Restoration Notes

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Evaluating Effects of Historic Agriculture and Current Restoration Activity on Succession and Plant Diversity in the New Jersey Pine Barrens

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The New Jersey Pine Barrens consists of 400,000 hectares of upland, aquatic, and wetland habitats with sandy, acidic soils (Good and Good 1984, Forman 1998). The area played an important role in New Jersey's agricultural history, with cranberry (*Vaccinium macrocarpon*) as one of the state's staple crops (Procopio 2010, Wen 2010). In the late-1800s and early-1900s, Atlantic white-cedar (*Chamaecyparis thyoides*) and red maple (*Acer rubrum*) swamps were often converted into cranberry bogs. Initially, there was little soil alteration because agricultural areas were within floodplains and hydric soils were present. In the mid- to late-1900s, the process of "modernizing" cranberry bogs allowed expansion onto higher elevations including upland and wetland areas on the edges of floodplains. Sites were intensively managed with deep ditches, drains, perforated pipes, and diesel pumps that obtained water from aquifers during droughts. Sand overlaying the organic horizon of wetland soil was leveled to ensure even drainage and compacted to allow heavy equipment access. A new layer of sand was added every few years to bury cuttings and stimulate crop production (Eck 1990, Wen 2010).

There is substantial interest of restoring swamps on retired agricultural sites in the Pine Barrens (Mylecrane et al. 2004). In 2003, the New Jersey Conservation Foundation (NJCF) acquired the title to the Franklin Parker Preserve (FPP): 3,800 ha of forests, swamps, and bogs formerly owned by the AR DeMarco Cranberry Company (39.772444, -74.529383). This area is within the village of Chatsworth (Burlington County, NJ, US). Aerial photographs from 1930 show patches of cranberry bogs within a matrix of swamps prior to modernization (Reiser 2014). Starting in the 1960s, over 405 ha of this area was converted into modernized cranberry bogs, with the last harvest in 2001.

In 2004, with funds from the United States Department of Agriculture Natural Resource Conservation Service's (USDA NRCS) Wetlands Reserve Program, NJCF began restoring 445 ha of the FPP that were altered by agriculture, encompassing land that was modernized as well as bogs that were less intensively managed. In modernized bogs, restoration included plugging unnatural bypass ditches and drains and returning the original stream flow onto historic floodplains. The NJCF overturned compacted soil to create mounds and expose hydric soil buried beneath the sand. In older, peat-based cranberry bogs that had not been modernized, hydric soil was still at the surface and natural succession was occurring. Therefore, recontouring terrain was unnecessary. The restoration goal was to "deconstruct" agricultural modifications to hydrology so that tributaries could passively return to, and flow across agriculturally modified floodplains throughout the site. As of 2012, most of the 445-ha restored area had reestablished a natural hydrologic regime. In 2013, the wetland experienced its first full growing season under a naturally fluctuating stream corridor hydrology since it was intensively managed.

This study presents pilot research investigating effects of restoration efforts on species recruitment to a 32-ha subsection where hydrologic function was restored by early 2010. Our objectives were twofold: 1) to monitor preliminary effects of restoration activities on species recruitment, and 2) to document vegetative structure that exists within bogs that were modernized versus those that were not modernized. Plant diversity, coverage, tree recruitment, and hydric soil conditions were used as indicators of recovery.

This study focused on three habitats within the FPP: 1) a modernized bog undergoing restoration, 2) a peat-based bog in early succession, and 3) a modernized bog without restoration (control). Nine years after cranberry agriculture ceased, in 2010, we established four 100-m^2 (10 m × 10 m) plots within each environment, except on the modernized bog control site where its size limited us

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Table 1. Summary of descriptive statistics (mean ± SE) comparing three cranberry bog habitats, Franklin Parker Preserve, Chatsworth, NJ. For each variable, significant differences among bog habitats (α = 0.1) are noted with different letter superscripts.

Variable	Modern bog	Peat-based bog	Modern, restored bog
Mean species richness/1m² (# species)	7.20±0.41[a]	7.00±0.49[a]	8.05±1.07[a]
Herbaceous biomass (g)	13.01±2.96[a]	65.25±10.87[a]	120.32±28.8[b]
Woody biomass (g)	159.29±18.48[a]	109.38±21.90[a]	563.08±223.30[b]
Hydric soil depth (cm)	0.66± 0.11[a]	7.09±0.88[b]	2.75±0.74[a]
Standing water (%)	0.00±0.00[a]	0.05±0.05[a]	13.50±6.03[b]
Understory cover (%)	73.87±8.28[a]	100±0.00[b]	73.85±5.74[a]
Overstory cover (%)	0.00±0.00[a]	0.00±0.00[a]	38.80±5.14[b]
Red maple cover (%)	0.27±0.05[a]	0.28±0.13[a]	37.28±6.56[b]
Red maple height (cm)	1.40±0.53[a]	2.55±1.06[a]	3.20±0.77[b]

to three plots. There was no planting activity that occurred within the study area; therefore species encountered were introduced to the site through natural recruitment.

We randomly placed five 1-m² quadrats within each plot to sample plant percent cover and density. Due to multiple canopy layers, total percent cover can exceed 100% (Peet et al. 1998, Ahn and Dee 2011). Species were identified according to the USDA NRCS Plants Database (www.plants.usda.gov) and assigned a percent cover. Biomass for 1 m² was calculated by clipping above-ground understory vegetation in quadrats, separating it into separate bags for woody and herbaceous plants, and oven drying at 100°C at Kean University before weighing the material. To sample soil, the upper leaf layer was removed and 20 2 cm × 20 cm soil cores were randomly collected in each of the plots We used criteria established by the USDA NRCS (2010: 16–17) to measure hydric soil and root depth in each quadrat, focusing on "sandy soils". We observed a layer of mucky peat or peat starting near the soil surface, and underlain by sandy soil. Most plant remains in the "peat" were sufficiently intact to enable identification of plant remains, whereas "mucky peat" is at an intermediate stage of decomposition between peat and highly decomposed muck. Depths of hydric soil were measured with a metric ruler up to 15 cm.

Comparison between wetland types (modernized/restored, peat-based, and modernized/not restored) was conducted using ANOVA (α = 0.1) with the following dependent variables: root and hydric soil depth, herbaceous and woody biomass, percent cover of standing water, total overstory, total understory, species percent cover, and height and density of each woody species within each quadrat. A Tukey post-hoc test was conducted (PASW Statistics v.18.0, Quarry Bay, Hong Kong). We calculated species richness and diversity within the 100 m² plots to assess vegetative development. Species richness (S) was recorded as the number of vascular plant species. The Shannon-Weiner Diversity Index (H′) was calculated using MS Excel where H′ is a function of relative plant percent cover (p_i = species percent cover/total percent cover) (Ahn and Dee 2011).

We found notable differences in the structure and species diversity of the three sites (Table 1). There was evidence of a difference in woody plant biomass (ANOVA; F = 3.162, p = 0.097), and the percent cover of standing water (F = 4.228, p = 0.056), understory percent cover (F = 6.871, p = 0.018), overstory percent cover (F = 48.372, p = 0.000), and depth of hydric soil (F = 19.460, p = 0.001) were significantly different between habitats. Species diversity was greater in the restored site (H′=1.73) than in the modernized control bog (H′=1.59) and the less intensively managed bog (H′=1.54).

Effects of modernization were evident when compared to cranberry bogs with intact peat. The restored and control bogs had significantly less hydric soil depth (Tukey HSD; p = 0.001) and understory cover than the peat-based bogs. Interestingly, peat-based bogs had lower richness for the cumulative species found (S =10) than both modernized bog treatments (control: S = 20; restored: S = 21). When richness was calculated by quadrat, however, there was no significant difference between habitats.

In measuring restoration success based on the Tukey HSD post-hoc test, we found evidence that modernized bogs undergoing restoration had greater herbaceous biomass (Tukey HSD; p = 0.014, MD 107.31 ± 28.86 SE) woody biomass (Tukey HSD; p = 0.100, MD 453.70 ± 194.80 SE), and overstory coverage (Tukey HSD; p = 0.000, MD 38.80 ± 4.81 SE) than the control. In particular, red maple recruitment indicated accelerated succession in restored bogs when compared to the control. While there was little evidence of a difference for the number of red maple seedlings establishment (Tukey HSD; p = 0.167, MD 2.67 ± 1.31 SE) between restored and control modernized bogs, they grew taller (Tukey HSD p = 0.003, MD 51.60 ± 10.26 SE) and provided more canopy cover (Tukey HSD; p = 0.001, MD 37.01 ± 6.13 SE) in restored bogs.

Preliminary results indicate that modernization damaged the pre-existing peat layer, an essential foundation for succession into swamps. The absence of hydric soil at or near the surface in the modernized bog compared to the peat-based bog is limiting the herbaceous plant cover.

Without restoration, modernized bogs will likely have a delayed successional progression.

Noticeable differences in biomass occurred between the sites, with restored sites having significantly higher herbaceous and woody biomass. Although species richness in $1m^2$ quadrats was not markedly increased by restoration, H′ indices for species diversity increased, perhaps by reducing dominance of the species suited to the uniformity of the former agricultural habitat. The heterogeneity created by the restoration activities has allowed for biomass to accumulate more rapidly than in the control or the peat-based bog, and although not significantly different, there is a measurably thicker hydric layer in the soil of the restored sites. These changes in vegetative communities and soil formation are essential for the ecosystem to regain functions of wildlife and plant habitat value, restoring the water holding capacity of the system to resist drought and assist in flood and erosion control.

Compared with upland farms characterized by a homogenous landscape, the diverse habitats within cranberry farms create heterogeneity with various levels of anthropogenic disturbance (Wen 2010). This characteristic is particularly true for agricultural wetlands of the FPP. Our finding of lower species richness in the peat-based bog compared to modernized bogs is likely due to the fact that while cranberries were being grown, some native plant species coexisted within the intact wetland. After agriculture ceased and cranberry desiccated in winter, co-existing native wetland species gained dominance. In modernized bogs, however, cranberries were maintained in monoculture. After agriculture ceased, cranberry plants died and there were higher levels of species introduction and colonization.

We found that the FPP plant community is closely linked to the degree of agricultural manipulation or restoration of the vegetation and soil. Through reintroducing a dynamic hydrology, mounded topography, and hydric soil, restoration activities instituted by the NJCF are accelerating succession of the modernized bogs into swamps. In addition to the restoration activities reported here, the NJCF is actively reintroducing Atlantic white-cedar (*Chamaecyparis thyoides*) and native forbs to many of the modernized cranberry bogs after the microtopography is restored, although planting activity did not occur yet in the sites selected for this study. The planting of red maple is not necessary, since that species is being naturally recruited on the site by neighboring intact swamps, as demonstrated by this research. The development of vegetative communities resulting from the NJCF activities is essential in order for the ecosystem to regain functions of wildlife and plant habitat value and restoring the water holding capacity of the system.

References

Ahn, C. and S. Dee. 2011. Early development of plant community in a created mitigation wetland as affected by introduced hydrologic design elements. *Ecological Engineering* 37: 1324–1333.

Eck, P.1990. *The American Cranberry*. New Brunswick, NJ: Rutgers University Press.

Forman, R.T.T 1998. *Pine Barrens: Ecosystem and Landscape.* New Brunswick, NJ: Rutgers University Press.

Good, R. E. and N.F. Good. 1984. The Pinelands National Reserve: An Ecosystem Approach to Management. *BioScience* 34(3): 169–173

Mylecraine, K.A., G.L. Zimmermann, R.R Williams, and J.E. Kuser. 2004. Atlantic white-cedar wetland restoration on a former agricultural site in the New Jersey Pinelands. *Ecological Restoration* 22(2): 92–98.

Peet, R.K., T.R. Wentworth, and P.S. White. 1998. A flexible, multipurpose method for recording vegetation composition and structure. *Castanea* 63(3): 262–274.

Procopio, N.A. 2010. Hydrologic and morphologic of variability in streams with different cranberry agriculture histories. *Journal of the American Water Resources Association* 46: 527–540.

Reiser, J. 2014. New Jersey State Atlas: 1930s Aerial Photography. Trenton, NJ: New Jersey Department of Environmental Protection and Office of Information Technology. (http://njstateatlas.com/1930/Geo: 39.772444, -74.529383 USNG: 18S WK 4030 0260).

United States Department of Agriculture, Natural Resources Conservation Service. 2010. *Field Indicators of Hydric Soils in the United States,* Version 7.0. L.M. Vasilas, G.W. Hurt, and C.V. Noble (eds). Washington, DC: USDA, NRCS, in cooperation with the National Technical Committee for Hydric Soils.

Wen, A. 2010. Ecological functions and consequences of cranberry (*Vaccinium macrocarpon*) agriculture in the pinelands of New Jersey. PhD dissertation, Rutgers University.

Soil Amendment Increases Tree Seedling Growth but Reduces Seedling Survival at a Retired Gravel Mine

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Restoring forest vegetation within denuded settings requires identifying the abiotic factors that limit plant establishment (Bradshaw 1997, Whisenant 1999). In heavily disturbed sites such as quarries and gravel pits that have been denuded of native soils, degraded soil processes may limit planted tree seedling survival or growth

Table 1. Mean soil carbon and C to N ratio differed between amended and unamended plots at both 0–15 cm and 15–30 cm depths. Soil moisture and nitrogen were higher at shallow depths in the amended plots, while bulk density did not differ between amended and unamended plots. Letters in 2002 and 2005 data indicate significant differences (ANOVA; $F_{1,36} > 4.12$, $p < 0.05$ for all tests) between amendment treatments within a given soil depth and year. Due to limited sampling, no statistics were performed on pre-restoration soil monitoring.

		Pre-Restoration			2002		2005	
Soil Parameter	Depth	Gravel Mine	Volunteer Reference	Mature Reference	Amended	Unamended	Amended	Unamended
Carbon	0–15cm	0.15	2.37	2.20	40.41^{a}	4.3^{b}	25.82^{a}	3.21^{b}
	15–30cm	—	0.62	1.18	11.86^{a}	5.22^{b}	18.33^{a}	5.67^{b}
Nitrogen	0–15cm	0	0.08	0.08	0.49^{a}	0.26^{b}	0.81^{a}	0.22^{b}
	15–30cm	—	0.02	0.04	0.39^{ns}	0.36^{ns}	0.51^{a}	0.76^{b}
C to N Ratio	0–15cm	48	30	28	87.68^{a}	16.27^{b}	37.82^{a}	7.09^{b}
	15–30cm	—	29	24	30.86^{a}	13.79^{b}	40.42^{a}	11.12^{b}
Bulk Density	0–15cm	—	—	—	0.65^{ns}	1.28^{ns}	0.67^{ns}	1.17^{ns}
	15–30cm	—	—	—	1.66^{ns}	2.41^{ns}	1.71^{ns}	2.25^{ns}
% Soil Moisture	0–15cm	—	—	—	17.39^{a}	2.36^{b}	19.17^{a}	6.17^{b}
	15–30cm	—	—	—	—	—	8.72^{ns}	6.17^{ns}

(Williamson et al. 2011). To improve vegetation establishment on denuded sites, soil amendments are commonly used to manipulate soil fertility (Biederman and Whisenant 2011a, Hough-Snee et al. 2011a), introduce organic matter , soil microorganisms, or propagules (Sinnett et al. 2008, Hough-Snee et al. 2012), or to create heterogeneous microhabitats (Biederman and Whisenant 2011b, Hough-Snee et al. 2011b) that improve plant survival. In forest restoration, amendments that facilitate soil development can lead to increased plant survival and growth (Ortiz et al. 2011). Specifically, amendments that increase soil carbon and nitrogen have been shown to improve planted tree seedling growth (Wilson-Kokes et al. 2013). Soil amendments for denuded sites are typically designed to increase planted tree growth so that seedlings may outcompete early seral vegetation and survive to maturity (Bradshaw 1997). In this study we examined how soil amendments change soil properties at a highly disturbed site and how these amendments impact the growth and survival of three early successional tree species. We tested two sets of hypotheses:

1. Soil amendment will increase soil carbon and nitrogen, C to N ratio, soil moisture, and decrease soil bulk density.
2. The amendment-driven increase in soil fertility will increase the survival and growth of planted black cottonwood (*Populus balsamifera*), red alder (*Alnus rubra*) and Douglas fir (*Pseudotsuga menziesii*) seedlings relative to unamended seedlings.

The restoration site was a 1.7-ha retired gravel mine located on an alluvial terrace near Goodell Creek, a tributary to the Skagit River (Washington State, USA, elevation: 162 m). Gravel operations ceased in 1990 when the mine and surrounding area were incorporated into North Cascades National Park. The surrounding matrix consists of mature Douglas fir- and western hemlock- (*Tsuga heterophylla*) dominated conifer forest in uplands and black cottonwood-, red alder-, and western red cedar- (*Thuja plicata*) dominated riparian forest. Soil parameters were sampled within both forest types and the gravel mine to identify soil conditions prior to restoration (Table 1).

The primary restoration objective was to use amendments to establish soil properties that facilitate early-successional, coniferous-deciduous forest stand development. Temperate forests of Washington's western Cascades are generally low in available nitrogen, so amendments were designed to raise soil organic matter content and moisture retention capacity without increasing N mineralization that would favor competitive ruderal weed establishment. Prior to amendment application, the entire site was graded to a ~7% grade and stockpiled sandy loam aggregate was evenly spread to a depth of 15 cm. Soil amendment consisted of a secondarily digested paper pulp sludge stabilized with fly ash (Smukler 2003). The amendment had high initial nitrogen content, so the carbon to nitrogen (C to N) ratio was increased prior to application by adding partially decomposed alder sawdust (36% C, 0.29% N, C to N ratio = 125; Smukler 2003). In the summer of 2001, the amendment was spread across two 0.45 ha blocks and tilled into the sandy loam topsoil (15 cm deep) over the mine's semi-compacted subsoil. Two 0.45-ha blocks remained unamended and were not tilled. Thirty-six 20 m² circular experimental plots were created, nine per block. We measured soil percent carbon, percent nitrogen, moisture, and bulk density, and calculated the C to N ratio at plot centers in early summer of 2002 and 2005 (methods discussed in Smukler 2003). Soils were sampled at 0–15 cm and 15–30 cm depths to identify amendment effects across depths. As part of an additional experiment on promoting revegetation through seed rain recruitment (Pond 2005), three mulch treatments were randomly and evenly applied to plots within amended and unamended blocks,

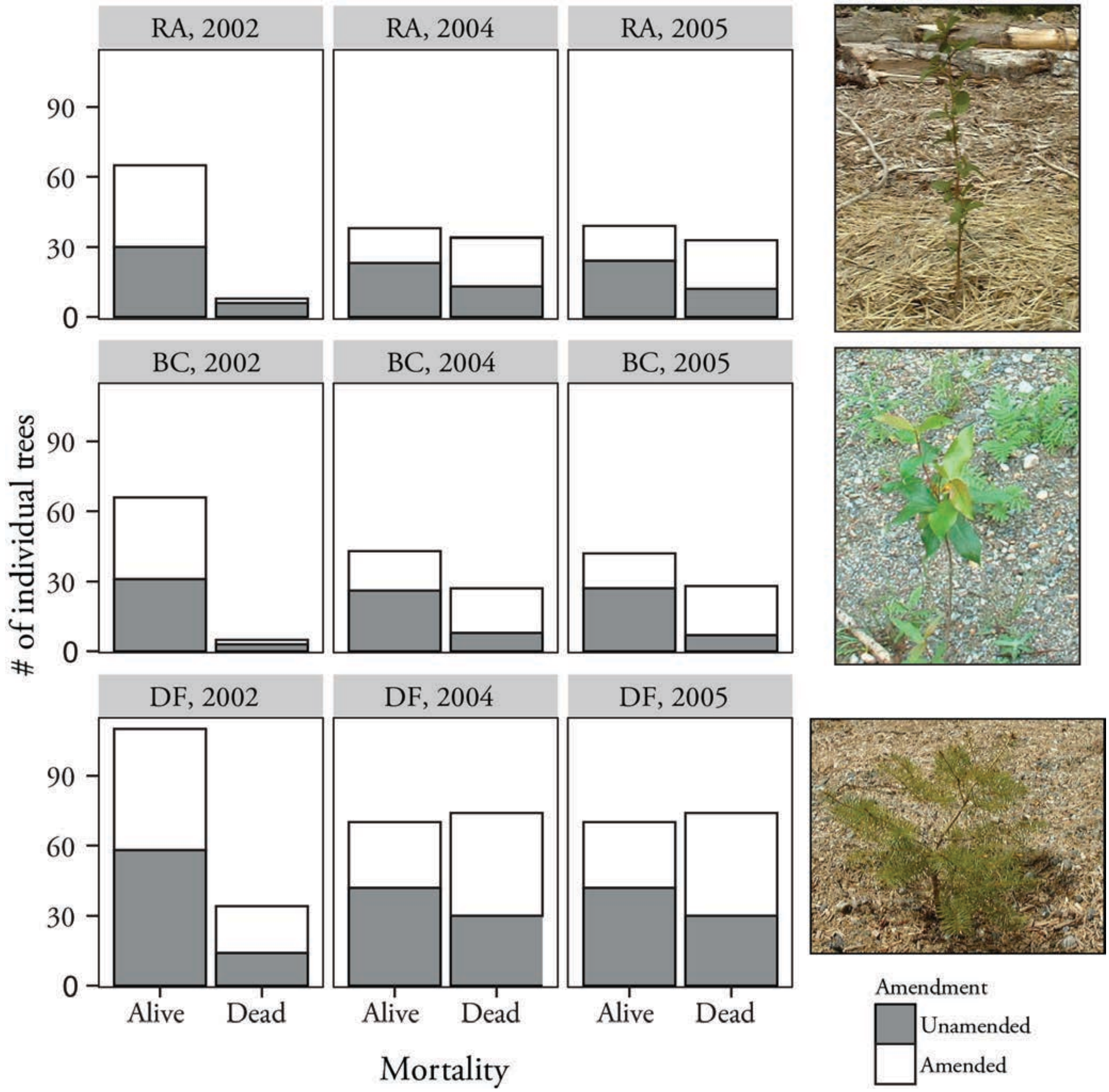

Figure 1. Frequency plots of mortality by year, species, and soil amendment show that amended individuals were more likely to die than unamended individuals. All species had similar mortality rates, but there were twice as many Douglas fir (DF) seedlings as red alder (RA) and black cottonwood (BC).

woodchips (chipped red alder bark), straw (weed-free), and a control. Mulches were applied to plot surfaces (2–3 cm deep) within each block to influence seed capture, retention, and germination, but not rooting zone soil properties.

Bare root seedlings (1–0) of red alder, black cottonwood, and Douglas fir were planted in November 2001. Four Douglas fir, two cottonwood, and two alder were planted within each circular plot at evenly spaced, random compass headings. Tree height was measured at planting and ranged from 40 to 60 cm for all species. Height was measured from the soil surface to the end of the main stem terminal bud. Tree survival and height were surveyed in 2002, 2004, and 2005. Growth was calculated as the difference in heights between each measurement year and 2001 height at planting. If tissue dieback occurred then measured heights less than those at the time of planting were recorded as negative measurements.

Initially we tested for the effects of amendment and mulch treatments on soil properties using two-way ANOVA. There were no significant mulch effects on soil properties at either year or depth, so this effect was removed and one-way ANOVA was performed on soil properties

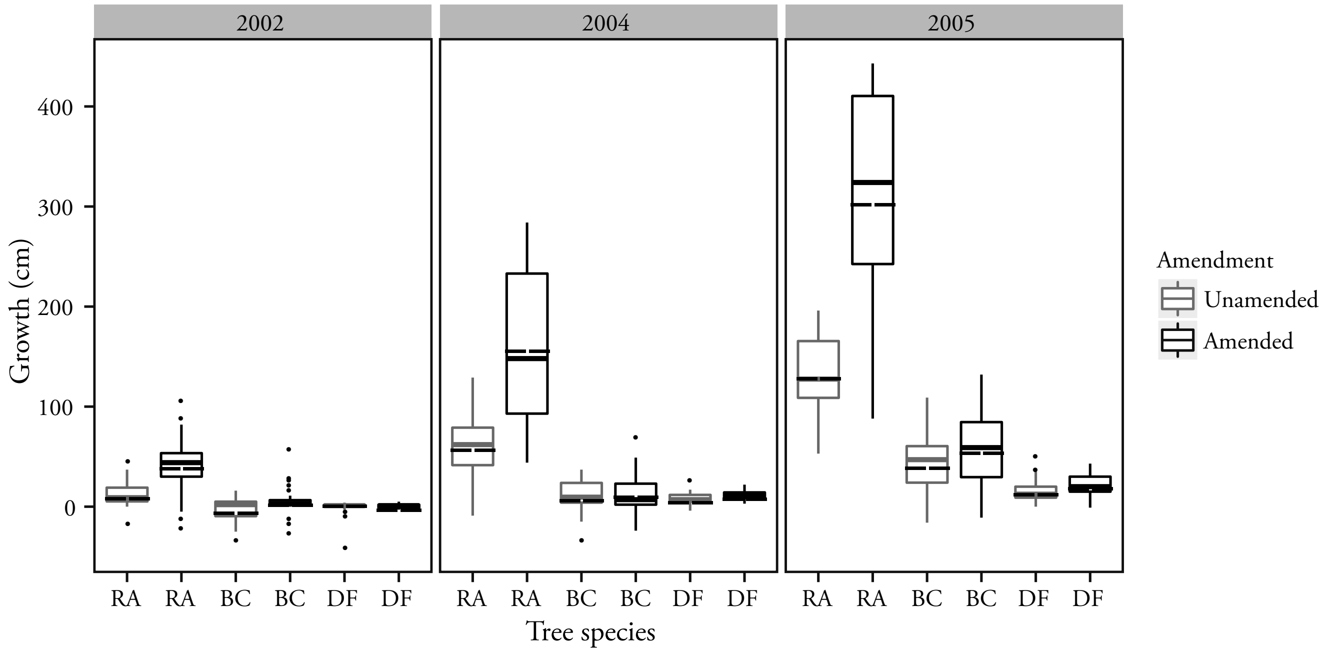

Figure 2. Box and whisker plots of each species' growth between 2001 and 2002, 2004, and 2005. Amended red alder (RA) grew the most of all species and treatments across all growth intervals. Amended black cottonwood (BC) and Douglas fir (DF) outgrew their unamended counterparts. Black bars (—) indicate the mean change in height for a given species within a given treatment and year.

for amendment. We used binomial regression to identify differences in tree survival between soil amendment treatments and ANOVA to test for the effects of amendment on surviving seedling growth. Each species' growth was analyzed independently between 2001–02, 2001–04, and 2001–2005. Because we were interested in the effect of amendment on soils and tree survival and growth, when mulch effects were not significant we removed this term from models, testing only for amendment effects.

Amendment elevated soil C, N, and C: N ratio above the levels found in volunteer and mature reference forests and the pre-restoration gravel mine. Soil carbon and C:N ratio were higher in amended plots than unamended plots at both shallow and deep depths in both 2002 and 2005 (Table 1). Nitrogen was higher in amended plots at shallow depths in both years. Bulk density was lower in amended plots than in unamended plots although not statistically significant in both years. Soil moisture was higher in amended plots than in unamended plots in both years. There were no significant mulch effects on soil properties in either year or depth. Soil treatment means and statistical results are presented in Table 1.

More unamended cottonwood seedlings died than amended seedlings in 2002. In 2004 and 2005 more unamended cottonwood seedlings survived than amended seedlings (Binomial regression; 2004: $Z = 2.76$, $p = 0.006$; 2005: $Z = 3.23$, $p = 0.001$; Figure 1). Amended Douglas fir seedlings experienced higher mortality than unamended seedlings over the study duration (Binomial regression; 2002: $Z = 2.04$, $p = 0.042$; 2004 and 2005: $Z = 2.56$, $p = 0.011$; Figure 1). Red alder mortality did not differ between amended and unamended seedlings in 2002, but was higher in amended seedlings in 2004 and 2005 (Binomial regression; 2002: $Z = 1.63$, $p = 0.104$; 2004: $Z = 1.70$, $p = 0.089$; 2005: $Z = 2.00$, $p = 0.046$; Figure 1). Mulch treatment was not significant in any survival models. Cottonwood seedling growth was higher in amended plots across the 2001–02 interval (ANOVA; $F = 7.39$, $p = 0.009$). Amendment did not significantly improve cottonwood growth between the 2001–04 and 2001–05 growth intervals (Figure 2). Douglas fir growth was higher in amended plots than in unamended plots between 2001–04 (ANOVA; $F = 6.51$, $p = 0.013$) and 2001–05 ($F = 5.25$, $p = 0.025$), but did not differ between 2001–02. Red alder growth was significantly higher in amended plots across all intervals (ANOVA; 2001–02: $F = 32.08$; $p < 0.001$; 2001–04: $F = 28.84$, $p < 0.001$; 2001–2005: $F = 53.67$, $p < 0.001$; Figure 2). Mulch was also a significant term in the model for alder growth between 2001–02 (ANOVA; $F = 3.484$, $p < 0.037$). Although straw mulch increased alder growth within amended plots in 2002, no other mulch treatment affected growth or survival, therefore we do not present mulch-amendment pairwise comparisons.

Amendment increased soil nutrition and moisture, resulting in increased growth of surviving tree seedlings. For all three species, amended plots had higher seedling growth than unamended plots. While these results confirmed our initial hypotheses, survival results were less intuitive. Soil amendment actually increased plant mortality over the four-year study duration. This could be due to excessively low soil bulk density coupled with increased nutrition that promoted rapid aboveground growth but only shallow rooting. The average bulk density of unamended soils was twice that of amended soils in both years across the soil column. Low amendment bulk density likely resulted from the use of sawdust to balance carbon and nitrogen ratios. At low bulk densities, macropores (airspaces that don't allow capillary water movement in the soil column) increase, reducing the soil's capacity to retain water near the soil surface during dry conditions. Although soil amendments retained moisture at shallow depths under wet conditions, it is possible that the loose texture and low density encouraged rapid drying in young seedlings' rooting zones during summer drought. Soil drying would be most disadvantageous to cottonwood and alder that have high transpiration and generally need consistent moisture to survive.

Another explanation for higher mortality in amended plots is that soil amendment increased growth and competition among planted seedlings. Competitive exclusion may have most adversely affected relatively slow-growing, shade intolerant Douglas fir, especially in amended plots in 2004 and 2005. Red alder, a nitrogen-fixing tree, grew rapidly and 100 times taller than Douglas fir in amended plots. This suggests that species' life history strategies will influence how planted vegetation communities respond to soil enrichment in stressful environments. When the primary filter that shapes community assembly is physical (i.e. soils), then using amendments to improve physical properties will allow vegetation to establish. However, once vegetation has established, biotic filters (i.e. competition) will shape planted vegetation survival and growth. At our site, faster-growing alder and cottonwood may have shaded Douglas fir, increasing mortality and slowing growth. When using amendments that give species with competitive strategies a growth advantage, successional management may be necessary to increase slow-growing species survival. Thinning adjacent woody vegetation, weeding, or using plastic mulches to prevent competing vegetation from establishing are all viable options to ensure that planted seedlings survive and grow to shape future community assembly.

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References

Biederman, L.A. and S.G. Whisenant. 2011a. Amendment placement directs soil carbon and nitrogen cycling in severely disturbed soils. *Restoration Ecology* 19:360–370.

Biederman, L.A. and S.G. Whisenant. 2011b. Using mounds to create microtopography alters plant community development early in restoration. *Restoration Ecology* 19:53–61.

Bradshaw, A. 1997. Restoration of mined lands using natural processes. *Ecological Engineering* 8:255–269.

Hough-Snee, N., J.D. Bakker and K. Ewing. 2011a. Long-term effects of initial site treatment on fescue in a novel prairie ecosystem (Washington). *Ecological Restoration* 29:14–17.

Hough-Snee, N., A.L. Long, L. Jeroue and K. Ewing. 2011b. Mounding alters environmental filters that drive plant community development in a novel grassland. *Ecological Engineering* 37:1932–1936.

Hough-Snee, N., Long, A.L. and R.L. Pond, 2012. Passive soil manipulation influences the successional trajectories of forest communities at a denuded former campsite. *Ecological Restoration* 30:9–12.

Ortiz, O., G. Ojeda, J.M. Espelta and J.M. Alcañiz. 2011. Improving substrate fertility to enhance growth and reproductive ability of a *Pinus halepensis* afforestation in a restored limestone quarry. *New Forests* 43:365–381.

Pond, R.L., 2005. Low elevation riparian forest restoration on a former gravel mine, North Cascades National Park USA: Native plant germination, growth and survival in response to soil amendment and mulches. Masters Thesis. University of Washington, Seattle, WA.

Sinnett, D., J. Poole and T.R. Hutchings. 2008. A comparison of cultivation techniques for successful tree establishment on compacted soil. *Forestry* 81:663–679.

Smukler, S.K., 2003. Optimizing the use of residuals as soil amendments for ecological restoration. Masters Thesis. University of Washington, Seattle, WA.

Whisenant, S.G. 1999. *Repairing Damaged Wildlands a Process-oriented,Landscape-scale Approach*. Cambridge, UK: Cambridge University Press.

Williamson, J.C., E.C. Rowe, P.W. Hill, M.A. Nason, D.L. Jones and J.R. Healey. 2011. Alleviation of both water and nutrient limitations is necessary to accelerate ecological restoration of waste rock tips. *Restoration Ecology* 19:194–204.

Wilson-Kokes, L., C. DeLong, C. Thomas, P. Emerson, K. O'Dell and J. Skousen. 2013. Hardwood tree growth on amended mine soils in West Virginia. *Journal of Environment Quality* 42:1363–1371.

Decisions . . . Decisions . . . How to Source Plant Material for Native Plant Restoration Projects

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How far away should the genetic origin of plant material be from the restoration site? This is a primary question for restoration practitioners, but there are no simple answers. Issues involving cost, availability, adaptability, population genetics, and community resilience complicate practitioners' abilities to determine precise locations and distances from the restoration site. The majority of formalized guidelines for sourcing plant material are determined on a project-by-project basis. This important decision can affect the longterm sustainability of the restored community and potentially negatively impact levels of adaptive variation in local populations of native species.

"Local is best" is a commonly held tenet among restoration professionals. Unfortunately, "local" means different things to different people and, depending on the long term goal of your project, local may not be best. Local or local ecotype is an extension of the concept of plant ecotypes that has been used to describe and identify populations that originated and are adapted to local conditions (e.g., climate, soils, pathogens, etc.). Using local seed sources is an effort to identify populations that have experienced similar evolutionary selective forces (abiotic and biotic interactions), which should result in higher fitness of plants introduced at restoration sites. However, our ability to predict the spatial and temporal scale of variation in adaptive traits differs among populations and species (Linhart and Grant 1996). Significant differences in fitness may occur between individuals a meter apart and another set of differences between individuals located 100s of kilometers away (Waser and Price 1985, Galloway and Fenster 2000). Defining and identifying local populations is difficult, typically occurs with imperfect knowledge of underlying genetic differences, and results in an inconsistent set of assumptions among practitioners, such as policies that state anywhere from 40 kilometers to over 320 kilometers from the site of concern (Saari and Glisson 2012).

The reason for intense scrutiny of this issue is the possibility of short-term or longterm failure of introduced plants, potential inbreeding resulting from low genetic diversity and/or increased invasive characteristics within restored populations, and introduction of novel genes into adjacent local populations (Hufford and Mazer 2003). Failure to thrive can result from maladaptation of introduced plants to local conditions that can cause poor germination, establishment, or disruption of plant-animal interactions, such as pollination (Keller et al. 2000). Local native populations experience outbreeding depression (reduced survival, seed set, and seed viability) as a result of the introduction of alien genes (Hufford et al. 2012).

Potential negative impacts to local populations and longterm success of the restored community have motivated the selection of plant material for native plant restoration. However, longterm success also depends on the restored community's ability to adapt to changing environments and adaptation is more likely to occur in genetically diverse populations (Fant et al. 2008). In regions that need restoration, nearby remnant populations may have reduced genetic diversity because of their small size and isolation. If local sources are constrained to these remnant populations, the amount of genetic variation may not be sufficient for population persistence over time.

In order to explore this complex issue, the U.S. Army Corps of Engineers' restoration ecologist, Brook Herman, Chicago District, organized and hosted the *Plant Material Sources for Ecological Restoration Conference*, focused on the restoration of native plant communities using plant material sourced from outside the project site. More than a dozen restoration practitioners, researchers, and nursery professionals gathered in Chicago, IL on July 25, 2012 to present their study results, real world examples, and expert opinions. Approximately 50 people attended. Participants ranged from local forest preserve ecologists to endangered species specialists to biologists working on mine reclamation projects. The workshop consisted of 14 presentations with intermittent open discussion among the presenters and participants. Discussion focused on the pros and cons of options for sourcing seed based on the conservation goals, type of project, and budgetary constraints.

As presenters conveyed their experiences, several key questions arose: Can the type of pollen/seed dispersal mechanisms of different functional groups (e.g., grasses vs forbs) inform how to source species? Should project type (e.g., urban park vs. high quality remnant) dictate the distance to a source population? Should the project site conditions (e.g., soils, microclimate, etc.) constrain which sources (and by extension nursery microclimate) are considered for sourcing material? Given predictions for climate change, should sources come from further South (or North if located in southern hemisphere) of the project site? Is the cost of sourcing multiple populations, increasing genetic diversity and resilience, justifiable for long term sustainability in light of future climate change? What is the relative importance of inbreeding or outbreeding depression for local and introduced restored plant populations?

There are three distinct groups of professionals that play a role in the design and construction of native plant restoration projects. They represent restoration practitioners, academic researchers, and nursery professionals. The first presenters (Stephen Packard, Cathy Pollack, Gregory Houseal, Shawn Sinn and Chip O'Leary) represented restoration practitioners. An overarching theme from this group was that each project site presents unique challenges that should be met with a flexible set of restoration goals and objectives. Clear precise restoration goals will inform decisions about where to locate seed sources. For example, the goals of the North Branch restoration (Stephen Packard) supported source collection protocols within a 24-kilometer radius of the restoration site. Shawn Sinn pointed out that many contract specifications call for a radius of 240–400 kilometers from the restoration site. While distance from the restoration site was considered, matching the characteristics (e.g., soils) of donor with recipient sites also played a role. Chip O'Leary described the history of the Kankakee Sands restoration project in terms of first delineating general seed source areas (80-km radius) to fine-tuning areas based on geomorphology and soil type. And finally, when sources for specific species are not available within predefined areas, working with other agencies, private landowners, and commercial suppliers should be considered. Consideration should also be given to the time it will take to cultivate these relationships and efficiently propagate and prepare enough plant material for current (e.g., phase in species as they become available) and future restoration projects. Consistent demand for desired species and specific sources will incentivize nursery professionals to supply them in quantities needed.

The second group (Jeremie Fant, Abigail Derby Lewis, Stuart Wagenius, Danny Gustafson, and Kristina Hufford) presented their academic research, ranging from evaluating gene flow between populations, genetic diversity of rare species, effects of climate change on species distributions, cases of outbreeding depression, and failure of non-local ecotypes. Jeremie Fant provided an overview of genetic issues that should be considered in locating sources. For instance, practitioners should consider amount of genetic diversity of donor populations, how to identify distinct local populations and why they are distinct, and potential for adaptation of introduced material to local microclimatic variables. Genetic diversity and resilience of a plant community to climate change should be carefully considered during project plan formulation. Abigail Derby Lewis advocated a flexible range of distances for source material based on climate change model projections and long term functional success of restored plant communities. Also, species at the southern extent of their range (in the northern hemisphere) within the area may not be suitable targets for restoration. Local adaptation can be difficult to detect with molecular genetic tests, and there may be differences in adaption to microclimates of populations within species that are assumed to be similar. Danny Gustafson's research on dominant grasses of the tallgrass prairie, southeastern coastal salt marsh, and sweetgrass (*Muhlenbergia filipes*) plant communities showed genetic and ecological differences between local and non-local plant material. These differences in morphology, plant-insect interactions, and genetic signature persisted more than 20 years after planting adjacent to a remnant prairie, despite evidence of gene flow between local and non-local genotypes. Stuart Wagenius's research indicates that prairie remnants adjacent to restoration sites may be in danger of becoming less fit because of inadvertent introduction of non-local genes into their gene pool and disruption of pollination of rare species from closely related non-local species. Kristina Hufford ended with a review of possible strategies for sourcing material based on each species' pollen and seed dispersal mechanisms. For example, genetic similarity between two sites is assumed to be greater in wind-dispersed compared to animal-dispersed populations. Life history traits can be used to predict the distance over which species are likely to adapt to local environmental conditions.

The final group of presenters represented professional nurseries specializing in propagating native plants (Bob Allison, Kelsay Shaw, Steve Haines, and Corrine Daniels). Growers expressed a commitment to work within any distance or microclimate constraints and with any list of species given enough time to locate and propagate quantities required for a contract. Frequently, contracts require uncommon or rare species with insufficient time to locate and propagate them, requiring substitutions with less desirable species. Kelsey Shaw emphasized the economic realities of growers needing a consistent demand for uncommon species or specific sources to result in a readily available supply from growers. Also, Bob Allison said that many growers do not keep track of specific microclimate characteristics of source populations that are used for propagation, although this can be retroactively added and tracked if there is a demand for this type of information.

From the workshop it became clear that decisions involving where and how to source native plant material should consider species characteristics (e.g., wind-pollinated), the material (seeds vs. live plugs) being sourced, longterm goals of the restoration project, budget, site condition and location of restoration site in relation to local native populations. Participants concluded the workshop with a critical discussion of the issues. We attempted to summarize general guidelines that could be applied during the planning process for projects that involve restoration of native plant communities. Foremost in the decision-making process is consideration of the goals and objectives of the restoration project. If the goals are to enhance a degraded plant community adjacent to a high quality remnant, the best course of action would to be source material from the remnant. In contrast, if the goal is to reestablish a functional wetland within a residential neighborhood, sources may be sought from farther away. Once goals and objectives of a project are clear, protocols for plant material selection can be defined to include the type of microclimate conditions within the site (e.g., loamy vs. sandy soils) and type of species (e.g., wind/animal pollination). If allowable, projects should use a larger region to locate sources, and then plan on sourcing the same species from two or more populations. This increases genetic variation; however, it also includes a risk of outbreeding depression. Be aware that multiple years may be required for many uncommon or rare species and budget accordingly. Finally, we should plan with future environmental conditions in mind. If results from climate models are not readily available for your region, at least consider an increase in average temperature. Although changing climate was one of many issues to consider, there are other local characteristics not projected to change, such as day length. This will fine tune decisions regarding specific species and how far north and south appropriate sources are from your restoration site.

How and where to source plant material for ecological restoration continues to be important for current and future projects. Even though this conference improved our understanding of the complex issues involved with native plant restoration, many questions still remain. For example, how important is finding multiple sources of locally or regionally adapted plant material for the persistence of the restored plant community under changing environmental conditions (Broadhurst et al. 2008, Pickup et al. 2012)? Continued research efforts, using greenhouse studies, common garden plots, and monitoring restoration sites, should be encouraged (Golay et al. 2013). Data gathered from restoration sites are particularly scarce (Gibson et al. 2013). The use of climate change distribution modeling (Potter and Hargrove 2012, Breed et al. 2013) should also be explored. These tools can help to delineate material transfer zones that will encourage more efficient and effective seed sourcing policies and coordination with private industry.

References

Breed, M.F., M.G. Stead, K.M. Ottewell, M.G. Gardner and A.J. Lowe. 2013. Which provenance and where? Seed sourcing strategies for revegetation in a changing environment. *Conservation Genetics* 14:1–10.

Broadhurst, L.M., A. Lowe, D.J. Coates, S.A. Cunningham, M. McDonald, P.A. Vesk and C. Yates. 2008. Seed supply for broadscale restoration: maximizing evolutionary potential. *Evolutionary Applications* 1:587–597.

Fant, J.B., R.M. Holmstrom, E. Sirkin, J.R. Etterson and S. Masi. 2008. Genetic structure of threatened native populations and propagules used for restoration in a clonal species, American beachgrass (*Ammophila breviligulata* fern.). *Restoration Ecology* 16:594–603.

Galloway, L.F. and C.B. Fenster. 2000. Population differentiation in an annual legume: Local adaptation. *Evolution* 54:1173–1181.

Gibson, D.J., S.G. Baer, R.P. Klopf, L.K. Reed, B.R. Wodika and J.E. Wiland. 2013. Limited effects of dominant species population source on community composition during community assembly. *Journal of Vegetation Science* 24:429–440.

Golay, M.G., R. Manatt, C. Mabry, J. Thompson and R. Kolka. 2013. Restoration of herbaceous woodland plants: Persistence, growth, and reproductive success of local and non-local propagules. *Ecological Restoration* 31:378–387.

Hufford, K.M. and S.J. Mazer. 2003. Plant ecotypes: Genetic differentiation in the age of ecological restoration. *Trends in Ecology and Evolution* 18:147–155

Hufford, K.M., S.L. Krauss and E.J. Veneklaas. 2012. Inbreeding and outbreeding depression in *Stylidium hispidum*: Implications for mixing seed sources for ecological restoration. *Ecology and Evolution* 2:2262–2273.

Keller, M., J. Kollman and P.J. Edwards. 2000. Genetic introgression from distant provenances reduces fitness in local weed populations. *Journal of Applied Ecology* 37:647–659.

Linhart, Y.B. and M.C. Grant. 1996. Evolutionary significance of local genetic differentiation in plants. *Annual Review of Ecology and Systematics* 27:237–277.

Pickup, M., D.L. Field, D.M. Rowell and A.G. Young. 2012. Source population characteristics affect heterosis following genetic rescue of fragmented plant populations. *Proceedings of the Royal Society* 280:1–9.

Potter, K.M. and W.W. Hargrove. 2012. Determining suitable locations for seed transfer under climate change: A global quantitative method. *New Forests* 43:581–599.

Saari, C. and W. Glisson. 2012. Survey of Chicago Region restoration seed source policies. *Ecological Restoration* 30:162–165.

Waser, N.M. and M.V. Price. 1985. Reciprocal transplant experiments with *Delphinium nelsonii* (Ranunculaceae): Evidence for local adaptation. *American Journal of Botany* 72:1726–1732.