AFFORESTING AGRICULTURAL LANDS IN THE MISSISSIPPI ALLUVIAL VALLEY (USA): EFFECTS OF SILVICULTURAL METHODS ON UNDERSTORY PLANT DIVERSITY

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Abstract--To compare methods for bottomland hardwood reforestation on marginal farmlands in the Mississippi Alluvial Valley, four afforestation treatments (natural colonization, sown oak acorns, planted oak seedlings, cottonwood–oak interplant) were established in 1995 on former soybean cropland. Natural, sown, and planted-oak plots were not managed after establishment. Interplant plots received intensive management including two seasons of weed-control disking between planted cottonwoods, after which oaks were interplanted. Previous work found that forest canopy development was accelerated by interplanting; however, the best methods for establishing trees could have different effects on forest community diversity. Multi-year data on understory plant composition were analyzed to determine if less intensive methods promoted greater diversity. Ground-layer vegetation was sampled annually from 1996 to 1998, and again in 2006. Only total biomass was affected by afforestation technique, with the greatest declines in the interplant treatment. Changes in all species composition measures were a function of successional time. Although diversity did not vary substantially with reforestation method, lack of hydrologic restoration favored an understory flora more typical of moist old-fields than natural floodplain forests.

INTRODUCTION

Bottomland hardwood forests once covered 10 million ha in the Lower Mississippi River Alluvial Valley (LMAV). By the 1980s, large flood-control projects and land clearing for agriculture had reduced forest extent by roughly 75 percent (Haynes 2004). This historic forest loss has been addressed by various reforestation efforts in the past few decades (King and Keeland 1999, King and others 2006, Schoenholtz and others 2001). Reforestation methods (termed afforestation when converting from agriculture or other non-forest land use) have varied from completely passive to intensive, depending on land ownership and management objectives. Active, low-cost methods were favored as a way to establish desired species (mainly ‘hard-mast’ oaks, *Quercus* spp.) and overcome the dispersal limitations of passive colonization. However, the mixed results from active low-intensity techniques raised questions about potential tradeoffs between satisfying habitat/diversity objectives versus enhancing economic returns (such as timber yield) on private lands (Haynes 2004, Stanturf and others 2001, Twedt and Wilson 2002).

A long-term experiment was established in 1995 to compare four afforestation methods ranging from passive to intensive, so that ecological and economic trade-offs could be assessed at operational scales (Gardiner and others 2008, Schweitzer and Stanturf 1999). The methods were natural tree colonization, establishing oak species by direct-seeding or by planting, and interplanting oak seedlings with a fast-growing early-succession tree species. A specific goal was to evaluate if the interplant method could accelerate forest development for both timber and habitat values. Results from this experiment and analogous studies indicated that interplanting favors rapid development of tree height, vertical structure and canopy closure, whereas less intensive methods may allow for greater tree diversity (Stanturf and others 2009, Twedt 2004, Twedt and Wilson 2002).

Most of the afforestation research has focused on forest structure or overstory tree diversity. Understory plant composition is an aspect that has been evaluated only infrequently. Ground-layer biomass was assessed in the long-term experiment as a source of competition for planted tree seedlings (Stanturf and others 2009), but this layer is also a component of community diversity and contributes to habitat values. Intensive afforestation methods, while favorable for tree development, could have negative effects on ground-layer plant diversity. In this paper, we analyze data from the long-term experiment to determine if alternative methods led to differences in understory plant composition.

STUDY SITE AND METHODS

The long-term experimental site is located in Sharkey County, MS, on a tract that was in soybean cultivation until fall 1994. The soils are

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Citation for proceedings: Holley, A. Gordon; Connor, Kristina F.; Haywood, James D., eds. 2015. Proceedings of the 17th biennial southern silvicultural research conference. e–Gen. Tech. Rep. SRS–203. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 551 p.

mapped as shrink-swell clays (Vertisols) in the Sharkey series (Pettry and Switzer 1996). The area is in the Big Sunflower River drainage, part of the Yazoo River basin of the LMAV. Portions of the site may receive dormant-season backflooding in some years (Stanturf and others 2009), but generally the area is isolated from natural flooding of the Yazoo and Mississippi Rivers by an extensive system of flood-control levees and ditch-channels (cf. Faulkner and others 2011, Frederickson 2005).

The basic experimental design is summarized here; see Schweitzer and Stanturf (1999) and Stanturf and others (2009) for detailed descriptions. The experiment was a randomized complete blocks design, with three blocks of four treatment plots representing a gradient of silvicultural intensity: (1) natural tree colonization; (2) sown Nuttall oak acorns (*Q. texana* Buckl.); (3) planted Nuttall oak seedlings; and (4) phased interplanting of cottonwood (*Populus deltoides* W. Bartram ex. Marsh.) and Nuttall oak. Treatment plots were 8.1 ha (20 acres) in size. All plots were prepared by disking prior to establishment. Natural colonization plots had no other manipulation. Direct-seed and oak-seedling plots were planted during March through May 1995 and then received no further management. Acorns were sown at 1.1- by 3.7-m spacing (2,457 acorns/ha), and seedlings were planted at 3.7- by 3.7-m spacing (730 seedling/ha). Also in March 1995, cottonwood cuttings were planted in the interplant plots at 3.7- by 3.7-m spacing. These plots were treated with fertilizer, herbicides and pesticides. Additionally, sub-plot sections received either one or two seasons of weed-control disking between cottonwood rows before oak seedlings were interplanted in March 1997 at 3.7- by 7.4-m spacing (365 seedlings/ha). Cottonwood thinnings and harvest were scheduled for the year 2007 to study yields and eventual success of oak release.

For purposes of this study, the 'understory' layer was considered to be all ground-level vegetation (excluding tree seedlings) between planted or volunteer trees. Understory sampling occurred in years 2 through 4 after plot establishment (1996-1998) and again in year 12 (2006). Eight stratified-random 1-m^2 quadrats were sampled per treatment plot (interplant plots were sampled with eight quadrats per one-/two-season disked sub-plots in years 2 through 4 only). In late August-September of each year, all ground-layer vegetation (herbs, shrubs, woody vines) in each quadrat was clipped to ground level, sorted to species, and dried at 40 °C (104 °F) to obtain dry-weight biomass (g/m^2) as a metric of species abundance. In the analyses, we used the quadrat data from areas disked for two seasons (1995, 1996) to represent the 'intensive' interplant treatment; this equalized sample area to eight quadrats per treatment plot and provided a balanced statistical design for all years. Diagnostic tests verified that the plant data from the interplant plots did not differ between the one- and two-season disked areas for any sampling year.

Plot-level data were analyzed in a repeated-measures ANOVA model with afforestation treatment as the between-blocks factor and year as the within-blocks repeated measure (total n = 48). Analyses were performed in SYSTAT® (SPSS, Inc.). The F-test results were comparable to Greenhouse-Geisser and Huynh-Feldt adjusted *p*-values, indicating that model assumptions were appropriate (Wilkinson and Coward 1999). Variables were total biomass (g/m^2, averaged over 8 quadrats per plot), species richness per plot, and species composition (numbers and relative percentage biomass) in terms of growth form, wetland indicator class, and nativity. Growth forms were classed as herbaceous broad-leaved (forbs), herbaceous graminoid (grasses/sedges), or woody (shrubs/vines). Species indicator classes were defined from five categories (Reed 1997) as either wetland (OBL, FACW categories), facultative (FAC), or upland (FACU, UPL). Native or non-native (introduced) status was obtained from the U.S. Department of Agriculture PLANTS database (http://plants.usda.gov). Excepting total biomass, no variable had significant treatment effects; therefore, only the year and year-by-treatment tests are reported. Changes in presence and abundance between years 2 and 4 and year 12 were summarized for the 30 prevalent species that comprised > 95 percent of total biomass.

RESULTS

Total biomass (fig. 1) was the only variable that differed among afforestation treatments ($F = 30.1$; $df = 3, 6$; $P = 0.001$); weaker year and year-by-treatment effects ($P = 0.04$) were an artifact of low biomass in the 1996 natural treatment. Understory biomass was highest in the natural and sown treatments and lowest in the interplant treatment.

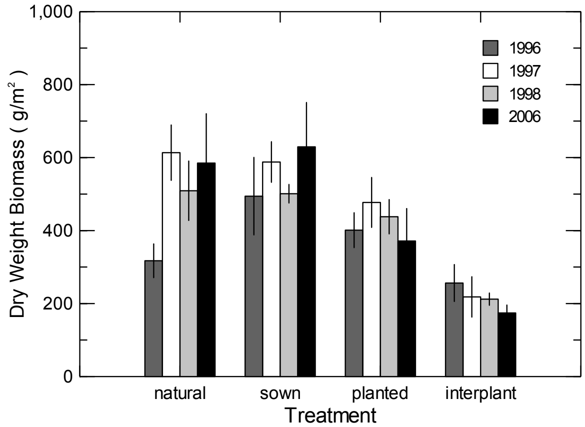

Figure 1 – Changes in ground-layer biomass among afforestation treatments and years. Data are means ± s.e. for n = 3 replicate blocks.

Species richness and all compositional variables differed among years, reflecting successional change over the 12-year period (tables 1 and 2). There were essentially no year-by-treatment interactions. Species richness per plot declined with time, mainly owing to losses of forb species (table 1). Number of herbaceous species (forbs and graminoids) collectively decreased from 13 to 8 species, while the number of woody species increased slightly. Relative biomass of herbaceous plants decreased from 84 to 33 percent, with woody plants increasing to 67 percent of total biomass by year 12. Non-native species were negligible by year 12 (table 1).

With respect to wetland indicator class (table 2), the number of upland species decreased over time, while the number of wetland species fluctuated slightly. Relative biomass of both upland and wetland species declined as the relative biomass of facultative species increased. The net result was that ‘hydrophytic’ species (wetland plus facultative) comprised nearly 75 percent of total biomass by year 12, mainly owing to facultative species.

The temporal trends reflected changes in particular species and species-groups (table 3). Many forb species were no longer detected after 12 years, particularly asters (*Symphyotrichum* spp.) and various upland annuals. Forb biomass became dominated by perennial clonal goldenrods (*Solidago* spp.) and marsh elder (*Iva annua* L.), a robust annual. Graminoid biomass shifted from the highly dominant Johnson grass [*Sorghum halepense* (L.) Pers.], an introduced upland species, to native broomsedge grass (*Andropogon virginicus* L.) and sedges/rushes (*Carex* spp., *Juncus* spp.). The increasing woody biomass became dominated by facultative vines such as trumpet creeper [*Campsis radicans* (L.) Seem. ex Bureau] and poison ivy [*Toxicodendron radicans* (L.) Kuntze], and by blackberry shrubs (*Rubus* spp.). Of 77 identified taxa found over all years combined (data not shown), only 16 of 62 herbaceous species were present by the final year, whereas woody species had increased from only 3 in year 2 to 13 by year 12.

DISCUSSION

Effects of Afforestation Method

The main response to the afforestation treatments was decreased ground-layer biomass with greater silvicultural intensity. Unsurprisingly, this pattern was inverse to the gradient of canopy development. Average tree density after 7 years was lowest in natural and sown plots, intermediate in oak-planted plots, and highest in interplant plots (Hamel 2003). Tree heights after 3 years averaged < 2 m for recolonizing trees and sown/planted oaks versus 8 m for the cottonwoods; this height difference widened to 3 to 4 m versus > 14 m by year 7 (Hamel 2003, Stanturf and others 2009). The much lower ground-layer biomass in the interplant plots (fig. 1) also reflected an effect of rapid cottonwood height-growth, which allowed woody vines to climb vertically and thus displaced some vine biomass to the canopy layer (Personal observation. 2013. S.C. Hughes, Biological Science Technician, USDA Forest Service, Southern Research Station, Stoneville, MS 38776). Twedt and Wilson (2002) also found an inverse pattern attributable to differences in

Table 1--Temporal change in total species richness, and in numbers and relative biomass (percent) of species by non-native status and by growth form. Data for each year are per-plot means (s.e.) over blocks and treatments (n = 12)

Variable	1996	1997	1998	2006	Year effect[a]	Year-by-treatment interaction[a]
			number of species			
Species richness	16 (1)	16 (1)	12 (1)	14 (1)	*	ns
Non-native species	4 (0.4)	2 (0.2)	2 (0.4)	0.3 (0.1)	**	ns
Herbaceous forbs	10 (1)	10 (1)	7 (1)	4 (0.4)	**	ns
Herbaceous graminoids	3 (0.5)	4 (0.4)	3 (0.5)	4 (0.4)	n.s.	ns
Shrubs/woody vines	2 (0.1)	2 (0.1)	2 (0.2)	5 (0.4)	**	ns
			percent of total biomass			
Non-native biomass	47 (6)	37 (8)	21 (6)	0.1 (0.1)	**	ns
Forb biomass	40 (6)	46 (8)	63 (7)	29 (5)	**	*
Graminoid biomass	44 (6)	40 (7)	23 (5)	4 (1)	**	ns
Shrub/woody vine biomass	15 (3)	13 (4)	14 (5)	67 (6)	**	ns

[a]ANOVA F-test significance for year (df = 3, 24) and interaction (df = 9, 24) is noted as *P < 0.05, **P < 0.01, ns = not significant.

Table 2--Temporal change in numbers and relative biomass (percent) of species by wetland indicator class. Data and ANOVA tests as in table 1

Variable	1996	1997	1998	2006	Year effect[a]	Year-by-treatment interaction[a]
			number of species			
Wetland species	4 (0.3)	5 (1)	3 (0.5)	4 (0.3)	*	ns
Facultative species	7 (1)	7 (0.5)	6 (1)	6 (0.5)	ns	ns
Upland species	4 (0.3)	3 (0.4)	2 (0.4)	2 (0.2)	**	ns
			percent of total biomass			
Wetland species	13 (3)	27 (5)	47 (6)	7 (1)	**	ns
Facultative species	35 (7)	35 (5)	29 (5)	67 (4)	**	ns
Upland species	52 (8)	38 (7)	24 (5)	26 (4)	**	ns

[a]ANOVA F-test significance for year (df = 3, 24) and interaction (df = 9, 24) is noted as *P < 0.05, **P < 0.01, ns = not significant.

canopy development on reforested sites across the LMAV, with greater tree heights and lower ground-layer cover on tracts planted in oak seedlings compared to direct-seeded tracts.

Despite the contrast in forest structure between interplant plots and other treatments, afforestation method had no notable effects on understory species composition. One likely reason is that cottonwood canopies are not dense, thus light penetration to ground level (Gardiner and others 2001) would allow species of more open treatments to persist in the interplant plots as well. Possibly the species composition of interplant plots differed subtly in the first sampling year (year 2), when quadrat placement in that treatment was adjusted (as needed) to sample vegetated spots adjacent to bare soil patches that had not yet regrown from after disking (cf. Methods). However, species composition did not differ among treatments in years 3 and 4, suggesting that any subtle effects of the early disking were ephemeral.

Successional Change

Biomass as an abundance metric may overweight the contribution of woody species to total plant coverage, but the data were representative of species trends over time. The changes in plant composition paralleled the typical pattern of succession on abandoned fields and afforested tracts across the LMAV

Table 3--Change in mean dry-weight biomass of abundant herb-layer species from years 2 through 4 (1996–1998, averaged) to year 12 (2006). Species are grouped by growth form and wetland indicator class. Species nomenclature follows the USDA PLANTS database (http://plants.usda.gov)

Species	Indicator class	Life form[b]	1996–1998	2006
			------------ g/m^2 ------------	
Herbaceous broad-leaved				
Ipomoea wrightii[a]	wetland	vine (A)	2.1	0
Lythrum alatum	wetland	forb (P)	5.6	5.8
Sesbania sp. (*herbacea/exaltata*)	wetland	forb (A)	2.0	1.6
Symphyotrichum divaricatum	wetland	forb (A)	6.9	0
Symphyotrichum lanceolatum	wetland	forb (P)	3.0	0
Symphyotrichum spp. (*dumosum, ontarionis, pilosum*)	facultative	forb (P)	51.2	0
Ambrosia trifida	facultative	forb (A)	4.7	0
Iva annua	facultative	forb (A)	9.1	17.4
Desmanthus illinoensis	facultative	forb (P)	15.4	0
Eupatorium serotinum	facultative	forb (P)	2.2	1.4
Ambrosia artemesiifolia	upland	forb (A)	2.0	0
Chamaesyce spp. (*hyssopifolia, nutans*)	upland	forb (A)	2.2	0
Rudbeckia hirta	upland	forb (A)	9.0	0
Sida spinosa	upland	forb (A)	8.1	0
Triodanis biflora	upland	forb (A)	7.0	0
Solidago altissima/gigantea	upland	forb (P)	88.5	72.5
Herbaceous graminoid				
Carex frankii	wetland	sedge (A)	3.3	3.6
Juncus spp. (*dichotomus, diffisissimus*, others)	wetland	rush (P)	1.2	3.8
Andropogon virginicus	facultative	grass (P)	2.4	6.9
Paspalum dilatatum[a]	facultative	grass (P)	6.6	0
Setaria spp. (*geniculata, glauca*[a])	facultative	grass (P/A)	5.1	0
Sorghum halepense[a]	upland	grass (P)	125.5	0.1
Woody shrub/vine				
Brunnichia ovata	wetland	vine	13.5	15.6
Ampelopsis arborea	facultative	vine	0	3.9
Campsis radicans	facultative	vine	41.0	173.2
Toxicodendron radicans	facultative	vine	0.1	106.2
Rubus trivialis	facultative	shrub	0.9	2.5
Rubus spp. (*argutus*)	upland	shrub	0.4	25.2

[a]Non-native.
[b] (A) = annual, (P) = perennial.

(Battaglia and others 2002, Twedt 2004). Early-succession herb species, particularly annuals and non-native agricultural weeds, were excluded by expansion of perennial and woody species. Only 15 of the 30 most prevalent species were detected after 12 years, with dominance shifting to woody vines, blackberry shrubs, and old-field herbs such as goldenrod and broomsedge.

The vegetation became more hydrophytic, mainly owing to greater importance of facultative woody vines. These vines are common species in native floodplain forests (Sharitz and Mitsch 1993); many are animal-dispersed and thus could readily colonize afforested sites. In contrast, wetland species were under-represented. The understory of experimental plots lacked many typical herbs of floodplain forests, such as wetland sedges (*Carex lurida* Wahlenb., *C. louisianica* L.H. Bailey), cutgrasses (*Leersia* spp.), panic-grasses [*Phanopyrum gymnocarpon* (Elliott) Nash, *Panicum rigidulum* Bosc ex Nees), false-nettle [*Boehmeria cylindrical* (L.) Sw.], lizard's tail (*Saururus cernuus* L.), water-willows (*Justicia* spp.), and bugle-weeds (*Lycopus* spp.) (Sharitz and Mitsch 1993, Wharton and others 1982). Few of these species are animal-dispersed, and many may depend on floodwaters for their distribution.

They may also be less likely to persist in the seedbank following intensive farming (Middleton 2003).

Implications

Ability of floodplain plants to colonize reforestation sites in the LMAV will be limited by relative isolation from river flooding and from remnant native forests. Local habitat conditions also constrain understory plant composition. Intermittent backwater flooding at the study site is largely rainfall-driven and ephemeral. Because the experiment was designed partly to assess afforestation options for timber production, there was no hydrology manipulation to enhance local wetness conditions for wetland species. In contrast, some LMAV tracts that are afforested under federal conservation programs often include attempts to enhance local site hydrology. Typically, ditches are blocked to create shallow-water and managed moist-soil areas; occasionally, flood-control levees may be breached to allow local flooding from adjacent streams (Hunter and others 2008, King and Keeland 1999, King and others 2006). Such practices can promote greater abundance of wetland plants, but they do not restore the original river flooding regime (see Lockaby and Stanturf 2002).

Active reforestation efforts can be successful in providing forest habitat, productivity, and carbon-sequestration functions (Faulkner and others 2011, Hamel 2003, Haynes 2004). Given the constraints on restoring natural river hydrology in the LMAV, a continuing challenge is that these efforts will not necessarily replicate the plant diversity of native bottomland forests. Understanding the potential trade-offs of alternative methods can assist in selecting restoration options that meet a variety of ecological and landowner objectives.

ACKNOWLEDGMENTS

This study was supported in part by funding from NCASI (National Council of the Paper Industry for Air and Stream Improvement), International Paper Co., and the Southern Research Station, and by in-kind assistance from Crown Vantage Corp., U.S. Fish and Wildlife Service, and Yazoo National Wildlife Refuge. Dexter Bland, Bryan Britton, Todd Parker, and Allen Moore provided technical support. We thank Charles Bryson for assistance with plant identification, Ray Souter for statistical advice, and Katherine Elliott and Rebecca Sharitz for content reviews.

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