

Restoration Notes

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Roots of Chaparral Shrubs Still Fail to Penetrate a Geosynthetic Landfill Liner after 16 Years

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In the past, legislation in the United States (USEPA 1989) and most of Europe (Forster 1993) restricted the planting of woody shrubs and trees on landfills, largely because regulators feared that roots of woody plants would penetrate the landfill liner (Dobson and Moffat, 1995). This limitation precluded revegetating landfills with pre-disturbance ecosystems, such as chaparral, in which woody species are an important part of the vegetation. More recently, a wider range of species, including shrub and tree species, have been permitted for revegetation on landfills, but root morphology and depth remain major criteria for species selection (Calrecycle 1999; USEPA 2006).

Most previous studies across several temperate ecosystems suggest that tree roots do not penetrate landfill liners (Gillman 1989, Dobson and Moffat 1995, Robinson and Handel 1995, Handel et al. 1997, Hutchings et al. 2001). Moffat et al. (2008), however, found that tree roots in Great Britain occasionally penetrated weaker areas of a mineral landfill liner, particularly when the soil layer over the cap was < 1 m. In an earlier paper, we reported that the roots of 11 species of California coastal chaparral shrubs did not penetrate a geosynthetic landfill liner, but shrubs and trees were only 3–5 years old at the time and growth is notoriously slow in the sandy infertile soils and low rainfall conditions (Holl 2002). Here we provide an update on this study after 16 years in order to provide more conclusive guidance on whether woody chaparral species should be used in landfill revegetation.

In 1997, we established the study at the former Fort Ord Army base in the city of Marina, Monterey County, California. Maritime chaparral in this region is dominated by shrubs, including several species of manzanita

(*Arctostaphylos*) and ceanothus (*Ceanothus*), and a high diversity of annual herbs (Griffin 1978). The soils are medium-grained sands (92–96 percent sand) that are well-drained, and have low organic matter content and fertility (Holl 2002). Mean rainfall is 19 inches (475 mm) per year with high interannual variability (11 to 38 inches/275 to 957 mm; National Climate Data Center, Asheville, NC).

We planted eight shrub and one tree species separated by 1.5 m on an experimental area with a 40-mil polyethylene geomembrane liner (Poly-Flex Construction, Inc., Grand Prairie, Texas) covered with approximately 65 cm of soil in order to match regulations at the time. For more experimental design details see Holl (2002). In 2013, 16 years after planting, we used shovels and trowels to excavate 2–3 of the largest individuals of four planted species: chamise (*Adenostema fasciculatum*), sandmat manzanita (*Arctostaphylos pumila*), Monterey ceanothus (*Ceanothus rigida*), and coast live oak (*Quercus agrifolia*). We also excavated three naturally colonizing individuals of Torrey pine (*Pinus torreyana*) (> 4 m in height and > 10 cm diameter at breast height), as well as one particularly large (2 m height, ~10 m² aerial) cover coyote brush (*Baccharis pilularis*) on the landfill liner. All originally-planted coyote brush had senesced. We followed all major roots of each plant until they became 2–3 mm diameter. We also excavated three Torrey pine and one coast live oak tree in maritime chaparral adjacent to the experimental area to compare taproot morphology off the landfill liner.

No roots of any of the six species penetrated the landfill liner, although roots of most species excavated often reached the liner, consistent with our earlier results (Holl 2002). Roots of four of the species, chamise, sandmat manzanita, coyote brush, and Monterey ceanothus usually had a taproot that split into 2–5 medium sized (> 2 cm diameter) roots that primarily spread laterally in the top 30 cm of soil. Occasionally, the roots would grow deeper and run along the liner for up to 4 m. The lateral extent of the excavated roots (> 2–3 mm diameter) was generally only 1–2 m longer than earlier excavations (Table 1, Holl 2002).

Both Torrey pine and coast live oak have thick taproots; for example in Torrey pine the main taproot of all individuals was 8–10 cm diameter immediately below the soil surface. On the liner, the taproots of both species abruptly turned laterally ~20–40 cm below the soil

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Table 1. Results of excavations of chaparral shrubs and trees on landfill liner.

Species	Number excavated	Max. roota depth (m)	Max. lateral root spread (m)
chamise (<i>Adenostema fasciculatum</i>)	2	liner	3.0
sandmat manzanita (<i>Arctostaphylos pumila</i>)	2	0.6	4.5
coyote brushb (<i>Baccharis pilularis</i>)	1	liner	5.0
Monterey ceanothus (<i>Ceanothus rigidus</i>)	3	liner	5.3
Torrey pineb (<i>Pinus torreyana</i>)	3	liner	> 7.0
coast live oak (<i>Quercus agrifolia</i>)	2	0.4	3.2

^aThe liner was at 0.65–0.70 m depth on the test cap.

^bNaturally colonizing species.

surface (Figure 1) and split into several smaller roots which occasionally reached, but did not penetrate, the liner. Off the experimental area, the tap roots grew straight down (Figure 1) and were all > 7 cm diameter at 65 cm depth (a comparable depth to the liner). Our prior excavations of shrubs off the landfill cap (Holl 2002), as well as published information (Hellmers 1955, Kummerow et al. 1977, Kummerow and Mangan 1981), suggest that most chaparral shrub roots grow at less than 60 cm depths, but that taproots of some chaparral species can extend well beyond the 60–70 cm depth of the liner when grown in deeper soils.

Differences in root morphology on landfills versus deeper soils are common and may affect growth (Handel et al. 1997, Holl 2002). Chaparral soils have highly variable depths to base rock (Kummerow et al. 1977) so it is not surprising that chaparral species have evolved a variable

root morphology. One caveat is that the growth rate on this site is clearly nutrient limited (Holl 2002) and, therefore, studies are needed on more fertile soils, as coastal chaparral is found across a range from low to medium fertility (Callaway and Davis 1993, Vasey 2012).

Our results demonstrate that a number of species of chaparral shrubs are able to survive and grow on landfills without constituting a threat to the integrity of a 40-mil geosynthetic landfill liner and, therefore, do not need to be excluded from landfill revegetation efforts nor removed when naturally colonizing landfill sites. This result agrees with most previous research on landfills suggesting that woody plant roots rarely penetrate intact liners and that woody plant roots have fairly plastic morphology allowing them to adjust to their immediate microenvironment (Handel et al. 1997, Parsons et al. 1998). The 60–70 cm soil layer used in our study was sufficient for chaparral shrub roots, but greater depths of soil overlaying the liner may be needed for forested systems (Moffatt et al. 2008).



Figure 1. Taproot of ~15-cm diameter Torrey pine (*Pinus torreyana*) trees growing on (left) and off (right) a landfill liner. Note that the taproot of the tree on the liner turns abruptly left and splits into several smaller roots, whereas the taproot of the tree growing off the liner is straight.

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Effects of Oak Woodland Restoration Treatments on Sapling Survival and Tree Recruitment of Oaks in an Upland Mesic Oak-dominated Forest

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Oak-dominated (*Quercus* spp.) woodlands, defined here as mixed woody and diverse herbaceous plant communities with > 70% tree canopy containing few non-oaks (see also Taft 1997), comprise a small fraction of the area in North America occupied prior to fire suppression. Historically, frequent low intensity fires maintained oak-dominated forests, woodlands, and savannas across the

Midwest and interior South of the United States (Anderson and Bowles 1999, Fralish et al. 1999, Heikens 1999, Brewer 2001, Van Lear 2004, Ruffner and Groninger 2006). Fire exclusion in the 20th century enabled fire-sensitive hardwoods to colonize previously fire-maintained oak woodlands (Hart et al. 2008, Nowacki and Abrams 2008). These tree species produced a more closed canopy (> 90%) than the historically open, sparse canopies of oak woodlands and savannas (Bowles and McBride 1998), leading to widespread oak regeneration failure and losses of groundcover plant diversity (Abrams 1992, Bowles and McBride 1998). These diverse oak communities are rare ecosystems (Anderson and Bowles 1999) and are thought to occupy less than 0.02% of the area in North America they occupied before fire exclusion (Nuzzo 1986).

One challenging aspect of oak woodland management and restoration is identifying prescribed fire regimes that will effectively promote oak regeneration and maintain groundcover plant diversity. Effective methods for promoting oak regeneration involve opening the canopy sufficiently to foster growth of an existing oak seedling/sprout layer and then, when (or if) necessary, implementing fire in such a way as to reduce competition with non-oak saplings but not cause excessive oak sapling mortality (Loftis 1990, Kruger 1997, Brose et al. 1999, Albrecht and McCarthy 2006, Iverson et al. 2008, Cannon and Brewer 2013). It remains unclear, however, whether the fire regimes necessary to maximize oak regeneration are compatible with the maintenance of groundcover plant diversity in oak woodlands. To the extent that fire regimes that promote oak regeneration conflict with the maintenance of groundcover plant diversity, practitioners must either implement fire regimes that create a mosaic of oak regeneration patches interspersed with patches of diverse groundcover vegetation or they must identify fire regimes that forge a compromise between oak sapling survival and the maintenance of groundcover plant diversity.

This study examined the effects of frequent fires proven to promote groundcover plant diversity in a mesic oak-dominated system (Brewer and Menzel 2009, K.S. Spiegel and J.S. Brewer, University of Mississippi, unpubl. data) on the survival of established oak sapling recruits and on oak tree recruitment. I tested two hypotheses: 1) frequent (biennial) burning of an established sapling layer in persistent gaps reduces sapling densities, including those of upland oak species; and 2) frequent burning prevents recruitment of tree-sized individuals from saplings (oak and non-oak).

In 2003, I established an oak woodland restoration experiment at Strawberry Plains Audubon Center (SPAC) in the loess plains of north-central Mississippi, a region characterized by gently rolling hills with moderately fertile, mesic silt and sandy loams in the uplands and floodplains. The primary objectives of the restoration experiment (i.e., the reference model; *sensu* Clewell and Aronson 2013) were 1) to increase the abundance and diversity of open oak

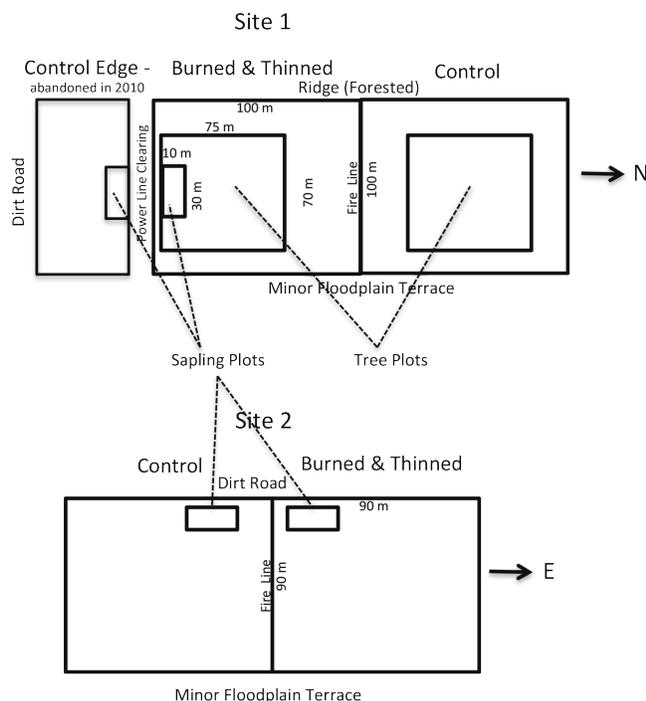


Figure 1. Diagram of the two sites at Strawberry Plains Audubon Center, MS, USA, showing the layout of sapling and tree plots and the treatment arrangement and description. Sapling and tree plots at Site 1 were established in 2003; first burned and thinned in 2004; subsequently burned in 2006, 2008, 2010, and 2012. Sapling plot at the edge of a control area was damaged in 2009 and abandoned in 2010. Sapling plots at Site 2 were established in 2007; first burned and thinned in 2008, subsequently burned in 2010 and 2012. Tree plots (not shown) at Site 2 were established in 2012.

woodland groundcover vegetation and 2) to restore natural regeneration of upland oak species (see Brewer 2001 and Surrrette et al. 2008 for evidence that upland communities in this region were historically open oak woodlands). The current study examines the latter objective.

The experiment was a paired design (adjacent 1 ha treated and control areas) replicated at each of two upland mesic sites several kilometers apart [site 1 (34°49'60"N, 89°28'32"W; site 2 (34°49'52"N, 89°27'17"W) Figure 1]. Both sites contain a mixture of silt loam alfisols and sandy loam ultisols, with silt loam predominating at site 1, and sandy loam predominating at site 2 (Morris 1981, Maynard and Brewer 2013). Both sites contained closed-canopy, mature (100+ years old) oak-hickory-gum forests, long protected from fire.

At each site, I established a single 10 × 30 m sapling plot at an open, forest edge of both the treated and control areas (Figure 1) to ensure adequate numbers of established saplings (and seedlings) of both oaks and non-oak species (Rebertus and Burns 1997). At the beginning of the study, overstory canopy cover averaged 74.7% (± 2.1%) within these forest-edge sapling plots. In addition, I established a

larger 75 × 70 m tree sampling plot (inclusive of the sapling plot) in the treated and control areas at site 1 (Figure 1).

Treatments included thinning and prescribed burning. I thinned fire-sensitive non-oak tree species in the treatment areas beginning in 2004 at site 1 and 2008 at site 2. I also thinned overstory oaks that occurred at higher densities today because of fire exclusion [e.g., Southern red oak (*Quercus falcata*), Brewer 2001]. I killed overstory trees by girdling combined with Pathway herbicide application (picloram and 2-4-D; DowAgrosciences) and left them in place. By 2012, canopy cover in the sapling plots had decreased 7.5% (± 1%) in the treated areas at both sites and increased 15.1% in the control area at site 2. Overall, the tree sampling plots exhibited an average overstory canopy cover of 87.5% in the treated areas in 2012 (down from an average of 95% at the beginning of the study) and 98% in the control areas in 2012, up from 94% at the beginning of the study.

Chad Pope (SPAC staff ecologist), several volunteers, and I conducted the prescribed fires within the treatment area at site 1 in September 2004 and October 2006, and at both sites in July 2008, April 2010, and March/April 2012 in accordance with permitting guidelines set forth by the Mississippi Forestry Commission. The sapling plot at the edge of a control plot at site 1 was damaged by expansion of the power-line clearing to the south in 2009 by power-line maintenance crews. For this reason, I could provide no data on long-term changes in sapling density in the control sapling plot at site 1.

Immediately following the initial establishment of each 10 × 30 m sapling plot, I counted and identified to species all saplings (i.e., all stems of all woody species 1 m or greater but less than 10 cm diameter at 1.5 m height). I quantified the immediate effects of the spring 2012 fires on saplings based on counts associated with the September 2011 and July 2012 censuses. I quantified tree recruitment at site 1 by identifying to species all live stems > 10 cm dbh within each 75 × 70 m tree plot and marking them with numbered aluminum tags in summer 2003 and then revisiting the tree plots in summer 2013 to locate tree recruits.

Between the beginning of the treatments and 2011 (a year without fire), repeated fires appeared to reduce oak sapling densities at both sites, whereas their effects on non-oak saplings were not consistent between sites. At site 1, a similar decrease in sapling density was apparent for both upland oak saplings and non-oak saplings [from 13 to 4 (69%) for upland oaks; from 35 to 24 (31%) for non-oak saplings; goodness-of-fit $\chi^2 = 2.79$, $p = 0.1$, $df = 1$]. At site 2, sapling density of all species combined increased to a similar extent in both the treated plot (13%) and the control plot (30%); goodness-of-fit $\chi^2 = 0.72$, $p = 0.4$, $df = 1$. However, most of the increase in the treated plot resulted from a 65% increase in stems of non-oak saplings (from 23 in 2007 to 38 in 2011). Densities of upland oak saplings in the treated plot decreased 27% (from 30 in

Table 1. Counts of trees within the 75 x 70 m treated and control tree plots in 2003 and 2013 at Site 1. Upland oaks include black oak (*Quercus velutina*), blackjack oak (*Q. marilandica*), post oak (*Q. stellata*), scarlet oak (*Q. coccinea*), and Southern red oak (*Q. falcata*). Non-oaks include American beech (*Fagus grandifolia*), black cherry (*Prunus serotina*), black gum (*Nyssa sylvatica*), Eastern redcedar (*Juniperus virginiana*), flowering dogwood (*Cornus florida*), mockernut hickory (*Carya tomentosa*), red maple (*Acer rubrum*), sweetgum (*Liquidambar styraciflua*), white ash (*Fraxinus americana*), and winged elm (*Ulmus alata*).

	Treatment Plots			Control Plots				
	Density 2003	Recruits	Deaths	Density 2013	Density 2003	Recruits	Deaths	Density 2013
Upland Oaks	65	0	20	45	49	2	6	45
White Oak (<i>Quercus alba</i>)	11	0	6	5	8	0	3	5
Non-Oaks	85	1	64	22	101	34	11	124
Total	161	1	90	72	158	36	20	174

2007 to 22 in 2011). Hence, the proportional change in saplings differed significantly between oaks and non-oaks in the treated plot at site 2 (goodness-of-fit $\chi^2 = 9.7$, $p < 0.01$). Sapling densities of both upland oak species and non-oak tree species increased to a similar extent between 2007 and 2011 within the control plot at site 2 (from 40 to 49 for upland oaks; 29 to 41 for non-oak trees; $\chi^2 = 0.46$, $p = 0.5$).

The prescribed fires in spring 2012 appeared to reduce sapling densities to a greater extent in non-oaks than in oaks. In the treated plot at site 1, the 93% reduction in non-oaks (from 28 to 2) was significantly greater than the lack of change in oaks (4 to 4; $\chi^2 = 16.1$, $p < 0.01$). In the treated plot at site 2, the 64% reduction in non-oaks (from 52 to 16) was significantly greater than the 14% reduction in oaks (from 22 to 19; goodness-of-fit $\chi^2 = 10.09$, $p < 0.01$, $df = 1$). In the control plot at site 2, densities of both oaks and non-oaks increased slightly to a similar extent (from 49 to 50 in oaks and from 54 to 56 in non-oaks; $\chi^2 < 0.01$, $p = 0.93$, $df = 1$).

There was minimal oak tree recruitment at site 1. I observed only a single tree recruit (a mockernut hickory) in the treated plot at site 1, which contained 161 trees in 2003 (Table 1). In contrast, I observed 36 tree recruits, of which 34 were non-oaks, in the control plot, which contained 158 trees in 2003 (Table 1). As a result of the higher recruitment of non-oaks (and slightly higher mortality of oaks) in the control plot, the density of non-oak trees increased significantly relative to oak trees in the control plot (Table 1; goodness-of-fit $\chi^2 = 4.06$, $p = 0.04$, $df = 1$).

Frequent prescribed burning in oak woodlands is necessary to meet important management goals such as increased herbaceous plant diversity and reduced fuel loads (Bowles and McBride 1998, Brewer and Rogers 2006, Brewer and Menzel 2009). The current study showed, however, that despite oak saplings showing greater initial resistance to fire than non-oak saplings, repeated biennial burning of an established sapling layer in an upland mesic system reduced oak sapling density as much or more than non-oak sapling density. In addition, repeated fires did not increase

oak tree recruitment. To permit oak regeneration at SPAC, fire regimes other than biennial burning in the spring need to be considered (e.g., variable fire frequencies or seasons; Rebertus et al. 1993). Simply reducing fire frequency runs the risk of increasing the abundance of non-oak saplings, which could negatively impact groundcover plant diversity.

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Augmenting Populations of Spalding's Catchfly (*Silene spaldingii*) Populations in Northwest Montana

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Spalding's catchfly (*Silene spaldingii*, Caryophyllaceae) is considered threatened throughout its range in the Columbia Plateau under the U.S. Endangered Species Act (USFWS 2007). Many populations, especially in the center of the range are small and will need to increase in size in order to attain recovery. Natural recruitment of Spalding's catchfly is sporadic at some sites (Lesica 1997), and inbreeding may predominate in small populations (Lesica 1993, Baldwin and Brunsfeld 1995), both factors that can lead to lowered population growth. Human-mediated population augmentation can result in more rapid population growth by directly adding individuals and indirectly reducing inbreeding depression (Guerrant 1996).

Transplanting seedlings is often a more successful reintroduction strategy than sowing seeds directly (Guerrant and Kaye 2007, Reckinger et al. 2010, Godefroid et al. 2011, Albrecht and Maschinski 2012). Unfortunately little is known about establishing nursery-grown stock of Spalding's catchfly. Cold-stratification enhances germination of catchfly seed (Lesica 1993), but protocols for post-germination culture have not been developed. The purpose of this study was to develop restoration protocols for Spalding's catchfly usable in Montana and adaptable in other portions of the species range. In particular, we determined how five factors affect survival and growth of nursery-grown stock in the field: (1) age of outplanted seedlings; (2) soil type used in culture; (3) watering in the field; (4) type of nursery container; and (5) date of outplanting.

Spalding's catchfly is a long-lived iteroparous herb with one or few vegetative or flowering stems arising from a caudex surmounting a long taproot (Hitchcock and Maguire 1947). Plants flower in late June through August

and set seed in August and September, depending on location. Rosettes are formed the first years after germination after which vegetative stems are produced. Vegetative and flowering plants emerge in mid- to late-May and senesce in September. Spalding's catchfly demonstrates prolonged dormancy in which plants do not appear or have only ephemeral, above-ground vegetation for one or more consecutive summers (Lesica 1997). Prolonged dormancy of germinated plants is common, with bouts of dormancy usually lasting 1–2 years (Lesica 1997, Lesica and Crone 2007).

We conducted our study on Wildhorse Island at the southwest end of Flathead Lake, 20 km north of Polson in Lake County, Montana, U.S. (47° 50' N, -114° 12' W) at 985 m in elevation. Vegetation at the study site is grassland dominated by rough fescue (*Festuca campestris*), Idaho fescue (*F. idahoensis*) and bluebunch wheatgrass (*Agropyron spicatum*) with scattered ponderosa pine (*Pinus ponderosa*). Climate is semi-arid with cold winters and hot summers. Mean annual precipitation is 382 mm and mean July maximum and January minimum temperatures were 28.4°C and -6.7°C respectively at Polson (WRCC 2013). Most of the island is natural area in Wildhorse Island State Park.

We collected seeds of Spalding's catchfly for all experiments from the Dancing Prairie population 160 km north of our study site. Seeds were placed in cold stratification in November, and germinants were obtained in mid-January to mid-February. Germinants were raised in potting soil with native grassland inoculum, first in a greenhouse followed by an outdoor garden. We planted seedlings at 1-meter intervals along four permanent 50-meter transects in an area that currently supports a small population of Spalding's catchfly. We mapped survivorship in all four transects between mid-June and early July, 2010 through 2013. Prolonged dormancy (see above) made it difficult to unambiguously assign presence or absence of Spalding's catchfly transplants across all years. Any plant that was present in at least one year between 2010 and 2013 was assumed to be a survivor. Plants observed to be damaged by rodents or ungulates were eliminated from analyses. We used Fisher's exact test of association (GraphPad Software 2013) to determine whether there was a difference in Spalding's catchfly survival across treatments. We conducted three experiments begun in three separate years: 2008, 2009, and 2011.

Experiment 1 (2008)

Spalding's catchfly germinants were transplanted into pots 9 cm wide and 11 cm deep (hereafter 4-inch pots) in mid-February. In May, 96 plants were stratified into two size classes and randomly transplanted into 4-inch pots with one of two potting media: potting soil (Sunshine Mix # 1, Sun Gro Horticulture, Agawam, MA: *Sphagnum*, perlite, lime wetting agent) and compost (humus, worm castings,

chicken and bat manure, kelp meal, lime). We transplanted the 96 seedlings on June 9 into the study site, randomly assigned to one of three watering treatments: (1) no water, (2) water three times, (3) water six times at weekly intervals. Each plant scheduled for watering received 1 liter of water.

Six of the Spalding's catchfly plants were dug out by rodents, deer or sheep in the summer of 2008. Twenty-six of the remaining 90 plants survived. Plants that received water were not more likely to have survived than those that did not ($p = 0.62$), and there was no difference in survival among the watering treatments ($p = 0.99$). There was a tendency for plants grown in compost to survive better, but this trend was not statistically significant ($p = 0.24$).

Experiment 2 (2009)

Forty-nine Spalding's catchfly germinants from 2008 and an additional 49 germinants obtained in a similar manner in 2009 were grown in potting soil in 10-cm pots. We planted the 49 first-year and 49 second-year catchfly seedlings on June 1. Plants were randomly assigned to one of two watering treatments: (1) water three times at weekly intervals or (2) water six times at weekly intervals. Waterings were performed on June 10, June 20, June 27, July 4, July 11, and July 18. Each plant scheduled for watering received 1 liter of water at each watering. Second-year plants that were starting to bolt at the time of planting were pinched back. Each second-year plant scheduled for watering received 1 liter of water.

Second-year plants were much larger than first-year plants at the time of planting. Four of the Spalding's catchfly plants were dug out by rodents or ungulates after planting in 2009. Of the remaining 94 plants, only 12 survived. There was no difference in survival among the different watering treatments or between the two age classes ($p = 0.99$).

Experiment 3 (2011)

Ninety-nine germinants were transplanted into conetainers (25 cm long with 6 cm-diameter openings at the top) in February. Thirty-three randomly-chosen plants were planted into the field site on each of three dates: May 2, May 30, and October 4. Plants from each planting treatment were placed along the transect lines at 3-meter intervals and given 1 liter of water. Each plant from the May plantings was also watered with 1 liter of water on June 5 and again on June 12. The October plants were not given additional water after planting. Plants transplanted into the field in early October were large and dormant or nearly so.

Fifty-one of the 99 plants survived. There was no difference between transplants made in early May and late May ($p = 0.99$), but there was a tendency for spring transplants to have survived better than fall transplants, but this difference was not significant ($p = 0.21$).



Figure 1. Six-month Spalding catchfly seedling in a 25 cm container. Note the white taproot reaching almost to the bottom of the plug.

Across experiments (2008, 2009, and 2011)

Survival of Spalding's catchfly planted in 2008 (29%) was greater than for those planted in 2009 cohort (13%) ($p = 0.01$). Survival Spalding's catchfly grown in conetainers and planted in 2011 (51%) was greater than that for either the 2008 cohort ($p = 0.002$) or the 2009 cohort ($p < 0.001$).

Our results provide no support for a survival advantage of older transplants or providing supplemental water; however, survival was low across treatments so power to detect differences was curtailed. Compost may be a better growth medium than potting soil, but the difference in survival between the two was small and statistically non-significant. May planting fared better than October planting; however, the small non-significant difference between spring and fall planting occurred in a year with above-average spring precipitation, and results may have been different in an average or dry year.

Differences in survival among years are likely to have been caused by either weather and/or planting conetainers. Spring precipitation (April–June) was above average in 2008, well below average in 2009 and above average again in 2011 (WRCC 2013, Table 1). Low survival of Spalding's catchfly transplants in 2009 may well have been due to the dry spring, although additional watering

Table 1. Spring (April–June) precipitation and percent Spalding's catchfly survival for the three experimental planting years.

Year	Spring precipitation	Survival
2008	152 mm	29%
2009	55 mm	13%
2011	165 mm	51%

had no discernable effect in either 2008 or 2009. Spring precipitation was similar in 2008 and 2011, so the difference in survival between these two years was more likely due to the use of 25 cm conetainers in 2011 as opposed to the 10 cm deep pots employed in 2008. *Silene spaldingii* develops a long taproot which may reach the bottom of the 25 cm conetainer after only six months of growth (Lesica and Divoky observations; Figure 1). Using a 10 cm deep planting conetainer likely curtailed proper root growth and resulted in the plant's inability to obtain adequate moisture during the latter part of the growing season.

We recommend that first-year Spalding catchfly germinants be transplanted into compost-filled conetainers at least 25 cm in length in January or February. Conetainer-grown plants should be transplanted into the field in early spring if a wet spring is anticipated. Otherwise it may be better to plant in mid-autumn to avoid drought-stress. Plants should be watered in upon planting, but supplemental watering is probably not necessary unless the post-planting weather is particularly dry. These recommendations are for populations in northwest Montana, but are likely applicable, perhaps with some modification, to other portions of Spalding's catchfly's geographic range.

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Effect of Season and Number of Glyphosate Applications on Control of Invasive Mexican Petunia (*Ruellia simplex*)

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Effective control measures for invasive species are particularly needed in forested wetland-urban interface locales. Floodplain forest communities occupy a unique niche as a transition between upland and aquatic ecosystems and support botanically rich vegetation communities compared to adjacent habitats (Nilsson and Svedmark 2002). However, urban-bordered floodplain forests are particularly vulnerable to invasion by aggressive non-native species due to alterations in flooding regime, which may then favor invading non-native species (Predick and Turner 2008). Propagule pressure from invasive plant species is also especially high in the wetland-urban interface due to runoff from urban landscape sources (Loewenstein and Loewenstein 2005).

Mexican petunia, (*Ruellia simplex*, Syn. *R. brittoniana*) is a commonly cultivated ornamental herbaceous perennial. Environmental tolerance and abundant seed production have contributed to its spread from urban landscapes into natural areas. It has been documented in parts of Texas, Louisiana, Georgia, Alabama, Mississippi, and South Carolina (USDA-NRCS 2012a), and vouchered in 29 Florida counties (Wunderlin and Hansen 2011). The Florida Exotic Pest Plant Council lists Mexican petunia as a Category 1 invasive species defined as: “altering native plant communities by displacing native species, changing community structures or ecological functions, or hybridizing with natives” (Florida Exotic Pest Plant Council 2011).

Preliminary research on Mexican petunia in natural areas suggests herbicides may provide control of Mexican petunia (Hupp et al. 2009). When herbicide was applied twice, 120 days apart, all five herbicides evaluated resulted in low percent cover (< 0.5%) 183 days after initial treatment (see Wiese et al. 2013). No work has been done, however, to address the efficacy of treatment as a function of application season, or to determine if additional glyphosate treatments are necessary to achieve optimal control of Mexican petunia. Preliminary research found that locations consisting primarily of Mexican petunia cover (75% or greater) pre-treatment shifted to $\geq 50\%$ non-Mexican petunia composition within 6 months of treatment (Hupp et al. 2009) suggesting the potential for native species recovery following control. Establishment of native species likely suppresses further Mexican petunia invasion; Hupp (2007) found that survival of young Mexican petunia seedlings was reduced when native vegetation was present, compared to bare soil.

Our objectives were to evaluate the effects of glyphosate application season and number of applications on 1) control of Mexican petunia, and 2) species composition and quality of resulting post-treatment plant cover.

The study site was located at Paynes Prairie State Preserve, Alachua County, Florida (29°37'21.7" N, 82°19'20.8" W). The site was a bald-cypress (*Taxodium distichum*) dominated floodplain forest habitat with dense stands of Mexican petunia in the herbaceous vegetation layer. The soil was predominantly from the mulat sand (loamy, siliceous, subactive, thermic Arenic Endoaquults) series (USDA-NRCS 2012b). Six 3 × 3 m plots were randomly located on both sides of a branch tributary bisecting the bald-cypress preserve area (12 plots total). Each plot was divided into four 1.5 × 1.5 m subplots designated by permanent markers.

Herbicide application treatments were applied 0, 1, 2 or 3 times to each subplot in one of two application initiation seasons (fall or spring-initiated application). Percent cover of each species present was measured using a modified Mueller-Dombois scale (0 = 0 %, 1 = <1 %, 2 = 1–4 %, 3 = 5–24 %, 4 = 25–49 %, 5 = 50–74 %, 6 = 75–94 %, 7 = 95–100 %; Mueller-Dombois and Ellenberg 1974).

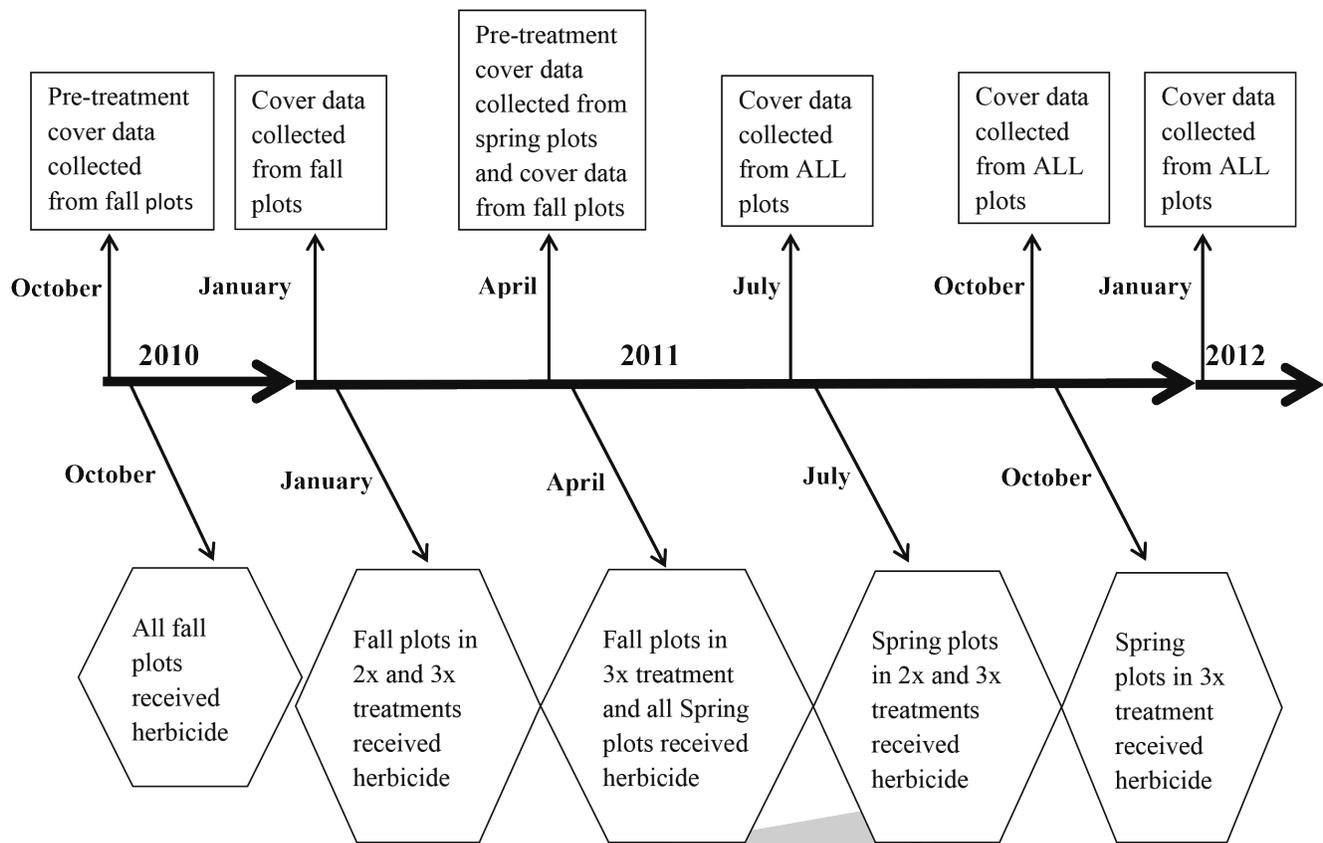


Figure 1. Timeline of herbicide applications (hexagons) and collection of aboveground cover data (squares). Fall-initiated glyphosate treatments began in October 2010. Spring-initiated glyphosate treatments began in April 2011. Plots were treated with herbicide either 0 times (control), once (1x), twice (2x), or three times (3x) over a 1-year period. Squares denote monitoring activities and polygons denote treatment applications.

Percent cover was measured before the first glyphosate application and every three months thereafter, with the final data collection occurring three months after the third spring glyphosate application (Figure 1). Pre-treatment percent plant cover data were collected in October 2010 and the first fall season herbicide application was applied immediately after (Figure 1). Spring pre-treatment plant cover data were collected in April 2011. As in the fall, pre-treatment data collection was immediately followed by the first spring season herbicide application (Figure 1). A foliar application of a 2% solution of glyphosate (Roundup WeatherMAX®, 48.8% a.i., Monsanto, St. Louis, MO) was applied with a compressed-air backpack sprayer at each herbicide application at a rate of 11,206 L ha⁻¹ (1198 gallons acre⁻¹). Total species richness, native species richness, and Floristic Assessment Quotient for Wetlands (FAQWet index; Ervin et al. 2006) were calculated from percent cover data to assess the effect of glyphosate applications on the composition of resulting cover. FAQWet index is a calculated score used to determine overall floristic quality of wetland habitats when coefficients of conservatism are not available for the habitat or location (Ervin et al. 2006).

The experiment was designed as a randomized split-plot design. Experimental plots were blocked by location (NW

or SE side of the Sweetwater Branch tributary). Six plots were randomly located on each side of the Sweetwater Branch tributary. The main plot factor, season of initiation of herbicide application (fall or spring; Figure 1), was randomly assigned to 6 main plots at each of two locations. The subplot factor, number of herbicide applications (0, 1, 2, or 3 applications), was randomly assigned to the four subplots within a main plot. Each of the eight treatment combinations was replicated six times. Data were analyzed using the PROC MIXED procedure in SAS with date as a repeated measure and checked for normality by examining histogram and normality plots of the conditional residuals (SAS v.9.2, SAS Institute Inc., Cary, NC, USA). Pairwise comparisons were completed using Tukey's honestly significant difference (HSD) test at a significance level of $p < 0.05$. Initial Mexican petunia cover, native species richness and total species richness were included as covariates in the model to account for differences in initial plot cover. Mean Mexican petunia cover was also calculated at the completion of the experiment for treatment with a single glyphosate application.

A single glyphosate application significantly reduced Mexican petunia cover, whether initiated in fall or spring ($t = 4.11$, $df = 40$, $p = 0.0002$) (Figure 2A, B). Our results

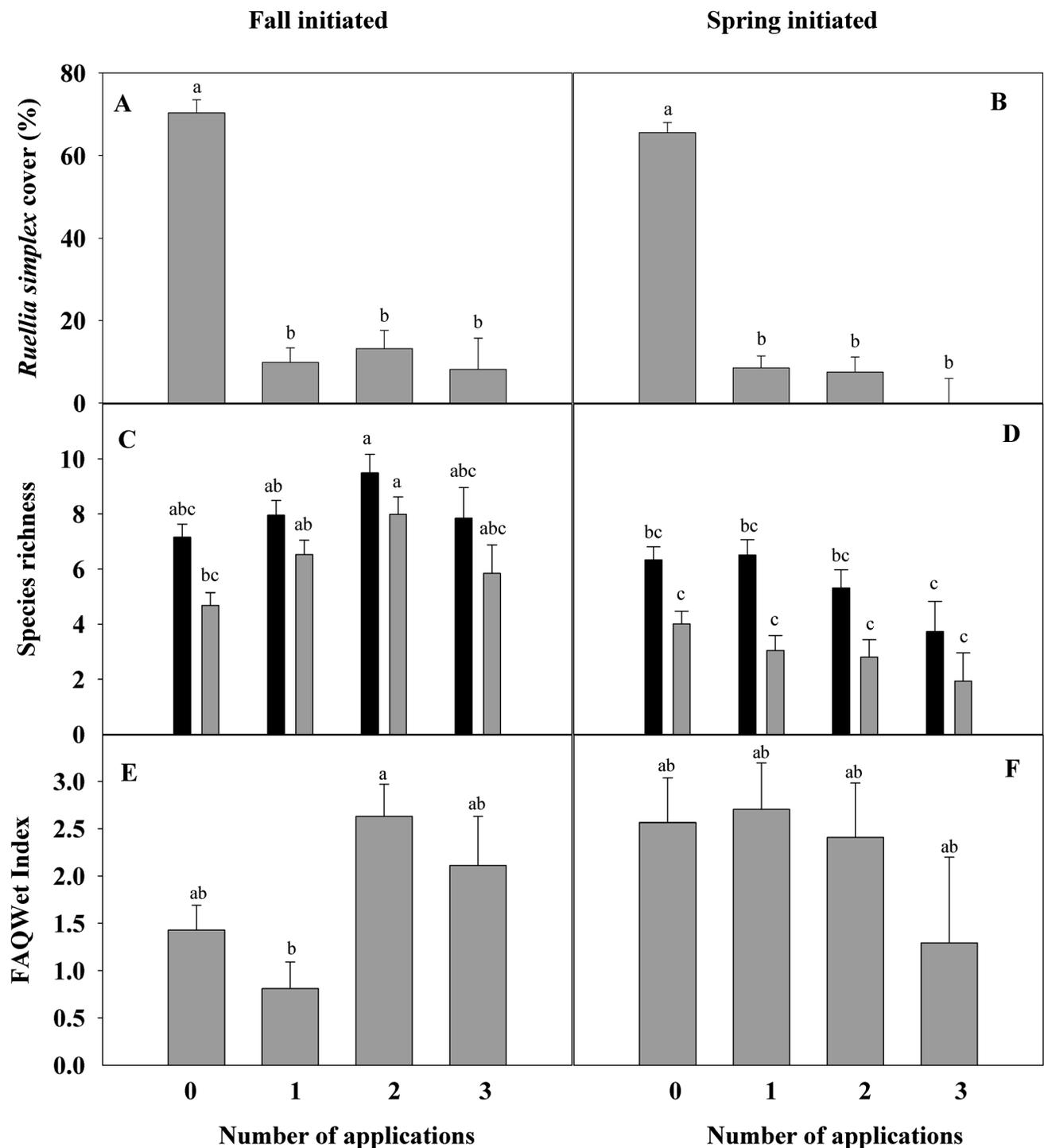


Figure 2. (A–B) Mean percent cover of Mexican petunia (*Ruellia simplex*) after glyphosate application (0, 1, 2, or 3 applications over a 1-year period) initiated in either fall (A) or spring (B). Means were generated by the repeated measures statement in SAS and represent percent cover of all monitoring periods over the course of one year. Lowercase letters indicate mean separations between treatment interactions ($p < 0.05$). (C–D) Total and native species richness of resulting vegetation after glyphosate application (0, 1, 2, or 3 applications over a 1-year period) initiated in either fall (C) or spring (D). Means were generated by the repeated measures statement in SAS and represent species richness of all monitoring periods over the course of one year. Lowercase letters represent mean separation of either total (black bars) or native (gray bars) species richness between treatment interactions ($p < 0.05$). (E–F) Quality (FAQWet index) of above-ground vegetation cover after glyphosate applications (0, 1, 2, or 3 over a 1-year period) beginning in either fall (E) or spring (F). Means were generated by the repeated measures statement in SAS and represent FAQWet index of all monitoring periods over the course of one year. Lowercase letters represent mean separation of FAQWet index between treatment interactions ($p < 0.05$).

Table 1. Test statistics from the ANOVA model for native species richness for the interaction between season of initiation of glyphosate spray application and number of spray applications. The mean, degrees of freedom (df), t-statistic (t), and p-value for each interaction is shown.

Native Species Richness					
Season	Number of applications	Mean	df	t	p-value
Fall	0	7.1524	40	14.91	$p < 0.0001$
Fall	1	7.9450	40	14.65	$p < 0.0001$
Fall	2	9.4887	40	14.18	$p < 0.0001$
Fall	3	7.8532	40	7.15	$p < 0.0001$
Spring	0	6.3344	40	13.20	$p < 0.0001$
Spring	1	6.5122	40	11.88	$p < 0.0001$
Spring	2	5.3084	40	7.97	$p < 0.0001$
Spring	3	3.7285	40	3.40	$p = 0.0015$

support previous research which noted that plots with pre-treatment Mexican petunia cover of 95% were reduced to between 0 and 10% Mexican petunia cover 51 days after herbicide application (see Wiese et al. 2013). Similarly, we found mean Mexican petunia cover was reduced from 72% (fall plots) or 89% (spring plots) to 10% (fall plots) or 11% (spring plots) after 90 days (Figure 2A, B). Percent cover of Mexican petunia (41%) during periods of seasonally low coverage (January) was still greater in untreated control plots than in plots treated with herbicide by 5 to 50%. Previous research also indicated that two applications were needed to achieve adequate control during the 183-day study (see Wiese et al. 2013). However, we observed similar Mexican petunia control in plots treated with either one ($t = 4.11$, $df = 40$, $p = 0.0002$) or two ($t = 3.58$, $df = 40$, $p = 0.0009$) glyphosate applications (Figure 2A, B).

Total and native species richness cover was influenced by the interaction between season of glyphosate application initiation and number of applications (Figure 2C, D). Dominant native species (average cover $\geq 10\%$) were primarily herbaceous perennials with lesser percentages of herbaceous annuals and woody perennials. Plots that received two fall-initiated applications (Figure 1) had greater total species richness than all plots that received spring-initiated (Figure 1) applications (one spring-initiated application: $t = 3.42$, $df = 40$, $p = 0.0290$; two spring-initiated applications: $t = 4.42$, $df = 40$, $p < 0.0017$; three spring-initiated applications: $t = 4.49$, $df = 40$, $p < 0.0014$) (Figure 2C, D). Similarly, plots that received one or two fall-initiated applications had greater native species richness than all plots that received spring-initiated applications, as well as control plots that received no glyphosate (Table 1; Figure 2C, D). Above-ground vegetation quality was slightly higher when two fall-initiated applications were applied (FAQWet = 2.63) compared to a single fall-initiated application (FAQWet = 0.81; $t = -4.18$, $df = 40$, $p = 0.0035$) (Figure 2E, F), but there were no other differences.

We suspect that the higher total and native species richness observed post-treatment in the fall compared to spring, was due to seasonal germination patterns. Ruffner and Barnes (2010) noted similar variation in resulting grassland vegetation after herbicide treatment of invasive tall fescue (*Festuca arundinacea*) and attributed differences to season of sampling (spring, summer, or fall). However, seasonally limited percent cover of some native species may have been a result of herbicide application at a phenologically susceptible period of growth. Ruffner and Barnes (2010) similarly reported that application timing of the herbicide clethodim affected cover of native grasses after application, such that early (April 4) applications had fewer non-target effects than late (April 20) applications. Likewise, Young et al. (2002) found that timing of herbicide application affected the balance between reduction of invasive perennial pepperweed (*Lepidium latifolium*) and increase in a competitive native, tall wheatgrass (*Elytrigia elongata*).

As timing of herbicide application had no effect on Mexican petunia cover and a minimal effect on overall quality of resulting vegetation, land managers can likely be flexible in timing their glyphosate application without loss of effectiveness of Mexican petunia control. Fall initiated applications may better promote native species recovery, but more research is needed to confirm this. Restoration of the plant community, as well as long-term control of Mexican petunia, appear unlikely to result from natural recolonization, and may require active revegetation following herbicide applications. Mean Mexican petunia cover nine months after one glyphosate application was 57% (fall) and 26% (spring). Therefore, both underlying causes of reinvasion, as well as barriers to native species recolonization, should be addressed in future research.

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Evaluation of Five Herbicide Treatments to Control Yellow Toadflax (*Linaria vulgaris*)

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Yellow toadflax (*Linaria vulgaris*) is an invasive weed of rangelands, forests, and croplands in North America and a common problem for land managers (Wilson et al. 2005). Endemic to central Europe and the Mediterranean

region, yellow toadflax was intentionally introduced to North America during the 1800s (Wilson et al. 2005) as an ornamental, a fabric dye, and for ethnobotanical use. The species escaped cultivation and has become an opportunistic invader throughout the continental United States, Canada, and parts of Mexico (Jacobs and Sing 2006). Yellow toadflax has been legally designated as a noxious weed in much of western North America (USDA 2013); it thrives in disturbed, open habitats and can easily dominate native or desirable plant communities (Wilson et al. 2005).

Yellow toadflax poses a serious threat to ecosystems across North America due to its high reproductive potential and ability to invade existing plant communities (Ward et al. 2008). The species reproduces sexually via seed and vegetatively via adventitious shoots from spreading roots (Ward et al. 2008). Dense patches of yellow toadflax also can lead to problems in agricultural fields (Wilson et al. 2005); studies in Montana have shown that infestations of 12 yellow toadflax stems/m² can reduce yields of canola by 20%; similar reductions have been reported in wheat fields with infestations of 74 stems/m² (Jacobs and Sing 2006). Presence of yellow toadflax can cause a shift in flower timing, as well as decreased abundance, species richness, species evenness, and flowering duration of common, flowering native species (Wilke and Irwin 2010). These effects have the potential to alter vital pollination networks within ecosystems (Wilke and Irwin 2010). Additionally, yellow toadflax is not preferred forage for livestock or wildlife (Sing and Peterson 2011); therefore, invasion in pasture and rangelands can lead to diminished forage abundance and habitat quality.

Currently, effective methods for yellow toadflax control are limited. Mowing and prescribed burning have no effect on killing roots or decreasing seed bank (Lym and Travnicek 2010, Erskine-Ogden and Renz 2011) and only select herbicidal treatments have had limited success (Lym 2002). Biological controls, such as the toadflax stem-mining weevil (*Mecinus janthinus*) and the toadflax flower-feeding beetle (*Brachypterolus pulicarius*), decrease overall fitness (e.g., delay seed production) but do not result in eradication (Jacobs and Sing 2006). Thus, land managers need accurate information on treatment methods to implement effective control measures for yellow toadflax.

We evaluated five herbicide treatments, along with differences in the timing of application (flowering and post-flowering), to determine their ability to decrease yellow toadflax stem densities. Four herbicide treatments (E-2, Perspective®, Telar®, Tordon®) were applied alone and one treatment was a mixture of two herbicides (Tordon®/Overdrive®; Table 1). The study was conducted on two heavily invaded U.S. Fish and Wildlife Service Waterfowl Production Areas (WPAs) located in the Kulm Wetland Management District in North Dakota, USA. These WPAs provide critical nesting habitat for waterfowl and other

Table 1. List of herbicides evaluated for controlling yellow toadflax (*Linaria vulgaris*) in Kulm Wetland Management District in North Dakota, USA. Year and location of treatments are listed, along with application rates used for the study and corresponding costs.

Herbicide	Active Ingredients	Manufacturer	WPA treated	Year treated	Application Rate	Cost ¹
E-2	2,4-D Fluroxypyr Dicamba	Nufarm Americas, Burr Ridge, IL, USA	Todd	2011	1.75 L/ha (1.5 pt/ac)	\$44.00/ha (\$17.81/ac)
Perspective®	Aminocyclopyrachlor Chlorsulfuron	DuPont, Wilmington, DE, USA	Todd	2011	146 mL/ha (2 oz/ac)	\$27.15/ha (\$10.99/ac)
Telar XP®	Chlorsulfuron	DuPont, Wilmington, DE, USA	Malm	2012	146 mL/ha (2 oz/ac)	\$123.42/ha (\$49.97/ac)
Tordon 22K®	Picloram	Dow Agroscience, Indianapolis, IN, USA	Malm	2012	4.67 L/ha (2 qt/ac)	\$98.67/ha (\$39.95/ac)
Tordon®/ Overdrive®	Picloram Fluroxypyr Dicamba	BASF Specialty Products, Research Triangle Park, NC, USA	Malm	2012	2.33 L/ha (T) (1 qt/ac) + 438 mL/ha(O) (6 oz/ac)	\$97.36/ha (\$39.42/ac)

¹ Costs are from the 2012 North Dakota Herbicide Compendium

species of ground-nesting birds. Our study sites (Todd WPA and Malm WPA) were situated adjacent to one another and therefore, shared geographical and biotic characteristics. Climate of the region is characterized by cold winters (January average -12°C) and hot summers (July average 21°C). Total annual precipitation averages 48 cm (National Weather Service 2013). Todd WPA (64.7 ha) and Malm WPA (129.5 ha) were considered non-native grassland and seeded with dense nesting cover (intermediate wheatgrass [*Thinopyrum intermedium*], tall wheatgrass [*T. ponticum*], and alfalfa [*Medicago sativa*]) in 1979 and 1983, respectively. However, at the time of our treatments, vegetation at both sites was dominated by smooth brome (*Bromus inermis*) and Kentucky bluegrass (*Poa pratensis*) with established populations of Canada thistle (*Cirsium arvense*), absinth wormwood (*Artemisia absinthium*), leafy spurge (*Euphorbia esula*), and yellow toadflax.

We divided each WPA into equally sized study plots (Todd WPA: 1.62 ha plots, Malm WPA: 2.63 ha plots) and used a randomized block design to assign treatments. All herbicide treatments (Table 1) were applied using a vehicle equipped with a boom sprayer. To examine timing of herbicide application, treatments were applied during flowering (early August) on half the treated plots and post-flowering (late September) on the remainder of the treated plots. We created random points ($n = 20$) within each plot to assess vegetation pre- and post-treatment. At each random point, yellow toadflax stem density was recorded within a 1 m^2 frame (Daubenmire 1959). Due to the one year difference between treatment dates, we report two years of post-treatment data for Todd WPA and one year of post-treatment data for Malm WPA. We tested for differences in yellow toadflax stem density and timing of treatments (flowering and post-flowering) using

analysis of variance (ANOVA). We used Tukey-Kramer pairwise comparisons to test for differences among treatments and conducted statistical tests using SAS version 9.3 (SAS Institute, Cary, NC), with an experiment-wide error rate of 0.05.

Pre-treatment yellow toadflax stem density was moderately higher in the plots eventually assigned to the Perspective® treatment (mean stem density = 10.8 stems/ m^2) and the Tordon® treatment (mean stem density = 16.7 stems/ m^2) compared to the other plots (Todd WPA: $F = 2.83$, $df = 2$, 184, $p = 0.06$; Malm WPA: $F = 2.49$, $df = 3$, 395, $p = 0.06$) at Todd WPA (mean stem density = 7.5 stems/ m^2) and Malm WPA (mean stem density = 11.8 stems/ m^2), respectively. Herbicide treatments were not effective when applied post-flowering (Todd WPA: one year post-treatment $F = 1.24$, $df = 2$, 295, $p = 0.20$, two years post-treatment $F = 1.07$, $df = 2$, 296, $p = 0.35$; Malm WPA: $F = 1.12$, $df = 3$, 249, $p = 0.34$). Because post-flowering treatments were not effective, all subsequent results are from treatments applied during the flowering stage.

One year after herbicide application on Todd WPA, we observed a 27% reduction in yellow toadflax stem density in plots treated with E-2, and an 83% reduction in stem density in plots treated with Perspective®, compared to the control plots ($F = 14.78$, $df = 2$, 292, $p < 0.001$; Figure 1A). Two years after treatment, without reapplication of herbicides, we observed a 42% reduction in yellow toadflax stem density in plots treated with E-2, and an 84% reduction in stem density in plots treated with Perspective®, compared to the control plots ($F = 21.14$, $df = 2$, 291, $p < 0.001$; Figure 1A). One year after we treated Malm WPA, we observed a 65% reduction in yellow toadflax stem density in plots treated with Telar®, an 86% reduction in stem density in plots treated with Tordon®/Overdrive®,

and a 25% reduction in stem density in plots treated with Tordon[®], compared to the control plots ($F = 9.29$, $df = 3$, 240 , $p < 0.001$; Figure 1B).

The five herbicide treatments resulted in differing levels of yellow toadflax control (Figure 1). The two herbicide treatments with the highest reductions in yellow toadflax stem densities were Tordon[®]/Overdrive[®] and Perspective[®]. These herbicides were nearly identical in their effectiveness one year post application. Telar[®] also reduced yellow toadflax, but not as effectively as Tordon[®]/Overdrive[®] or Perspective[®]. E-2 and Tordon[®] resulted in the least yellow toadflax control. This result is interesting because whereas E-2 was minimally effective in its first year, it had higher reductions in yellow toadflax stem densities two years after application. Yet, after two years, its success at reducing yellow toadflax was below that of Tordon[®]/Overdrive[®] and Perspective[®]. Tordon[®] by itself was the least effective herbicide but the Tordon[®]/Overdrive[®] mixture produced the highest reduction in yellow toadflax. This result is consistent with several other studies; mixing Overdrive[®] with Tordon[®] increased yellow toadflax control from 70% to 98% two years post treatment (DiTomaso et al. 2013). Tordon[®] and Overdrive[®] have previously been examined as a means to control invasive weeds in North Dakota (Lym 2005) and, in agreement with our results, mixing Overdrive[®] with Tordon[®] doubled the effectiveness of control, compared to Tordon[®] applied alone (Lym 2005).

Timing of herbicide application proved to be a critical factor in the success of herbicide treatments. None of the herbicide treatments applied during the post-flowering stage produced reductions in yellow toadflax. However, all of the herbicide treatments applied during the flowering stage reduced yellow toadflax stem density. Most herbicide treatments for yellow toadflax are recommended for application during the flowering stage and/or while plants are growing rapidly (Erskine-Ogden and Renz 2011, DiTomaso et al. 2013) and our results support herbicide application during this period.

Herbicide costs and legal restrictions are major determinants when deciding what herbicide treatment to use. Of the five treatments examined, two significantly reduced yellow toadflax stem density and may be viable treatment options for control. Telar[®] produced moderate reductions in yellow toadflax, but was the most expensive option (Table 1). Tordon[®] applied alone was the second most expensive option and was the least effective herbicide applied. However, Tordon[®] combined with Overdrive[®] produced the greatest reduction in yellow toadflax stem density. This option, though, was almost as expensive as Tordon[®] applied alone (due to the higher application rate). E-2 was the fourth most expensive, and did not control yellow toadflax. In this experiment, Perspective[®] was highly effective, and was much less expensive than the Tordon[®]/Overdrive[®] mixture. E-2, Telar[®], Tordon[®], and Overdrive[®] were approved for use on rangeland and Conservation

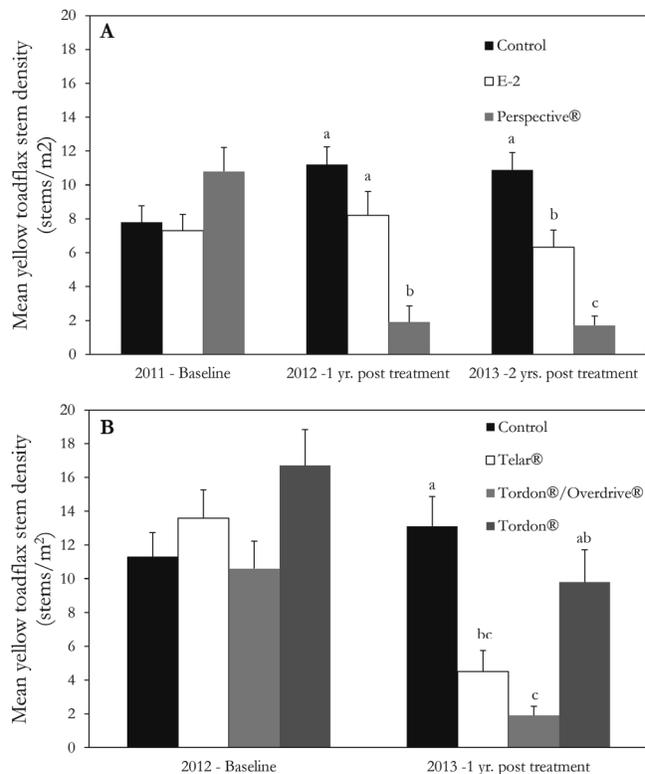


Figure 1. Reductions of yellow toadflax (*Linaria vulgaris*) by five different herbicide treatments. All results are from herbicides applied during the flowering stage of yellow toadflax. Perspective[®] had the greatest reductions both years on Todd WPA (A), while Tordon[®]/Overdrive[®] had the greatest reductions on Malm WPA (B). Bars marked with different lowercase letters are significantly different from each other with the same year ($p < 0.05$).

Reserve Program lands, and Perspective[®] was approved for use on natural areas (wildlife management areas, wildlife openings, and wildlife habitats).

Although yellow toadflax presents a challenge to land managers, our results demonstrate that effective and cost-efficient chemical methods of control do exist. Our results indicate that herbicides should be applied during the flowering stage of yellow toadflax for maximum control. All herbicide treatments produced reductions in yellow toadflax when applied while flowering. Tordon[®]/Overdrive[®] produced the largest decrease in yellow toadflax stem density, was labeled for rangeland use, and cost was \$97.36/ha. Perspective[®], however, was nearly as effective, was labeled for natural areas, and cost was \$27.15/ha. Perspective[®] and Tordon[®]/Overdrive[®] both produced significant reductions in yellow toadflax stem density; however, due to the cost difference, land managers should consider using Perspective[®] during the flowering stage because it was the most cost-effective chemical control of yellow toadflax. All herbicides should be used in accordance with their labeled use and under appropriate conditions with regard to site-specific characteristics.

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Using Fish Behavior to Assess Habitat Quality of a Restored Oyster Reef

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Human activities in the coastal realm have led to drastic declines in population sizes of marine organisms (Lotze et al. 2006). Of particular concern have been declines in foundation species (e.g., oysters, seagrasses, and coral) that form the structural basis for ecosystems and are associated with numerous ecosystem processes that we highly value. Oyster reefs, for instance, provide habitat for other fishery species, enhance water quality, sequester carbon, and stabilize shorelines; economic valuations suggest the sum of these ecosystem services may total \$99,000/hectare/year (Grabowski et al. 2012). Despite this significant value, oyster population declines have been substantial, e.g., oyster biomass declined by 88% in U.S. coastal waters between the early 1900s and early 2000s (Ermgassen et al. 2012).

Restoration projects are increasingly being implemented to stem or reverse these declines. Although many oyster reef restorations are designed to increase production for commercial purposes, another important goal is to reestablish the multiple ecosystem services that are supported by living oyster reefs. In the context of coastal fishery declines (Lotze et al. 2006), of particular interest is the efficacy of oyster reef restoration projects in augmenting populations of economically and ecologically important fish and invertebrate species. New fish production is likely supported by restored oyster reefs (Peterson et al. 2003), although precise estimates of increased fishery yields are difficult to quantify. In fact, simply assessing the extent that nekton utilize restored oyster reefs can be difficult, due to the relatively small size of many restored reefs within a heterogeneous seascape, temporal dynamics in species abundance, and the challenges associated with sampling transient fauna in structurally complex habitat (Grabowski et al. 2005).

Here we provide an alternative perspective to assess the success of oyster reef restorations in the context of fishery ecology, namely using the behavior of individual fish to

reflect habitat quality. We utilize acoustic telemetry to quantify fish movement patterns in a patch habitat before and after an oyster restoration project, with the *a priori* assumption that increased residence time of individuals within a habitat patch is related to the quality of that habitat (Lindell 2008).

The study was conducted in the Loxahatchee River (26°57' N, -80°06' W), a 27-kilometer-long coastal river that flows into the Atlantic Ocean near Jupiter, Florida, USA. Oyster reefs have been significantly degraded in the system, largely as a result of anthropogenic alteration of salinity. The section of river where salinities presently favor oyster growth is substrate limited, with a benthos composed largely of sand and silt that lacks settlement habitats for larval oysters. In 2008, the Loxahatchee River District and The Nature Conservancy initiated a community-based oyster reef restoration program. The goal of this project was to create new oyster reef habitat in the substrate-limited section of river by deploying bags of dried oyster shell and Reef Balls™ (Reef Ball Foundation, Inc., Athens, GA, USA) under privately owned residential docks (Figure 1).

This fish movement study was conducted at one of the restoration docks, located in the northwest fork of the river approximately 6km from the ocean (26°58.038'N, -80°07.192'W). In this section of the river, the entire shoreline is developed, and now consists of bulkheads and docks. As such, the main structurally complex habitats available (before the restoration project) were dock pilings and scattered rip-rap under docks; no significant oyster reefs or other natural habitat types (mangroves) were present along this shoreline. In October 2008, the restoration reef was constructed by adding a continuous layer of bagged oyster shell (150 black duronet diamond mesh bags, 33 cm wide, 19 mm mesh; DelStar Technologies, Inc., Middletown, DE, USA) under the dock (Figure 1). To provide additional surface area for oyster recruitment, 15 Reef Balls were placed under the deepest portion of the dock. A total of ~37m² of substrate were covered with the oyster bags and Reef Balls™. Following restoration, recruitment of sessile (e.g., oyster spat) and motile benthic organisms (e.g., small crustaceans) to newly created reefs occurs rapidly (Jud and Layman, in review), with new reefs supporting prey for mesopredators that utilize these habitats (Yeager and Layman 2011).

Our study organism was the mangrove snapper (*Lutjanus griseus*), a commercially and ecologically important fish species common throughout Florida and the Caribbean (Yeager and Layman 2011). Juvenile and sub-adult mangrove snapper tolerate a wide range of salinities, and inhabit a diverse suite of habitats, including seagrass beds, oyster reefs, mangroves, and human-made habitats. We acoustically tagged mangrove snapper at the study site before the restoration reef was constructed (October 2007) and again following reef construction (January 2009). Individual fish were captured by hook and line, and Vemco



Figure 1. Mesh bags filled with oyster shell (shown in Figure) and concrete Reef Balls™ were used for the restoration project. The dock in the background was the site used for fish movement monitoring in this study

V7 sonic transmitters (22.5 × 7 mm, Vemco, Inc., Bedford, Nova Scotia, Canada) were surgically implanted into their abdominal cavity (Hammerschlag-Peyer and Layman 2010). The tags were set to “ping”, i.e., emit a coded signal unique to that tag, at 3–4 minute intervals. Six fish were caught and tagged on each sampling date and their movements monitored for 28 days. A datalogging hydrophone (Vemco VR2, Vemco Inc., Bedford, Nova Scotia, Canada) was used to detect pings from tagged fish, but the standard omnidirectional capacity was modified such that detection area was limited to a ~90° cone directed under the focal dock (directed from deeper water toward the shoreline). A polyurethane foam backing was placed on the side of the receiver opposite to the dock, such that fish passing next to or behind the receiver would not be detected. That is, the directional receiver design allowed us to specifically document when an individual mangrove snapper was under the study dock (referred to as the “patch habitat” herein), but did not record fish movements in other adjacent habitats. Extensive *in situ* range testing confirmed effectiveness of this design.

Distinct residency and movement patterns were observed in the patch habitat before and after the restoration project. Before restoration, individual mangrove snapper were detected on average 15±9 (mean ± SD) times/day by the directional receiver over a 4 week period following tagging (Figure 2); fish tagged following the restoration were detected an average of 245±65 times/day, a 16-fold difference in the number of detections (t-test, $t = 9.0$, $p < 0.001$; SigmaStat Version 4.0). Inspection of more fine-scale detections suggests different movement patterns before and after restoration (Figure 3). Before the restoration, snapper moved through the habitat patch periodically, perhaps moving in small schools, as nearly synchronous detections from multiple tagged fish were

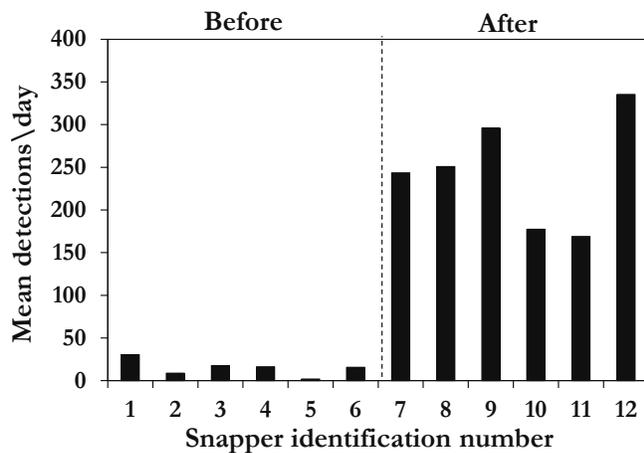


Figure 2. Mean detections/day for each individual fish over the 28 day observation period. Numbers 1–6 represent six individual mangrove snapper before restoration, and numbers 7–12 are the six tagged individuals after the restoration.

frequent. Likely due to the patchy, relatively rare, distribution of food resources along shorelines without oyster reef (or other quality habitat), snapper did not remain in a single habitat patch for extended periods of time (typically < 10 minutes). Following the restoration event, individual snapper spent significantly more time in the dock area, within or around the bags of shell and Reef Balls™; snorkeling surveys qualitatively support this observation, with dozens of mangrove snapper typically under the dock at any one time. If habitat use patterns can be assumed to reflect aspects of habitat quality (Lindell 2008), our data provides an individual-level, behavioral perspective to assess the success of oyster reef restorations.

These data are consistent with basic tenets of the Ideal Free Distribution, IFD (Fretwell and Lucas 1970), a theory which states that the number of individual animals that will aggregate in various habitat patches is proportional to the amount of resources available in each. IFD rests on the assumption that organisms can assess the relative availability of resources across potential foraging areas, and that they can freely move among those areas selecting preferred habitat. If resources are higher in one patch than another, IFD predicts that consumers should choose the habitat with greater availability of resources. In the case of oyster restoration, restored reefs in this estuarine system have been shown to support extremely high densities of small invertebrates and fishes (up to 350 g/m²) that would not be found in bare sand/silt substrate (Jud and Layman, in review); these are the primary food resources for mangrove snapper in these habitats (Yeager and Layman 2011). An alternative explanation is that the restored reef provides refugia from larger predators (e.g., juvenile bull sharks [*Carcharhinus leucas*], common snook [*Centropomus undecimalis*], crevalle jack [*Caranx hippos*]) that are common in the area. Whatever the underlying mechanistic

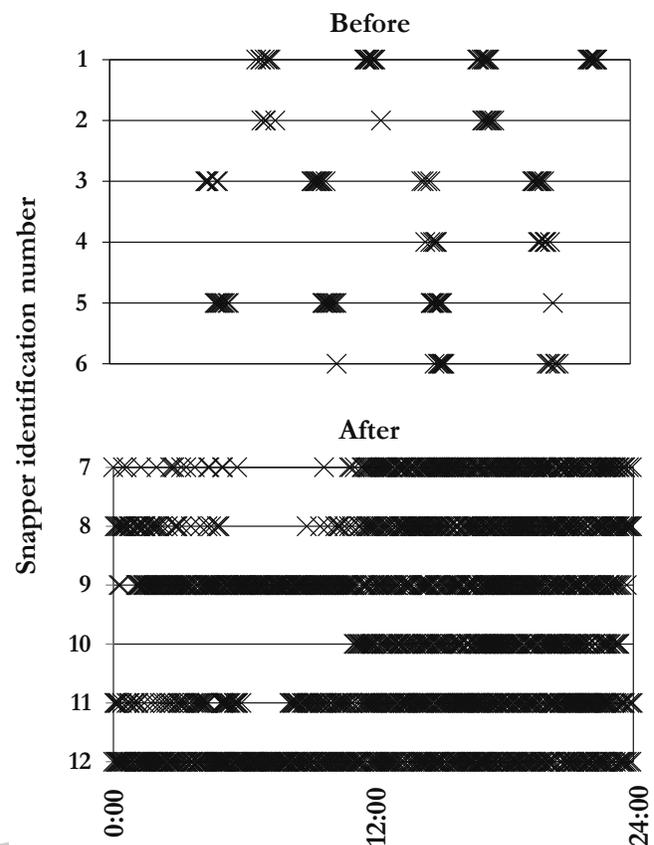


Figure 3. A representative 24-hour period displaying when individual fish (horizontal rows) were detected under the focal dock before (top) and after (bottom) restoration. Numbers 1–6 represent six individual mangrove snapper that were tagged before the restoration, and numbers 7–12 are six different individuals that were tagged after the restoration. Each X indicates a single acoustic detection by the directional receiver, with the dense concentration of Xs for some individuals representing continuous presence under the dock.

driver, our data show snapper use of the restored patch habitat increased substantially, providing insight into an enhanced ecosystem function (i.e., habitat provisioning) following the restoration.

More specialized individual-level behaviors, i.e., higher variation among individuals within a population, often emerge when competition for resources is high and resources are patchily distributed (Araújo et al. 2011). For example, mangrove snapper in another shallow, resource-limited, coastal system were shown to have highly individualized movement patterns related to divergent resource use patterns (Hammerschlag-Peyer and Layman 2010). Likewise in the Loxahatchee system, patchy resource distribution (before the reef restoration) was likely associated with differential utilization of patch habitats. Following restoration, variation in movement patterns among individuals was reduced, with all individuals utilizing the patch habitat extensively throughout the monitoring period. This pattern suggests that competition for resources was alleviated

to the point that seeking alternative habitat patches (and associated resources) was not necessary to maintain fitness.

Typical assessments of nekton utilization of restored habitats are based on snapshot estimates of species composition and abundance. Assessment of behavioral responses to habitat restoration remains a far less commonly applied perspective. With small sample sizes, the extent to which we can generalize our results is rather limited. Yet our data provide a simple example of the fundamental shifts in behavior that may be manifest following restoration of oyster habitat, even at relatively small spatial scales. This provides a case study where it may be useful to compare the behavior of individuals to reflect restoration success, rather than trying to quantify numerous other aspects of community composition or ecosystem function (Lindell 2008). Such an approach may be especially useful in coastal habitats where temporally dynamic and structurally complex habitats render it difficult to completely document population, community, and ecosystem-level change following restoration.

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