the August 2017 survey; the bulk of these were in plots at Whitewater, near existing trees. Our overall lack of volunteer trees supports the idea of Miller-Adamany et al. (*in press*) that distance to forest edge is a critical determinant of natural colonization during RCG control. Effective restoration requires knowing a community's ability to revegetate on its own following invader removal (Miller et al., 2017; Wainwright et al., 2018); systems lacking the ability to revegetate following invader removal will require supplemental native plant propagule introduction in conjunction with invader removal (Holl & Aide, 2011; Kettenring & Reinhardt Adams, 2011; Lindenmeyer et al., 2015; Török, Helm, Kiehl, Buisson, & Valkó, 2018). An analysis of the soil seedbank or a direct test of native herb and tree seeding coupled with RCG control would have provided more insight into propagule availability across our sites; as such, this is a future research need that would provide further insight into native plant recolonization ability within the UMR.

Management Implications

Providing longer term RCG control is critical for restoring floodplain forest in the UMR; while our initial treatments were effective in reducing RCG performance, the rate of RCG recovery exceeded the rate of colonization by volunteer plants. Reducing propagule pressure from invaders is critical in preventing re-invasion (Von Holle & Simberloff, 2005) and the presence of RCG has been shown to inhibit wetland herb and swamp white oak survival (Green & Galatowitsch, 2002; Kercher & Zedler, 2004; Reinhardt Adams & Galatowitsch, 2007; Reinhardt Adams et al., 2011; Zedler & Kercher, 2004). Our overall lack of tree colonization and lack of herb colonization at two sites indicate that there is likely a lack of propagule availability, thus future RCG suppression projects in the UMR should be coupled with planting native species. Planted native species should include desired species with dispersal limitations (McClain, Holl, & Wood, 2011; Morimoto et al., 2017; Salaria, Howard, Clare, & Creed, 2018; Török et al., 2018), including heavy seeded species such as oak, hickory, and walnut (Knutson & Klaas 1998). Plant mixes should also include aggressive native species with the potential to compete against RCG (Healy et al., 2015; Wisconsin Reed Canary Grass Management Working Group, 2019); of these, a high proportion should be known for reliable establishment and a wide range of tolerances to overcome challenges of year effects (Groves & Brudvig, 2018). Although a soil seedbank may not have sufficient propagule volume to revegetate sites within the constraints of a restoration program, the seedbank is a potential source of additional biodiversity for restoration projects given effective long-term RCG control (Dalton et al., 2017; Reinhardt Adams & Galatowitsch, 2006); indeed, many of our observed volunteer species (Appendix A) were consistent with those documented in floodplain forest ecological assessments (Guyon & Battaglia, 2018; MN DNR, 2018b; WI DNR, 2018). Introduced seed may take multiple years to germinate and the rate of RCG recolonization has been shown to exceed the establishment of planted species, thus continued treatments of the invader are required at least through the native plant establishment period (Healy et al., 2015; Hölzel, Buisson, & Dutoit, 2012; Reinhardt Adams & Galatowitsch, 2006, 2007; Smith, Reinhardt Adams, Wiese, & Wilson, 2016). Further, invasive plant seed dormancy can exceed 10 years, resulting in continued invasive propagule pressure (Blossey, Nuzzo, & Dávalos, 2017; Török et al., 2018). Grass-selective herbicides, while not shown to eradicate RCG, may be used to suppress RCG following effective initial control when non-grass plants are the desired outcome (Annen, Tyser, & Kirsch, 2005; Healy et al., 2015; Zedler, 2009). Lastly, Silliman et al. (2014) showed effective usage of grazing to control the wetland invader *Phragmites australis*, a similar exploration on RCG control could provide additional management options after initial control treatments are applied. Published information is often lacking on the long-term (i.e. five years or more) effectiveness of invasive plant control programs (D'Antonio & Flory, 2017; Kettenring & Reinhardt Adams, 2011; Lindenmeyer et al., 2015). However, Flory et al. (2017) showed that invasive dominance can decline over time, and our results are consistent with recommendations by Holl and Aide (2011) that restorations need adequate funding and a plan for a 5-10 year period to increase probability of success through adequate invasive control as well as overcoming setbacks caused by year effects (Groves & Brudvig, 2018). Closing gaps between research and land managers is critical for successful widespread ecological restoration (Lamb, 2018; Wainwright et al., 2018). Here, we see that directly applying the results of smaller-scale research to a large-scale project can be challenging, thus land managers need to carefully consider how to adapt the findings from smaller-scale research to large-scale restoration projects, and future research should focus on the applicability of small-scale results to large-scale projects.

RCG: Passenger or Driver?

While the drivers versus passenger dichotomy is a useful starting point, we understand that in truth an invader may act as either a driver or passenger depending on the context. Overall, it is difficult to assess whether RCG is a passenger or a driver (Fig. 1) with our data alone due to our RCG control not being effective for more than one growing season; however, Thomsen et al. (2012) and Miller-Adamany et al. (*in press*) had positive results where native plants recolonized following RCG suppression. Despite only being able to apply one RCG suppression treatment, we did observe colonization of other plants in response to RCG suppression in plots at Collischan and Whitewater in both 2017 and 2018. Thus, when combined with Thomsen et al. (2012) and Miller-Adamany et al. (*in press*), our results indicate that RCG behaves as a driver with native propagule limitation as a potential factor in the UMR. Conversely, the UMR is a highly-altered, complex system and hydrologic alteration and other disruptions have altered successional pathways. While our results show that flooding did not explain the variation in other plant response at the site level, flooding did have a negative effect on plot-level non-RCG colonization in plots that were flooded for extended periods in 2017. Flooding has historically been the primary disturbance to the system and flood tolerance is a major driver of floodplain forest composition (Curtis, 1959; De Jager et al., 2012; Knutson & Klaas, 1998; Romano, 2010; Turner et al., 2004), and alteration of natural hydrology due to the lock and dam system has been a significant stimulus of ecological change in the UMR (Knutson & Klaas, 1998; Romano, 2010). Hydrologic alterations include a raised water table and natural flooding regimes that have been disrupted in both duration and seasonality (Romano, 2010). Most current floodplain forest overstory trees pre-date the lock and dam system, thus were recruited under different hydrologic conditions than present day (Guyon & Battaglia, 2018). De Jager et al. (2012) indicated threshold effects due to flooding on floodplain forest overstory composition and soil texture at 40% of the growing season and forest understory diversity at 25% of the growing season. When considered with these other factors, our results where models that included 2017 flooding performed better than the null model indicate that RCG likely behaves as a passenger on sites where hydrologic alteration by the lock and dam system has caused greater average annual flood duration during the growing season.

In addition to hydrologic alteration, other environmental disruptions have influenced UMR floodplain forests. The arrival of Dutch elm disease (*Ophiostoma ulmi*) has impacted forest composition and it is likely that emerald ash borer (*Agrilus planipennis*) will have a similar effect since ash species are prominent within the system (Guyon & Battaglia, 2018; Johnson & Waller, 2013; Romano, 2010). Increased deer herbivory of young trees has the potential to alter tree species composition through selective browsing (Cogger et al., 2014; Ruzicka, Groninger, & Zaczek, 2010); additionally, deer herbivory can reduce resilience of early successional forests to flooding by increasing tree mortality during floods (De Jager et al., 2013). Despite these extensive landscape-scale alterations, our results when combined with those of Thomsen et al. (2012) and Miller-Adamany et al. (*in press*), are encouraging for the successful restoration of UMR floodplain forest in areas less impacted by hydrologic alteration where RCG is behaving more as a driver given the development of effective adaptive management strategies.

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APPENDIX A

SUMMARY OF VOLUNTEER PLANTS AMONG SITES

Appendix A. Summary of volunteer plants in response to RCG suppression among Collischan (CL), Root River (RR), Whalen (WH), and Whitewater (WW) research sites. Taxonomy follows USACE (2018a).

Species	Research Site Occurrence
Family Aceraceae	
Acer negundo L. (Boxelder)	WW
Acer saccharinum L. (Silver Maple)	CL, RR, WW
Family Alismataceae	
Sagittaria latifolia Willd. (Duck-Potato)	RR
Family Amaranthaceae	
Amaranthus powellii S. Watson (Green or Tall Amaranth)	WH
Amaranthus retroflexus L. (Red-Root Amaranth)	CL, RR, WH
Amaranthus tuberculatus (Moq.) Sauer (Rough-Fruit Amaranth)	CL
Family Araceae	
Arisaema triphyllum (L.) Schott (Jack-in-the-Pulpit)	WW
Family Asclepiadaceae	
Asclepias incarnata L. (Swamp Milkweed)	CL, RR, WW
Asclepias syriaca L. (Common Milkweed)	RR, WW
Family Asteraceae	
Ambrosia trifida L. (Giant Ragweed)	CL, WW
Bidens cernua L. (Nodding Burr-Marigold)	WW
Bidens vulgata Greene (Tall Beggarticks)	CL, WW
Cirsium canadense (L.) Scop. (Canada Thistle)	RR, WH, WW
Cirsium muticum Michx. (Swamp Thistle)	CL
Cirsium vulgare (Savi) Ten. (Bull Thistle)	RR
Erigeron canadensis L. (Canadian Horseweed)	CL, RR, WW
Helenium autumnale L. (Fall Sneezeweed)	WH

Family Asteraceae (continued)	
Hieracium sp.	RR, WW
Lactuca serriola L. (Prickly Lettuce)	WH
Senecio hieraciifolius L. (American Burnweed)	RR, WH, WW
Solidago sp.	WW
Solidago canadensis L. (Canadian Goldenrod)	WW
Solidago gigantea Ait. (Late Goldenrod)	WW
Symphyotrichum spp.	RR, WW
Taraxacum officinale G. H. Weber ex Wiggers (Common Dandelion)	RR, WH, WW
Xanthium strumarium L. (Rough Cockleburr)	CL
Family Balsaminaceae	
Impatiens capensis Meerb. (Spotted Touch-Me-Not)	WW
Family Boraginaceae	
Hackelia deflexa (Wahlenb.) Opiz (Nodding Stickseed)	WW
Family Brassicaceae	
Alliaria petiolata (Bieb.) Cavara & Grande (Garlic Mustard)	WW
Brassica nigra (L.) W.D.J. Koch (Black Mustard)	WW
Rorippa palustris (L.) Bess. (Bog Yellowcress)	CL, WH, WW
Family Convolvulaceae	
Calystegia sepium (L.) R. Br. (Hedge False Bindweed)	CL, WW
Convolvulus sp.	CL
Family Cucurbitaceae	
Echinocystis lobata (Michx.) Torr. & Gray (Wild Cucumber)	CL
Family Cyperaceae	
Carex spp.	CL, WH
Carex trichocarpa Muhl. ex Willd. (Hairy-Fruit Sedge)	RR
Cyperus esculentus L. (Chufa)	CL
Cyperus odoratus L. (Rusty Flat Sedge)	CL

Family Cyperaceae (continued)	
Schoenoplectus smithii (Gray) Soják	WW
Family Fabaceae	
Coronilla varia L. (Crownvetch)	RR
Lathyrus palustris L. (Marsh Vetchling)	CL
Trifolium sp.	RR
Trifolium pratense L. (Red Clover)	RR
Vicia sp.	CL
.	
Family Lamiaceae	
Galeopsis tetrahit L. (Brittle-Stem Hemp-Nettle)	CL, WW
Family Oleaceae	
Fraxinus pennsylvanica Marsh. (Green Ash)	WW
Family Poaceae	
Dactylis glomerata L. (Orchard Grass)	RR
Poa sp.	RR
Spartina pectinata Bosc ex Link (Prairie Cord Grass)	WH
Family Polygonaceae	
Fallopia sp.	CL
Persicaria amphibia (L.) S.F. Gray p.p. (Water Smartweed)	CL, RR, WH, WW
Persicaria hydropiper (L.) Delarbre (Mild Water-Pepper)	CL, WH
Persicaria lapathifolia (L.) S.F. Gray (Dock-Leaf Smartweed)	CL
Persicaria pennsylvanica (L.) M. Gómez (Pennsylvania Smartweed)	WH, WW
Family Ranunculaceae	
Ranunculus gmelinii DC. (Lesser Yellow Water Buttercup)	CL
Thalictrum dasycarpum Fisch. & Avé-Lall. (Purple Meadow-Rue)	WW
Family Salicaceae	
Populus deltoides Bartr. ex Marsh. (Eastern Cottonwood)	CL, WH, WW
Salix sp.	WH, WW

Family Smilacaceae	
Smilax sp.	WW
Family Typhaceae	
Typha sp.	CL, WH, WW
Family Ulmaceae	
Celtis occidentalis L. (Common Hackberry)	RR, WW
Family Urticaceae	
Laportea canadensis (L.) Weddell (Canadian Wood-Nettle)	CL
Boehmeria cylindrica (L.) Sw. (Small-Spike False Nettle)	CL, RR, WH
Urtica dioica L. (Stinging Nettle)	CL, WW, WH
Family Verbenaceae	
Verbena hastata L. (Blue Vervain, Simpler's Joy)	RR
Verbena urticifolia L. (White Vervain)	RR
Family Violaceae	
Viola sp.	WW
Family Vitaceae	
Vitis riparia Michx. (River-Bank Grape)	WW