Effects of Russian olive removal on soils and understory plant communities: Boulder Creek floodplain, Boulder, Colorado

FINAL REPORT

Submitted: February 15, 2016

Submitted to:

City of Boulder Open Space and Mountain Parks

Submitted by:

Dr. Gabrielle Katz
Research Scientist, Colorado State University
Assistant Professor, Metropolitan State University of Denver

Dr. Andrew Norton
Professor, Colorado State University

I. SUMMARY

Russian olive (*Elaeagnus angustifolia* L.) is an invasive exotic nitrogen (N)-fixing tree that is now one of the most common woody species in Western US riparian ecosystems. Its widespread invasion is considered detrimental to many riparian and rangeland ecosystems, and control and removal efforts are underway on rivers throughout the Western US. There is considerable need for assessments of the effects of Russian olive removal, as current understanding is based on a limited number of studies that have generally been short term in nature, or have lacked reference or baseline data. This project assessed the impacts of Russian olive, and the effects of Russian olive removal, on plant communities and soils on the floodplain of Boulder Creek. The purpose of the project was to compare two different removal treatments – cutting and removing Russian olive biomass (Cut/Remove treatment) vs. cutting and mulching Russian olive biomass on site (Cut/Mulch treatment), and two reference treatments – open areas not under Russian olive trees (Open treatment), and plots under Russian olive that was not removed (Control). Vegetation data were collected in August-September 2014 (pre-treatment), and in August-September 2015 (1-year post-treatment) in 20 plots per treatment type. Soils were collected in October of each year in a sub-set of 10 plots in the cut/mulch, open, and control treatments. Russian olive removal treatments were applied by OSMP staff in October, 2014. We encountered 66 species in the study plots in 2014, and 60 species in 2015. The plant community was dominated by non-native species. The five most abundant species were *Elymus repens*, *Poa pratensis*, *Bromus inermis*, *Camelina microcarpa*, and *Dipsacus follonum*. In 2014 prior to treatment, open plots had higher total plot level species richness, perennial species richness and native species richness, and lower total soil N, than plots under Russian olive. In 2015, one year after the tree removal treatments were applied, species richness of cut/remove plots was similar to that of open plots, though richness of cut/mulch plots was lower than that of open plots. Native species richness was still higher in the open plots than in all Russian olive treatment plots in 2015. Biennial species richness tended to be higher in the cut/remove and cut/mulch treatments, than in the control and open treatments in 2015, which is of potential concern since biennial species included *Arctium minus, Camelina microcarpa, Carduus nutans, Cirsium vulgare,* and *Dipsacus follonum*. Soil N did not differ among treatments in 2015, though soil N was higher at the site in overall 2015 compared to 2014. The higher levels of soil N in 2015 compared to 2014 could have been due to two factors (or to their combination) -- (1) ecological or environmental conditions in 2014 that reduced soil N (e.g., lingering effects of the 2013 flooding at the site), and (2) ecological or environmental conditions in 2015 that increased N levels (e.g., high precipitation and soil moisture in spring 2015).

II. INTRODUCTION

Russian olive (*Elaeagnus angustifolia* L.) is an invasive exotic nitrogen (N)-fixing tree that is now one of the most common woody species in Western US riparian ecosystems (Friedman et al. 2005). Although Russian olive was once promoted for windbreaks and wildlife plantings, its widespread invasion is now considered detrimental to many riparian and rangeland ecosystems. It is classified as a noxious weed in Colorado, New Mexico, Wyoming, and Connecticut and is a regulated plant in Montana. Russian olive invasion changes ecosystem physical structure by creating a mid-canopy layer in riparian forests and by forming dense thickets in formerly herbaceous habitats (Katz and Shafroth 2003). These changes tend to increase fire risk, reduce forage quality, and impede cattle management. Additional impacts to riparian zones include increased soil N (DeCant 2008, Follstad Shah et al. 2009). Impacts to streams include increased allochthonous litter inputs, benthic organic matter storage, and altered in-stream nutrient cycling (Mineau et al. 2011, Mineau et al. 2012). Russian olive water use may also impact riparian groundwater levels and streamflow (Hultine and Bush 2011). Our prior research (Tuttle et al. in review) suggests that increased soil N and reduced understory light levels following Russian olive invasion have profound impacts on surrounding plant communities, shifting dominance from native perennial species to exotic annual grasses and forbs. Identifying the duration of the Russian olive impact on soil N and plant community structure post-removal is a key information gap in developing treatment strategies for this species.

Russian olive control and removal efforts are underway on rivers throughout the Western US. Removal commonly utilizes cut stump herbicide treatments in association with biomass removal (O'Meara et al. 2010). Woody debris is usually piled, and either left in place or burned. An alternative approach is to chip the biomass, and to spread the woody debris as mulch in the riparian or floodplain ecosystem. Where piling biomass is not an acceptable tactic (e.g., in Boulder Open Space and Mountain Parks), chipping/mulching reduces the costs and disturbance associated with removal and disposal of the cut Russian olive trees. In upland forest ecosystems, cutting and in situ chipping treatments have been shown to affect soil conditions (Rhoades et al. 2012) and to alter understory plant communities (Wolk and Rocca 2009), compared to cutting and removal treatments 3-5 years post-treatment. To our knowledge, no studies have compared the effects of these contrasting treatment methods on riparian natural resources following Russian olive removal.

There is considerable stakeholder and manager need for assessments of the effects of Russian olive removal projects (Tamarisk Coalition 2009, Woody Invasive Control Committee 2010, Shafroth et al. 2010). Current understanding of the effects of Russian olive removal is based on a limited number of studies that have generally been short term in nature, or have lacked reference or baseline data. Merritt and Johnson (2006) monitored vegetation response to fuels reduction treatments (including Russian olive removal) on the Middle Rio Grande, NM. They found that woody plant removal affected herbaceous species richness and community composition one year post treatment, though this effect varied by site location along an upstream to downstream gradient. Reynolds and Cooper (2010) studied Russian olive removal in Canyon de Chelly National Monument, AZ. Removal caused changes to understory plant communities two years post-treatment, including shifts to more upland and native species. Further, soil N levels were elevated near killed Russian olive boles. Gaddis and Sher (2012) examined plant communities at twenty five sites in CO, WY and MT where Russian olive had been removed, and found an environmental influence (i.e., temperature and moisture related variables) on post-removal vegetation. These results suggest that elevated soil N may persist as a legacy of Russian olive invasion for several years after removal, though this effect is superimposed on existing environmental gradients. Many questions remain unanswered regarding the effects of Russian olive removal. In particular, none of these studies examined the effects of chipping/mulching compared to biomass removal treatments.

While convenience and cost-effectiveness make Russian olive chipping/mulching an attractive management strategy, decision makers and stakeholders require information about the ecological consequences of this treatment. Mulching adds chipped woody biomass to the soil surface, thereby changing the soil surface environment, and potentially altering soil physical conditions and biogeochemistry (Rhoades et al. 2012). The key effects of surface mulch on soils are increased soil moisture levels and reduced sub-surface temperature variability. Rhoades et al. (2012) also found higher soil N levels in mulched upland forest plots 3-5 years post treatment, compared to un-mulched treatments, though this response was likely due to enhanced moisture availability in mulched plots. Many studies have shown that adding carbon sources (e.g., sugar, sawdust or wood chips) to soil can reduce inorganic N pools and N availability, presumably by stimulating heterotrophic microbial activity which results in N immobilization (Perry et al. 2010). However, these effects are produced by actively mixing carbon into the soil (not simply applying it on the soil surface). Further, the effects of carbon addition treatments on soil N are temporary, as nutrient levels eventually return to pre-addition levels. Thus, over the short term mulching treatments are likely to alter soil conditions primarily via moisture and temperature effects. Over time, as the mulch is decomposed and incorporated into the soil, it could cause temporary changes to soil N levels. An additional effect of mulch treatments could be alteration of germination, emergence and establishment success of plant species due to physical effects of the mulch (i.e., if germinated seedlings have insufficient food reserves to emerge above the mulch layer, or if the mulch supports fungal pathogens that are lethal to seedlings; Donahue et al. 2006). For example, Donahue et al. (2016) found that emergence of transplanted non-native *Sapium sebiferum* seedlings and cover of native plants were reduced in mulched plots in a *Sapium*-invaded prairie in southeastern Texas, USA.

The purpose of the project was to compare two different removal treatments – cutting and removing Russian olive biomass (Cut/Remove treatment) vs. cutting and mulching Russian olive biomass on site (Cut/Mulch treatment). We also included comparisons with two reference treatments – open areas not under Russian olive trees (Open treatment), and plots under Russian olive that was not removed (Control). We collected baseline data on plant communities and soils in study plots in August-October, 2014. Removal treatments were applied by OSMP staff after data collection was completed in October, 2014. We re-sampled vegetation and soils in August-October 2015 in order to assess the initial effects of the removal treatments. We hypothesized that both Russian olive removal treatments would result in alterations to understory plant communities compared to the control and open treatments, including shifts in community composition. However, we predicted that the cut/mulch treatment would result in lower overall understory plant cover, and a shift towards dominance by perennial rhizomatous species at the expense of annuals. Further, we predicted that mulched plots would follow differing temporal trajectories of soil N availability compared to removal and reference plots. This project was conducted in collaboration with Eric Fairlee (IPM Coordinator, City of Boulder OSMP).

III. METHODS

Study area and study design

This study was conducted on the Weiser property, located on the Boulder Creek floodplain just east of 75th Street in Boulder, Colorado. The climate in the area is semi-arid and continental, with average precipitation (1970-2000) of 20.23 inches per year (NOAA ESRL, unpublished data). During the study period, precipitation was above the long term average (23.57 inches in 2014, 26.92 inches in 2015), but January to September precipitation was similar among years (20.16 inches in 2014, and 21.96 inches in 2015; Figure 1). The study site was located in an area of floodplain meadow without a cottonwood-willow overstory, immediately south of Boulder Creek and its narrow forested riparian zone. The site was invaded by Russian olive probably beginning in the late 1980's. At the time of sampling, Russian olive trees occurred in varying densities along floodplain side channels and the margins of floodplain marshes (Figure 2). In some areas of the site, there was considerable flood debris from high flood flows in September 2013. The flood debris consisted of large woody material (e.g., branches, tree trunks) as well as mud and fine organic material accumulated on the upstream side of Russian olive stems. The site was not grazed by cattle for the duration of the study, but had been grazed previously. Field work for this project was conducted in August-October, 2014 and August-October 2015 by Gabrielle Katz, Michael Denslow and Janet Hardin.

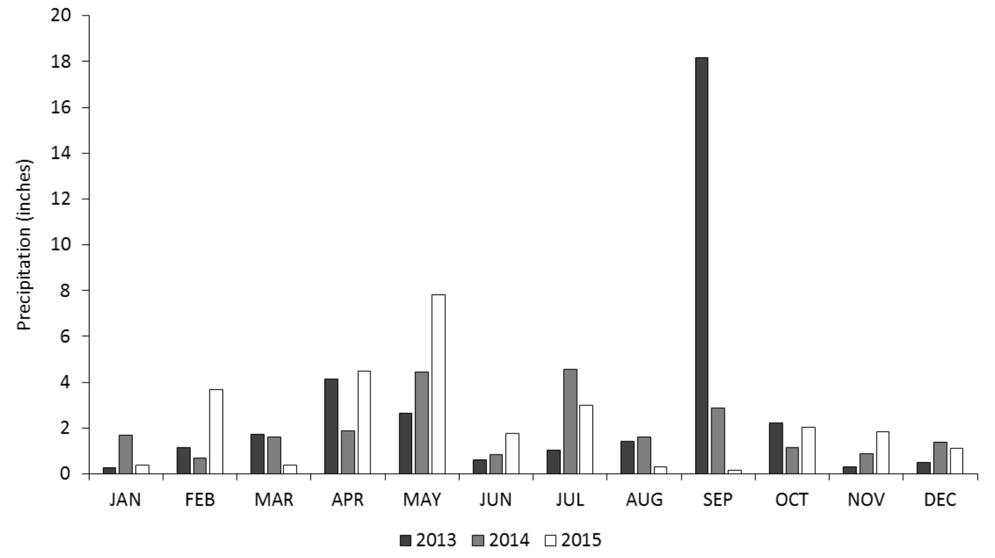

Figure 1. Monthly precipitation in Boulder, Colorado in 2013, 2014, and 2015. Data from NOAA ESRL (http://www.esrl.noaa.gov/psd/boulder/Boulder.mm.precip.html). The high precipitation in September 2013 produced high flows in Boulder Creek that flooded the study site.

Figure 2. Russian olive trees and open area, Weiser property, Boulder, Colorado. Photograph: Janet Hardin, September, 2014. Tree in the foreground is flagged for removal.

Using GIS, we established 21 random sample points at the study site (Figure 3). During field set-up and sampling, random points were omitted if they occurred within 5-10 m of a previously sampled point, or if there were no Russian olive trees within 5-10 m of the point. At each random point we established four 1m x 1m study plots. These plots represented four treatment types: Control (under a Russian olive tree that was not slated to be removed), Cut/Remove (under a Russian olive tree slated to be cut/removed without mulching), Cut/Mulch (under a Russian olive slated to be cut and chipped/mulched in place), and Open (no Russian olive overstory). Plots were permanently marked with survey whiskers, and locations were recorded with GPS.

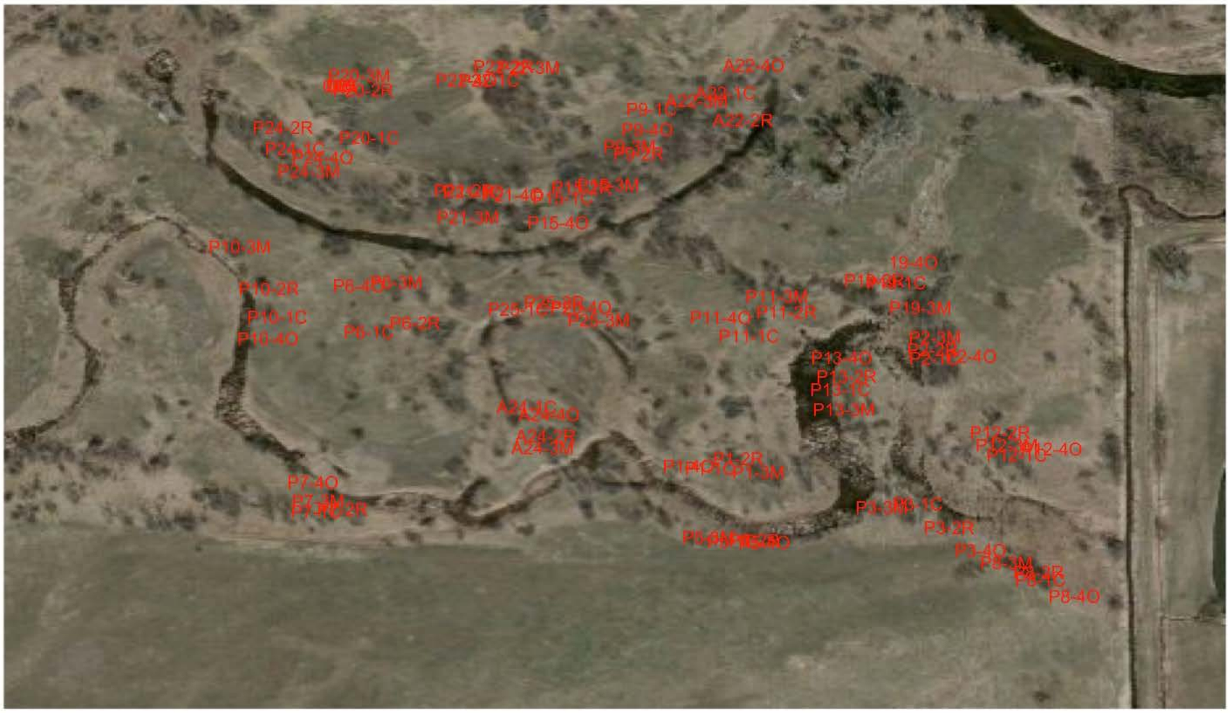

Figure 3. Study site on the Boulder Creek floodplain, showing plot locations. Plots occur in clusters of four plots per random sample point location.

Data collection and soil analysis

Vegetation data were collected in August-September 2014 (21 sample points, pre-treatment), and in August-September 2015 (20 sample points, 1-year post-treatment). One sample point (A22) was excluded from the removal treatment and post-treatment sampling in 2015 due to proximity to an eagle nest north of the study area. In each plot, we recorded the cover of each vascular plant species using cover classes. Nomenclature followed Great Plains Flora Association (1986), with Weber and Whitman (2012) providing an additional resource. We also recorded the cover of bare ground, and the cover and depth of coarse woody debris, fine woody debris, and litter in each plot, using USDA Forest Service size classes (USDA 2011). Percent overstory canopy cover was also recorded for each plot. During vegetation sampling, each plot was classified according to its predominant soil moisture status by visual inspection (1 = dry, 2 = moist ⁄ saturated, 3 = standing water to 3 cm; 4 = standing water from 3 to 10 cm).

Soil samples were collected from a sub-set of plots on October 13, 2014 (pre-treatment), and again on October 9, 2015 (1 year post-treatment). At 10 randomly chosen sample points, we collected soils from three of the treatment types (control, cut/mulch, and open), for a total of 30 soil samples. Soil samples were collected from two corners of each plot (and then combined), using a 2.2 cm x 30 cm stainless steel soil probe. Soils were refrigerated after being collected, and were brought to the EcoCore lab at Colorado State University for analysis of organic matter content and mineral nitrogen (N). NO_3-N (nitrate) + NO_2-N (nitrite), and NH_4-N (ammonium) were extracted from sieved soils with 40 ml of 2 M KCl. To determine potential rates of nitrification and net N mineralization, soils were incubated under aerobic conditions at 50% water holding capacity at 20°C for 6 weeks, followed by KCl extraction of NO_3-N + NO_2-N, and NH_4-N (Perkins et al. 2013). Net N mineralization was calculated as the difference between final and initial concentrations of NO_3-N + NO_2-N + NH_4-N. The KCl extractions were analyzed for mineral N using an Alpkem Flow Solution IV Automated wet chemistry system.

After data collection and soil sampling in 2014, Russian olive removal treatments were applied by OSMP staff in October, 2014. Treatments were applied at 20 of the 21 random points, because one point (A22) was located within 100 m of an eagle's nest on the property and could not be treated. For both cutting treatments (Cut/Remove and Cut/Mulch), Russian olive stems were cut with chain saws and herbicide was applied to the cut stumps within 5 minutes of cutting. For the herbicide, OSMP staff used 100% Garlon 4 Ultra with Hi-light (blue indicator dye) applied with a 4oz squirt applicator (Eric Fairlee, personal communication). For the Cut/Mulch treatment, Russian olive stems, branches, etc. were put through a chipper and the resulting mulch was applied with a 5-gallon bucket (approximately 2.5-3 buckets per plot) to completely cover the plot at a consistent mulch depth of 1 inch (Figure 4). Treatments were applied so as to avoid overlapping effects of one treatment on another, and buffers were maintained around each treated plot to minimize edge effects.

Figure 4. Left: Example of Cut/Mulch treatment plot. Right: photograph of Cut/Remove treatment example. Photographs by G. Katz, October, 2014.

Data analysis

We examined the effects of Russian olive on floodplain plant communities and soil conditions using the 2014 baseline data, and we used the 2015 data to examined the initial (year 1) effects of Russian olive removal on the floodplain ecosystem. We used ANOVA to compare the effects of the open (not under Russian olive) treatment with the three Russian olive treatments – control, cut/mulch and cut/remove. Response variables included total vascular plant cover, species richness, exotic and native species richness, species richness of annual, biennial and perennial species, total mineral N, and mineralization potential. For analysis of total soil N, ammonium (NH_4^+), nitrate (NO_3^-), and nitrite (NO_2) were combined into one value of inorganic soil N (µg N/g oven dried soil). Total soil N data were log transformed to better meet the assumptions of analysis of variance. We used Wilcoxan signed rank tests to examine changes within treatments between 2014 and 2015. We used SAS software to perform analysis of variance (using proc glm), with contrast statements to compare among the treatments tested, and non-parametric tests. We used non-metric multidimensional scaling (NMDS) to examine floodplain plant community composition, and we tested for differences among treatments using permutational MANOVA (PERMANOVA, Anderson 2001). Ordination and multivariate statistics were performed in R.

IV. RESULTS

A. Vegetation

Community Composition

We encountered 66 species in the study plots in 2014, and 60 species in 2015. The plant community was dominated by non-native species. The five most abundant species were *Elymus repens* (mean plot cover 36% in 2014, and 42% in 2015), *Poa pratensis* (mean cover 7.5% in 2014, 5.7% in 2015), *Bromus inermis* (mean cover 7.2% in 2014, 5.7% in 2015), *Camelina microcarpa* (mean cover 5.9% in 2014, 8.5% in 2015), and *Dipsacus follonum* (mean cover 3.4% in 2014, 6.1% in 2015). See Appendix A and B for species lists for each year.

In 2014 there was not a significant effect of treatment on plant community composition (PERMANOVA, F(3,76) = 0.8587, p = 0.68). The NMDS ordination yielded a three dimensional solution, with stress = 0.17. Despite the lack of statistical difference in community composition among treatments, the NMDS ordination diagram shows some separation of open treatment plots from the Russian olive treatments in terms of species composition (Figure 5). Table 1 lists species that occurred in plots at frequencies >0.2 in at least one treatment in 2014.

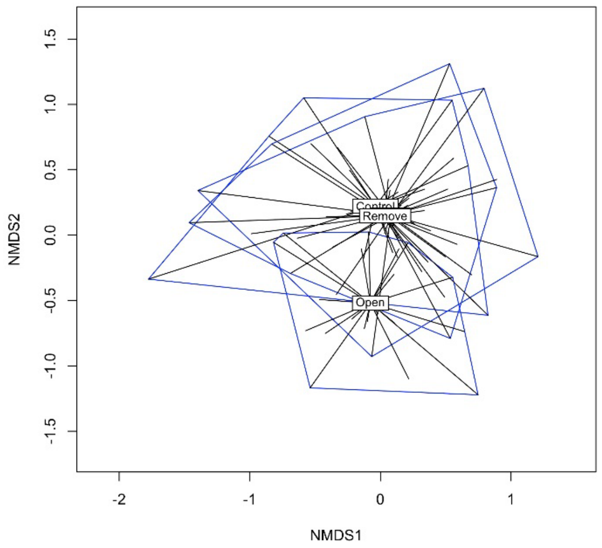

Figure 5. Spider plot of axes 1 and 2 of the three-dimensional NMDS ordination for 2014 plot data, n = 80 plots. Plots are labelled by treatment.

Table 1. Species constancy values (fraction of plots present per treatment) for study plots in 2014[1].

Species	Control	Cut/Mulch	Cut/Remove	Open
Elymua repens	0.75	0.80	0.75	0.95
Camelina microcarpa	0.75	0.75	0.65	0.55
Poa pratensis	0.30	0.25	0.30	0.35
Dipsacus follonum	0.20	0.20	0.30	0.35
Bromus inermis	0.25	0.30	0.30	*
Atriplex patula	0.20	0.25	0.20	*
Nepeta cataria	*	0.20	*	*
Cirsium arvense	*	0.20	0.25	*
Juncus articus	*	*	*	0.50
Carex praegracilis	*	*	*	0.30

Table 1 continued.

Species	Control	Cut/Mulch	Cut/Remove	Open
Convolvulus arvensis	*	*	*	0.30
Lotus tenuis	*	*	*	0.30
Carex sp.	*	*	*	0.25
Spartina pectinata	*	*	*	0.25
Distichlis spicata	*	*	*	0.20

[1] Only includes species present in >0.20 (20%) of plots in at least one treatment. Within table, * indicates frequency <0.20.

In 2015 there was a significant effect of treatment on plant community composition (PERMANOVA, F(1,76) = 1.5418, p = 0.03). The NMDS ordination yielded a three dimensional solution, with stress = 0.16. The NMDS ordination plot shows separation of the open treatment plots in terms of species composition, compared to the three Russian olive treatments (Figure 6). Table 2 lists species that occurred in plots at frequencies >0.2 in at least one treatment in 2015.

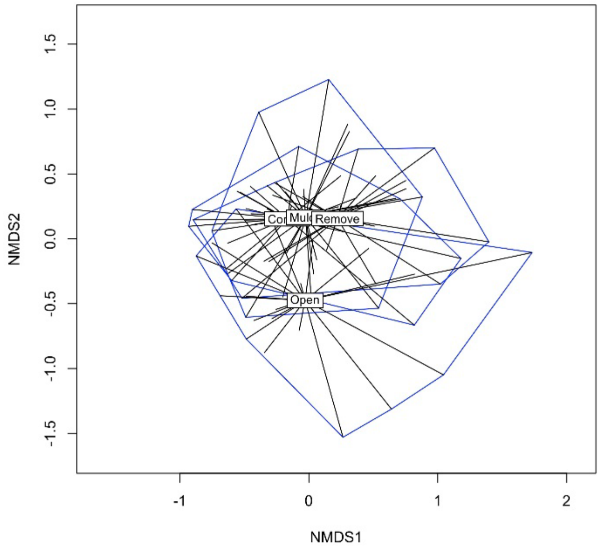

Figure 6. Spider plot of axes 1 and 2 of the three-dimensional NMDS ordination for 2015 plot data, n = 80 plots. Plots are labelled by treatment.

Table 2. Species constancy values (fraction of plots present per treatment) for study plots in 2015[1].

Species	Control	Cut/Mulch	Cut/Remove	Open
Elymus repens	0.90	0.80	0.80	0.80
Camelina microcarpa	0.60	0.65	0.65	0.40
Poa pratensis	0.30	0.20	0.25	0.40
Dipsacus follonum	0.20	0.35	0.55	0.25
Lepidium latifolium	0.20	0.10	0.20	0.10
Nepeta cataria	0.20	0.15	*	0.30
Atriplex patula	*	0.30	0.50	0.20
Bromus inermis	*	0.30	*	0.40
Cirsium arvense	*	0.40	0.60	*
Carduus nutans	*	*	0.25	*
Juncus articus	*	*	*	0.50

Table 2 continued.

Species	**Control**	**Cut/Mulch**	**Cut/Remove**	**Open**
Convolvulus arvensis	*	*	*	0.35
Symphyotrichum falcatum	*	*	*	0.30
Elymus trachycaulis	*	*	*	0.20
Lotus tenuis	*	*	*	0.20
Spatina pectinata	*	*	*	0.20

[1]Only includes species present in >0.20 (20%) of plots in at least one treatment. Within table, * indicates plot frequency <0.20.

Total Richness

In 2014, open plots had higher plot level species richness (per square meter) than plots under Russian olive (Figure 7). There was a significant effect of treatment on plot level species richness (F(3,76) = 11.36, p < 0.0001). Pair-wise contrasts indicated that open plots had higher species richness (mean = 7.25, SD = 0.72) than plots under all Russian olive treatments – Russian olive control (F(1) = 27.32, p = <.0001), Russian olive cut/mulch (F(1) = 21.35, p = <.0001), and Russian olive cut/remove (F(1) = 8.00, p = <.0001). According to pair-wise contrasts, there were no differences in species richness among Russian olive treatment types – Russian olive control (mean = 3.80, SD = 0.34), Russian olive Cut/Mulch (mean = mean = 4.20, SD = 0.38), and Russian olive Cut/Remove (mean = 4.45, SD = 0.30).

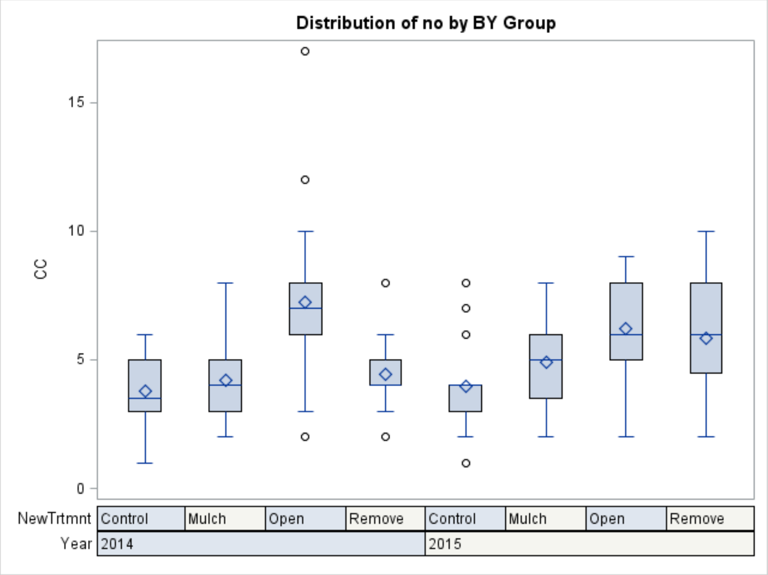

Figure 7. Plot level species richness in 2014 (left) and 2015 (right).

In 2015, there was also a significant effect of treatment on plot level species richness (F(3,76) = 5.59, p = 0.0016). However, the Russian olive removal treatments (carried out after vegetation/soil sampling in 2014) changed the richness patterns among treatment types compared to 2014. Pairwise contrasts indicated that Russian olive control plots had lower species richness (mean = 3.95, SD = 1.82) than open plots (mean = 6.20, SD = 1.94; F(1) = 13.82, p = 0.0004), and Russian olive cut/remove plots (mean = 5.85, SD = 2.25; F(1) = 9.85, p = 0.0024), but not Russian olive cut/mulch plots (mean = 4.90, SD = 1.59; F(1) = 2.46, p = 0.1207). Cut/remove plots had similar species richness to open plots (F(1) = 0.33, p = 0.5649), while cut/mulch plots had lower species richness than open plots (F(1) = 4.61, p = 0.0349).

Comparing between years, according to a Wilcoxon signed rank test there was a significant increase in species richness in the cut/remove treatment in 2015 (S = 51, p = 0.0232). However, 2014 vs. 2015 species richness was not significantly different for the control (S = 5.5, p = 0.8131), cut/mulch (S = 24, p = 0.2337), and open (S = -33, p = 0.1607) treatments.

Exotic and Native Richness

Exotic species richness did not differ among the treatments in 2014 (F(3,76) = 0.84, p = 0.4770), but there was an effect of treatment 2015 (F(3,76) = 3.07, p = 0.0326). In 2015, exotic richness was higher in the cut/remove treatment than in the open plots at p < 0.01 (F(1) = 3.33, p = 0.0718), and higher than the control treatment at p< 0.05 (F(1) = 8.74, p = 0.0041; Table 3). Exotic richness was not significantly different between the cut/remove and cut/mulch treatments (F(1) = 1.28, p = 0.2619).

There was a significant effect of treatment on native species richness in 2014 (F(3,76) = 17.99, p = <.0001) and 2015 (F(3,76) = 10.83, p = <.0001). Native species richness was highest in the open treatment plots in both years. In 2014 native richness in open plots was significantly higher than the control treatment (F(1) = 39.20, p = <.0001), the cut/mulch treatment (F(1) = 37.36, p = <.0001), and the cut/remove treatment (F(1) = 30.42, p = <.0001). None of the Russian olive treatments differed in terms of native species richness in 2014. In 2015 native richness in open plots was significantly higher than the control treatment (F(1) = 22.35, p = <.0001), the cut/mulch treatment (F(1) = 23.87, p = <.0001), and the cut/remove treatment (F(1) = 18.11, p = <.0001). None of the Russian olive treatments differed in native species richness in 2015.

Table 3. Mean (standard deviation) of exotic and native 1-m^2 plot richness by treatment, n = 20.

	Exotic Richness	Native Richness
2014:		
Russian olive control	3.15 (1.42)	0.55 (0.76)
Russian olive cut/mulch	3.50 (1.28)	0.60 (0.60)
Russian olive cut/remove	3.50 (1.32)	0.80 (0.89)
Open	3.90 (1.89)	2.65 (1.66)
2015:		
Russian olive control	3.20 (1.58)	0.70 (1.03)
Russian olive cut/mulch	4.25 (1.65)	0.65 (0.67)
Russian olive cut/remove	4.90 (2.20)	0.85 (0.81)
Open	3.85 (1.78)	2.20 (1.36)

Lifespan Richness

In 2014, there were no differences in annual (F(3,76) = 1.29, p = 0.2847) or biennial (F(3,76) = 0.03, p = 0.9922) species richness among treatments. However, there was an effect of treatment on perennial species richness (F(3,76) = 14.60, p = <.0001). Perennial species richness was higher in open plots (mean = 4.9 species per m^2, SD = 2.15) than in all Russian olive plots – control (F(1) = 31.22, p = <.0001), cut/mulch (F(1) = 32.45, p = <.0001), and cut/remove (F(1) = 22.19, p = <.0001).

In 2015, there was an effect of treatment on species richness of annual species (F(1) = 4.02, p = 0.0104), biennial species (F(1) = 4.64, p = 0.0049), and perennial species (F(1) = 9.56, p = <.0001). For annual species richness, control plots had lower species richness than all other treatment types – cut/remove (F(1) = 10.96, p = 0.0014), cut/mulch (F(1) = 6.48, p = 0.0129), and open (F(1) = 3.18, p = 0.0787). Contrary to our expectations, annual species richness was not reduced in the cut/mulch plots. Biennial species richness tended to be higher in the cut/remove (mean = 1.65, SD = 1.09) and cut/mulch treatments (mean = 1.25, SD = 0.91), than in the control (mean = 0.95, SD = 0.76) and open (mean = 0.65, SD = 0.75) treatments. For biennial species richness, open plots had lower species richness than cut/remove (F(1) = 12.71, p = 0.0006) and cut/mulch (F(1) = 4.58, p = 0.0356) plots, but not control plots. Control plots had lower biennial species richness than cut/remove plots (F(1) = 6.23, p = 0.0147). For perennial species, richness was higher in open plots than in than in all Russian olive plots – control (F(1) = 23.08, p = <.0001), cut/mulch (F(1) = 18.90, p = <.0001), and cut/remove (F(1) = 13.40, p = 0.0005).

Cover

Total plant cover was high in both years (Figure 8), and there was no effect of treatment on cover in 2014 (F(3,76) = 1.33, p = 0.2708) or 2015 (F(3,76) = 0.89, p = 0.4511). Mean and standard deviation of percent cover values were as follows: In 2014, control mean = 75.02, SD = 24.57, cut/mulch mean = 78.25, SD = 17.63, cut/remove mean = 74.12, SD = 25.08, and open mean = 87.62, SD = 27.34. In 2015, control mean = 94.15, SD = 20.68, cut/mulch mean = 99.30, SD = 23.30, cut/remove mean = 97.72, SD = 23.11, and open mean = 105.65, SD = 23.97.

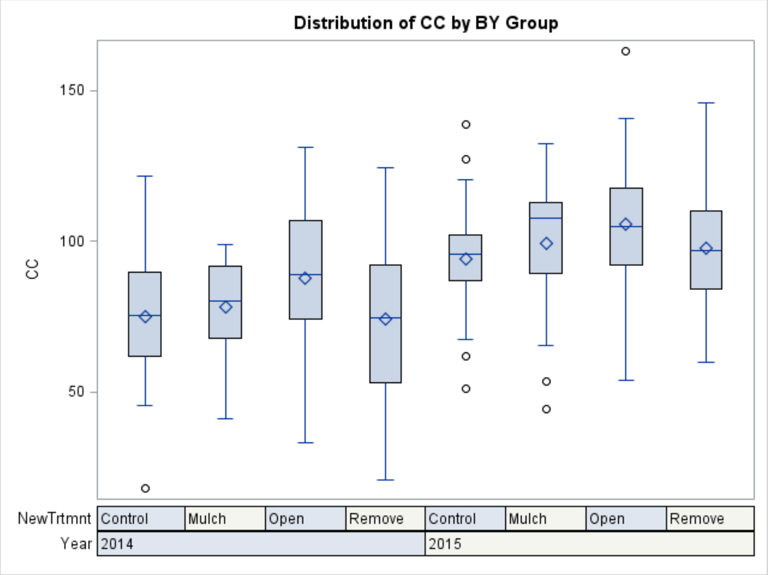

Figure 8. Total plot cover in 2014 (left) and 2015 (right).

Plot cover was higher in 2015 than in 2014 in all treatment types. According to Wilcoxon signed rank tests, cover was significantly higher in control plots (S = 67.5, p = 0.0097), cut/mulch plots (S = 76, p = 0.0032), open plots (S = 65, p = 0.0133), and cut/remove plots (S = 73.5, p = 0.0044).

B. Soil nitrogen

Total Soil N

In 2014, soil nitrogen (µg N/ g oven dried soil) was higher in Russian olive Control plots (mean = 12.00, SD = 3.64) and Russian olive Cut/Mulch plots (mean = 20.58, SD = 17.55), than in Open plots (mean = 7.66, SD = 4.92). The effect of treatment was significant (F(2,27) = 6.93; p = 0.0037, Figures 9 and 10). Contrasts indicated that both Control (F(1) = 5.00, p = 0.0339) and Cut/Mulch (F(1) = 13.66, p = 0.0010) had higher soil N than Open plots. However, Control and Cut/Mulch plots were not different from one another (F(1) = 2.14, p = 0.1555).

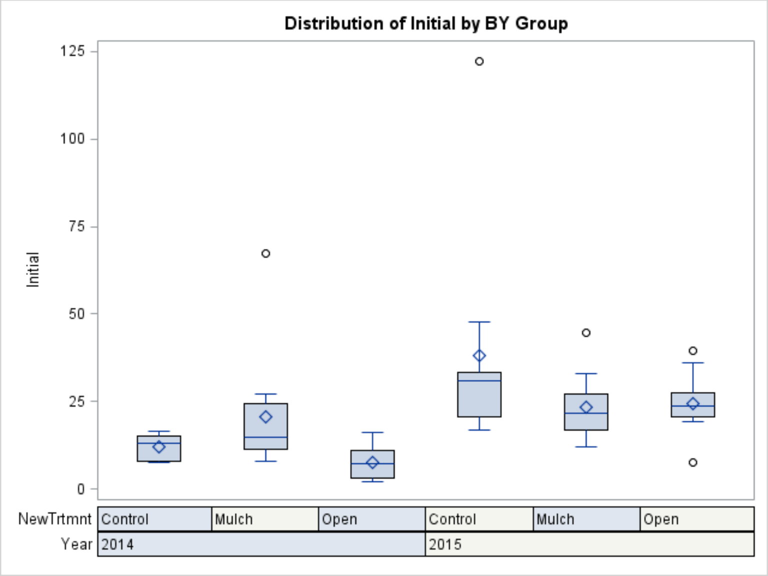

Figure 9. Soil N (µg N/g oven dried soil) in 2014 and 2015.

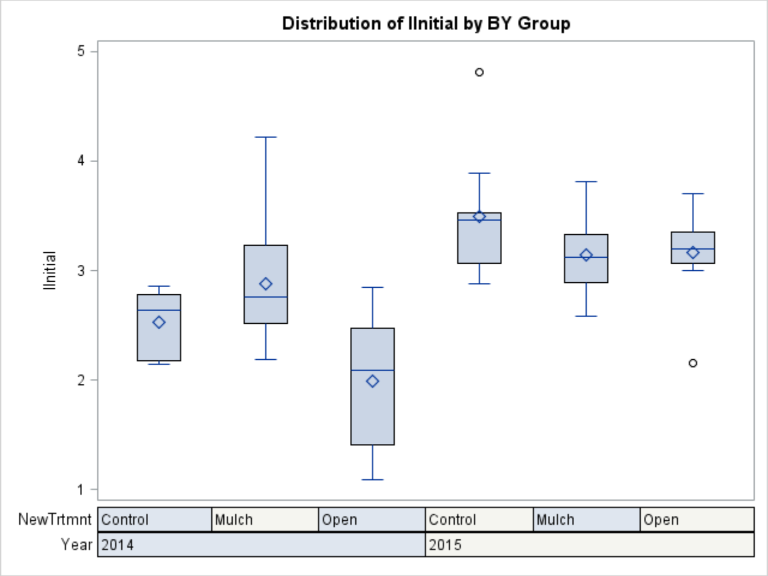

Figure 10. Log transformed values of soil N in 2014 (left) and 2015 (right). Note: soils were only collected in three of the four treatment types.

In 2015, there was not a significant effect of treatment on soil N (F(1,27) = 1.90, p = 0.1684). Cut/Mulch plots had mean soil N = 23.46 (SD = 9.41). Control plots had mean soil N = 38.19 (SD = 30.93), and Open plots had mean soil N = 24.51 (SD = 9.07). Total Soil N was higher in 2015 than in 2014 in the Control treatment (F(1,18) = 23.57, p = 0.0001) and the Open treatment (F(1,18) = 23.75, p = 0.0001), but not in the Cut/Mulch treatment (F(1,18) = 1.40, p = 0.2519).

Considering forms of soil N, levels of NH_4^+-N were substantially lower than levels of NO_3^--N for all treatments in all years (Table 4).

Table 4. Mean (standard deviation) of NH_4-N and NO_3-N (µg N/g dry soil) by treatment, n = 10.

	NH_4^+-N	NO_3^--N
2014:		
Russian olive control	0.66 (0.53)	11.34 (3.35)
Russian olive cut/mulch	0.78 (0.87)	19.80 (17.35)
Open	0.47 (0.32)	7.19 (4.90)
2015:		
Russian olive control	0.78 (0.52)	37.40 (30.86)
Russian olive cut/mulch	0.73 (0.30)	22.73 (9.34)
Open	0.70 (0.37)	23.81 (8.96)

Mineralized N

There was no effect of treatment on mineralized soil N in 2014 (F(2, 27) = 0.22, p = 0.8026) or 2015 (F(1,27) = 1.30, p = 0.2884). Mineralized soil N was higher in 2015 than in 2014 in all treatments – Control (F(1,18) = 7.25, p = 0.0149), Cut/Mulch (F(1,18) = 4.92, p = 0.0397), Open (F(1,18) = 7.24, p = 0.0150). In the Control treatment, mean mineralized soil N was 33.76 (SD = 12.29) µg N/g oven dried soil in 2014, and 69.46 (SD = 40.07) in 2015. In the Open treatment, mean mineralized soil N was 31.93 (SD = 12.07) µg N/g oven dried soil in 2014 and 50.08 (SD = 17.59) in 2015. In the Cut/Mulch treatment, mean mineralized soil N was 29.81 (SD = 15.31) µg N/g oven dried soil in 2014, and 51.73 (SD = 27.26) in 2015.

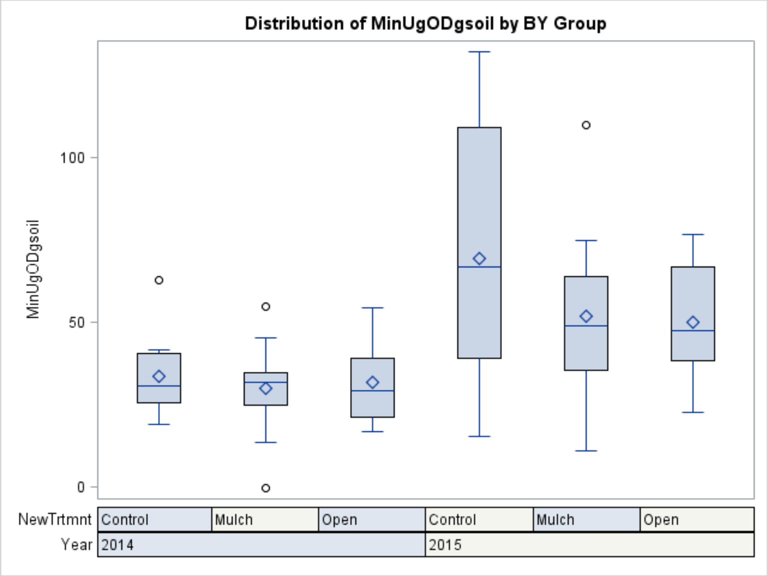

Figure 11. Mineralized soil N (µg N/g oven dried soil) in 2014 and 2015, by treatment (n-10). Note: soils were only collected in three of the four treatment types.

V. DISCUSSION

Russian olive impacts on the floodplain ecosystem

We demonstrated a significant impact of Russian olive on the floodplain ecosystem in 2014, prior to the tree removal treatments. Open plots, at least 5 m away from Russian olive canopies, had higher total species richness, perennial species richness, and native species richness than plots under Russian olive. Further, open plots had lower soil N than most Russian olive treatments. These results are consistent with prior research documenting Russian olive impacts. DeCant (2008) compared soils under sub-canopy Russian olives with those under only cottonwood canopy in riparian gallery forests on the Rio Grande, New Mexico, and found elevated soil N levels under Russian olive, as well as greater soil organic matter accumulation. In another study conducted in cottonwood gallery forests on the Rio Grande, Follstad Shah et al. (2010) found sub-canopy Russian olive leaf fall to be the strongest predictor of soil N levels across multiple study sites. Russian olive leaf litter contributed 19 percent of N entering the riparian system from leaf fall despite it comprising only five percent of litter fall mass. In a study of Russian olive impacts on riparian plant communities of the Great Plains (South Fork Republican River, Yuma County, eastern Colorado), Tuttle et al. (in review) found that plots underneath Russian olive had higher relative exotic cover (exotic/total cover), lower perennial C4 grass cover, and higher perennial forb cover, compared to reference plots outside of Russian olive canopy. These effects were stronger for Russian olive growing in open areas (as in the present study), than in the understory of the cottonwood gallery forest. Thus, our results add to the small but growing body of research documenting Russian olive impacts to riparian soils and plant communities.

Effects of Russian olive removal treatments

Removal of Russian olive altered floodplain plant communities, and effects differed somewhat between the cut/mulch and cut/remove treatments. In the first year after treatment, cut/remove plots had higher species richness than control plots, and similar species richness to open plots. In contrast, cut/mulch plots had similar species richness to control plots and lower species richness than open plots in 2015. Further, species richness increased in the cut/remove treatment in 2015 compared to 2014, but not in the cut/mulch treatment. However, much of this species richness was comprised of exotic species, which may not be a desired management outcome, and exotic species richness did not differ between the cut/remove and cut/mulch treatments. Of potential concern, biennial species richness tended to be high in both the cut/remove and cut/mulch treatments. Commonly encountered biennial species included *Arctium minus, Camelina microcarpa, Carduus nutans, Cirsium vulgare,* and *Dipsacus follonum* (all exotic species). There was not a significant effect of treatment on soil N in 2015, possibly due to increased N levels at the site overall (see below for discussion of this trend).

Certain aspects of the study site, as well as the removal treatments, should be considered when assessing the applicability of our results to other contexts. First, the site was dominated by *Elymus repens* (quackgrass), a non-native, rhizomatous, perennial grass. The abundance of this grass in all treatment types in 2014 (prior to application of the treatments) likely dampened the effect of the removal treatments on plant communities, since *E. repens* was able to spread easily into treated plots. Second, although the site experienced substantial flooding in 2013, it was essentially undisturbed during the study period, which likely limited the opportunity for species turnover in the study plots. This lack of physical disturbance was due to an absence of overbank flooding in 2014 and 2015, to the absence of livestock in the study area for the duration of the project, and to the careful application of the removal treatments, which entailed very little vehicle impact or soil disturbance.

Site level trends

Two key site level trends are apparent in our data, suggesting that environmental conditions may have been different at the site in the two sample years. First, total plant cover was higher in all treatment types in 2015 compared to 2014. Second, total soil N was higher in the control treatment and the open treatment (though not in the cut/mulch) treatment in 2015 compared to 2014. Further, mineralized soil N was higher in all three sampled treatments in 2015 compared to 2014. The increase in total plant cover in 2015 was likely due to the relatively high precipitation in May 2015 (7.82 inches vs. 4.43 inches in 2014) and June 2015 (1.76 inches vs. 0.84 inches in 2014, Figure 1) which promoted plant growth early in the growing season. The increased soil N is more difficult to explain, though we consider several possibilities below.

Conceptually, the higher levels of both total soil N and mineralized N in 2015 compared to 2014 could have been due to two factors (or to their combination) -- (1) ecological or environmental conditions in 2014 that reduced soil N, and (2) ecological or environmental conditions in 2015 that increased N levels. We do not attribute the increased site level soil N to the Russian olive removal treatments because the effect was observed in open plots (at least 5 m away from Russian olive trees), and because only a small proportion of the Russian olive trees at the site was removed, with minimal disturbance to the site overall. One factor that could have reduced soil N in 2014 was the flooding in 2013. That is, if the 2013 flood reduced soil N directly (via export of N from the site) or indirectly (via impacts to plant productivity, and/or soil microbial communities), the effect may have persisted to our sampling in 2014. Indeed, Follstad Shah and Dahm (2008) found that soil NO_3^--N and NH_4^+-N were both greater at cottonwood forest sites on the Rio Grande floodplain in the absence of flooding, and were reduced at frequently flooded sites. Reduced N levels at flooded sites may have been to N export by floodwaters in these open systems, and/or to reduced litter inputs at frequently flooded sites. Interestingly, this pattern was not observed for NO_3^--N at floodplain sites dominated by saltcedar, and NH_4^+-N was actually higher at frequently flooded saltcedar dominated sites. If the 2013 flood did suppress soil N levels in 2014, this effect appears to have been temporary, since increased levels of soil N were observed in 2015. The high rainfall in May 2015 is one factor that might have served to increase soil N in 2015. In contrast to the role of flooding in suppressing soil N on the Rio Grande floodplain, Follstad Shah and Dahm (2008) observed pulses in soil NH_4^+-N at all sites coincident with increases in soil moisture associated with springtime precipitation. It may be impossible to tease apart the possible influences on site level soil N with only two years of data.

VI. LITERATURE CITED

Anderson, M.J. 2001. A new method for non-parametric multivariate analysis of variance. Austral Ecology, 26: 32–46.

DeCant JP. 2008. Russian olive, *Elaeagnus angustifolia*, alters patterns in soil nitrogen pools along the Rio Grande River, New Mexico, USA. Wetlands, 28:896–904.

Donahue, C., Rogers. W., Siemann, W. 2006. Restoring an invaded prairie by mulching live *Sapium sebiferum* (Chinese tallow trees): Effects of mulch on *Sapium* seed germination. Natural Areas Journal 26(3):244-253.

Follstad Shah, J.J. and C.N. Dahm. 2008. Flood regime and leaf fall determine soil inorganic nitrogen dynamics semiarid riparian forests. Ecological Applications 18:771-788.

Follstad Shah, J.J., M.J. Harner, and T.M. Tibbets. 2010. *Elaeagnus angustifolia* alters soil inorganic nitrogen pools in riparian ecosystems. Ecosystems 13:46-61.

Friedman, J.M., Auble, G.T., and Shafroth, P.B. 2005. Dominance of non-native riparian trees in Western USA. Biological Invasions, 7:747–751.

Gaddis M, Sher A. 2012. Russian olive (*Elaeagnus angustifolia*) removal in the western United States: Multi-site findings and considerations for future research. Sustainability, 4:3346-3361.

Great Plains Flora Association. 1986. Flora of the Great Plains. University Press of Kansas, Lawrence, Kansas, USA.

Hultine KR, and Bush SE. 2011. Ecohydrological consequences of non-native riparian vegetation in the southwestern U.S.: a review from an ecophysiological perspective. Water Resources Research.

Katz, G.L. and Shafroth, P.B. 2003. Biology, ecology and management of *Elaeagnus angustifolia* L. (Russian olive) in western North America. Wetlands, 23(4): 763-777.

Merritt, D.M., and Johnson, J.B., 2006, Response of riparian vegetation to mechanical removal of invasive plants, RMRS Middle Rio Grande Fuels Reduction Study (FRS): Progress to date, 2005 Progress Report presented to Rocky Mountain Research Station, Albuquerque, NM, U.S. Forest Service, Rocky Mountain Research Station.

Mineau M. M., C.V. Baxter and A. M. Marcarelli. 2011 A non-native riparian tree (Elaeagnus angustifolia) changes in-stream nutrient dynamics. Ecosystems 14: 353-365.

Mineau M. M., C. V. Baxter, A. M. Marcarelli and G. W. Minshall. 2012. Invasive riparian tree reduces stream ecosystem efficiency via a recalcitrant organic matter subsidy. Ecology, 97(3): 1501-1508.

O'Meara, S., Larsen, D., Owens, C. 2010. Methods to control saltcedar and Russian olive. In: P.B. Shafroth, C.A. Brown, and D.M. Merritt (eds.). Saltcedar and Russian olive control demonstration act science assessment. Scientific Investigations Report 2009–5247. : U.S. Geological Survey. 119-136 p.

Perkins, L.B., et al. 2013. Quick start guide to soil methods for ecologists. Perspect. Plant Ecol. Evol. Syst. 15: 237-244.

Perry LG, Blumenthal DM, Monaco TA, Paschke MW, Redente EF. 2010. Immobilizing nitrogen to control plant invasion. Oecologia. 163(1):13-24

Rhoades, C. C., Battaglia, M. A., Rocca, M. E., Ryan, M. G. 2012. Short- and medium-term effects of fuel reduction mulch treatments on soil nitrogen availability in Colorado conifer forests. Forest Ecology and Management. 276: 231-238.

Reynolds, L. V., & Cooper, D. J. 2011. Ecosystem response to removal of exotic riparian shrubs and a transition to upland vegetation. Plant Ecology. 212: 1243-1261.

Shafroth, P.B., C.A. Brown, and D.M. Merritt. 2010. Saltcedar and Russian olive control demonstration act science assessment. Scientific Investigations Report 2009–5247. U.S. Geological Survey. 169 pp.

Tamarisk Coaltion, 2009. Colorado River Basin Tamarisk and Russian Olive Assessment December 2009. http://www.tamariskcoalition.org/PDF/TRO%20Assessment%20with%20Appendices%20FINAL%2012-22-09.pdf

Tuttle, G., Katz, G.L., Friedman, J.M. and Norton, A.P. *in review*. Ecological context conditions the impact of Russian olive in heterogeneous riparian ecosystems. (Ph.D. dissertation research by G. Tuttle, Colorado State University).

USDA Forest Service, Forest Inventory and Analysis National Program. 2011. Phase 3 Field Guide – Down Woody Materials, Version 5.1. Available at: http://www.fia.fs.fed.us/library/field-guides-methods-proc/docs/2012/field_guide_p3_5-1_sec25_10_2011.pdf

Weber, W. and Whitman, R.C. 2012. Colorado Flora: Eastern Slope. 4th edition. University Press of Colorado, Boulder, Colorado, USA.

Woody Invasive Control Committee. 2010. Woody invasive control plan. Appendix A of the Action Plan, Escalante Watershed Partnership.

Wolk, B. and Rocca, M. 2009. Thinning and chipping small-diameter ponderosa pine changes understory plant communities on the Colorado Front Range. Forest Ecology and Management. 257: 85-95.

VII. APPENDIX

Appendix A. 2014 species list with constancy values (fraction of plots with species present in each treatment).

	Control	Mulch	Open	Remove
Achillea millefolium	0	0	0.15	0
Agrostis scabra	0	0	0.05	0
Ambrosia psilostachya	0	0	0.15	0
Arctium minus	0.1	0.1	0	0.15
Asclepias speciosa	0.05	0.05	0.1	0.05
Atriplex patula	0.2	0.25	0.15	0.2
Bassia scoparia	0	0	0.05	0
Bromus inermis	0.25	0.3	0.1	0.3
Bromus japonicus	0	0	0.05	0
Bromus tectorum	0	0.1	0.05	0
Camelina microcarpa	0.75	0.75	0.55	0.65
Carduus nutans	0.15	0.05	0	0.1
Carex #3	0	0	0.05	0
Carex praegracilis	0.05	0	0.3	0.05
Carex sp	0	0	0.1	0.05
Carex sp1	0	0.1	0.25	0
Cichorium intybus	0	0	0.1	0
Cirsium arvense	0.1	0.2	0.05	0.25
Convolvulus arvensis	0	0.1	0.3	0.05
Dactylis glomerata	0	0	0.05	0
Daucus carota	0	0	0.05	0
Descurainia sp	0	0	0.05	0
Dipsacus fullonum	0.2	0.2	0.35	0.3
Distichlis spicata	0.1	0.05	0.2	0.05
Echinocystis lobata	0	0	0	0.05
Elaeagnus angustifolia	0	0	0	0.1
Eleocharis sp	0	0	0.1	0
Elymus trachycaulus	0	0	0	0.05
Elymus repens	0.75	0.8	0.95	0.75
Epilobium ciliatum	0	0	0.05	0
Equisetum arvense	0	0	0.05	0
Forb Sdlng	0.05	0	0.05	0
Forb2 Sdlng	0	0	0.05	0
Forb3 Sdlng	0	0	0	0.05
Gaura parviflora	0	0.05	0.05	0.05
Grindelia squarrosa	0	0	0.05	0
Holcus lanatus	0.05	0	0	0
Impatiens capensis	0	0.1	0	0.1
Juncus arcticus	0.05	0.05	0.5	0.05
Juncus confusus	0	0.05	0.05	0.1
Juncus sp	0.05	0	0.2	0.05

Appendix A continued.

	Control	Mulch	Open	Remove
Lactuca serriola	0	0	0.05	0
Lepidium latifolium	0.1	0.1	0.1	0.15
Lotus tenuis	0	0	0.3	0.05
Medicago sativa	0	0	0.05	0
Melilotus officinalis	0	0	0.05	0
Nepeta cataria	0.1	0.2	0	0.1
Pascopyrum smithii	0	0	0.15	0
Phalaris arundinacea	0.05	0	0.05	0.05
Poa pratensis	0.3	0.25	0.35	0.3
Polygonum amphibium	0.05	0	0	0.05
Polygonum douglasii	0	0.1	0.05	0
Polygonum persicaria	0.05	0	0	0
Polygonum ramosissimum	0.05	0.1	0.1	0.05
Polypogon monspeliensis	0	0	0.05	0
Rosa rubiginosa	0	0.05	0	0
Schizachyrium scoparium	0	0	0.05	0
Schoenoplectus americanus	0	0	0.05	0
Schoenoplectus pungens	0	0	0.1	0
Sonchus oleraceus	0.05	0	0.05	0
Spartina pectinata	0.1	0	0.25	0
Symphoricarpos occidentalis	0.05	0.05	0	0.05
Symphyotrichum falcatum	0	0	0.05	0
Taraxacum officinale	0	0	0.05	0.05
Thermopsis divaricarpa	0	0	0	0.05
Verbascum thapsus	0	0.05	0	0

Appendix B. 2015 Species list with constancy values (fraction of plots with species present in each treatment).

	Control	**Mulch**	**Open**	**Remove**
Achillea millefolium	0.05	0	0.15	0
Agrostis scabra	0	0	0.05	0
Ambrosia psilostachya	0	0	0.05	0
Arctium minus	0.1	0.15	0	0.1
Asclepias incarnata	0.05	0	0	0
Asclepias speciosa	0.1	0	0	0.05
Asparagus officinalis	0	0	0.05	0
Atriplex micrantha	0.1	0.05	0.05	0
Atriplex patula	0.05	0.3	0.2	0.5
Bromus inermis	0.05	0.3	0.4	0
Bromus japonicus	0	0.05	0.05	0
Bromus tectorum	0	0	0.1	0
Camelina microcarpa	0.6	0.65	0.4	0.65
Carduus nutans	0.15	0.15	0.05	0.25
Carex douglasii	0	0	0.15	0
Carex praegracilis	0	0	0.1	0
Carex sp1	0.05	0	0	0
Celtis reticulata	0	0.05	0	0
Cichorium intybus	0	0	0.05	0
Cirsium arvense	0.15	0.4	0.1	0.6
Cirsium vulgare	0.05	0	0	0.1
Convolvulus arvensis	0	0.05	0.35	0.05
Conyza canadensis	0	0	0	0.1
Dactylis glomerata	0.05	0	0	0
Dipsacus fullonum	0.2	0.35	0.25	0.55
Distichlis spicata	0	0.05	0	0
Elaeagnus angustifolia	0.05	0.15	0	0.05
Elymus elymoides	0	0	0.05	0
Elymus trachycaulus	0	0	0.2	0.1
Elytmus repens	0.9	0.8	0.8	0.8
Equisetum arvense	0	0	0.05	0
Festuca rubra	0	0	0	0.05
Forb sp	0	0	0	0.05
Gaura parviflora	0	0	0	0.05
Hordeum jubatum	0.05	0	0	0
Juncus arcticus	0.1	0.15	0.5	0.1
Juncus confusus	0.05	0	0.05	0.05
Juncus sp	0	0	0.15	0
Juncus sp2	0	0	0	0.05
Lactuca serriola	0	0.05	0	0.15
Lepidium latifolium	0.2	0.1	0.1	0.2
Lotus tenuis	0	0.05	0.2	0

Appendix B continued.

	Control	**Mulch**	**Open**	**Remove**
Melilotus officinalis	0	0	0.15	0
Nepeta cataria	0.2	0.15	0	0.3
Pascopyrum smithii	0	0.05	0.05	0
Phalaris arundinacea	0.05	0	0.05	0.05
Poa pratensis	0.3	0.2	0.4	0.25
Polygonum amphibium	0.1	0	0	0
Polygonum douglasii	0	0.1	0.1	0.1
Polygonum lapathifolium	0	0.05	0	0
Polygonum persicaria	0	0	0	0.1
Polygonum ramosissimum	0	0.05	0	0.05
Polypogon monspeliensis	0	0.05	0	0.05
Rosa rubiginosa	0	0.05	0	0
Schoenoplectus pungens	0	0	0.15	0.05
Spartina pectinata	0.15	0.1	0.2	0.05
Symphoricarpos occidentalis	0	0.05	0	0
Symphyotrichum falcatum	0	0	0.3	0.05
Thinopyrum intermedium	0.05	0.15	0.15	0.15
Verbascum thapsus	0	0.05	0	0
Verbesina encelioides	0	0	0	0.05