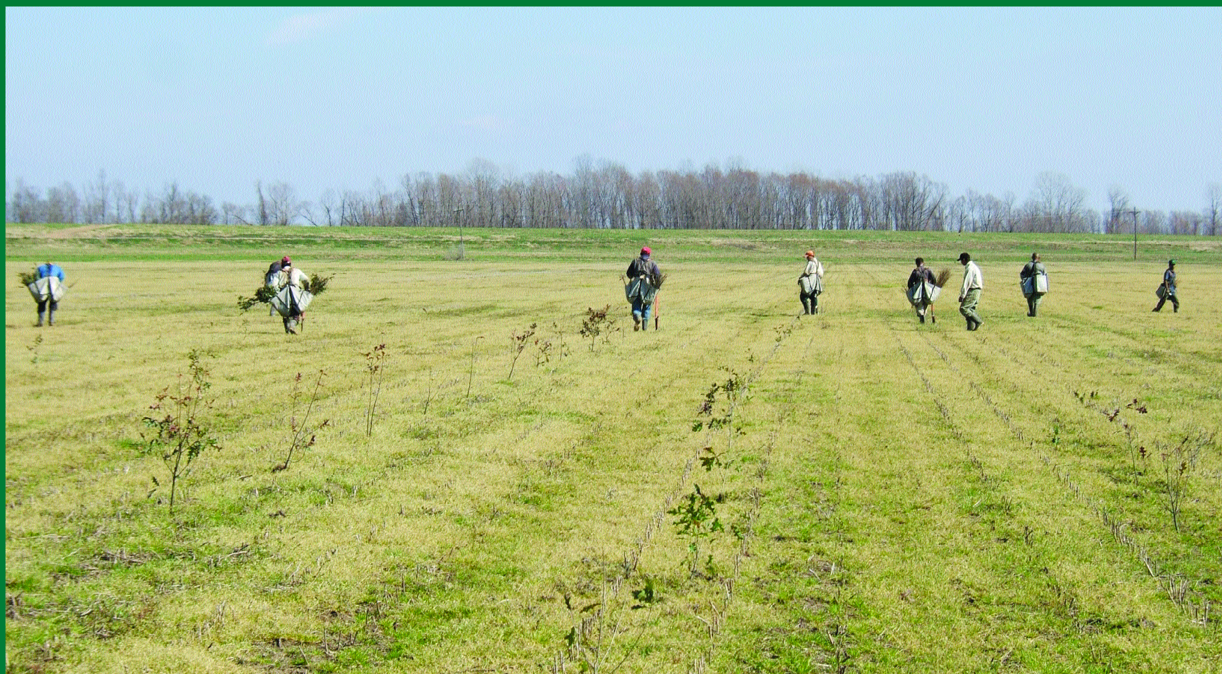
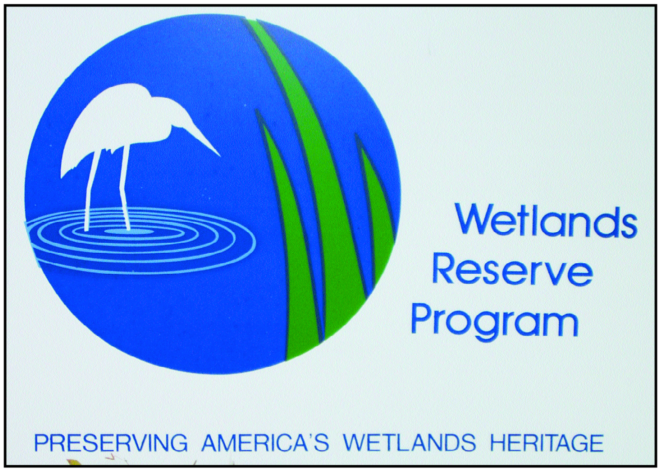


Seedling planting following soybean harvest - LDWF

RESTORATION OF BOTTOMLAND HARDWOOD FORESTS

As early as the 1960s land managers within the MAV expressed concerns over the degradation of forested habitats due to hydrologic and geomorphic alterations to river systems and widespread clearing of bottomland hardwood forests. Although little information was available on reforestation techniques, these land managers began a trial and error process to plant trees on abandoned agricultural land (Tim Wilkins, Yazoo NWR, personal communication, Savage et al. 1989). In most cases, available land consisted of heavy clay soils that flooded too frequently to be profitable for agriculture. More serious efforts to restore forest cover on lands converted to agriculture began in the mid-1980s (Allen and Burkett 1996) when the U. S. Fish and Wildlife Service, Arkansas Game and Fish Commission, Louisiana Department of Wildlife and Fisheries and Tennessee Wildlife Resource Agency increased their tree planting efforts (Savage et al. 1989, Newling 1990). These efforts were furthered through contacts and field review meetings of the Southern Hardwood Forestry Group with input from researchers at the U. S. Forest Service's Center for Bottomland Hardwood Research in Stoneville, Mississippi (M. Blaney, personal communication). In most cases, the sole activity was to plant two or three species of trees with little monitoring of vegetation or wildlife response.

Since 1987, public agencies and private interests have reforested circa one million acres (R. Wilson, personal communication), with suggested restoration targets of more than two million acres (Haynes 2004). Numerous state and federal agencies have contributed to these totals, but the advent of the USDA's Wetland Reserve Program greatly accelerated reforestation efforts (King et al. 2006). The 1990 Farm Bill established the WRP, a voluntary program that provides technical and financial assistance to eligible landowners to restore wildlife habitat on wetlands through planting of vegetation and limited hydrologic restoration. Haynes (2004) stated that "*The Wetland Reserve Program is perhaps the most significant and effective wetland restoration program in the world, and has provided a tremendous opportunity to restore forested wetlands*." As of September 2004, nationwide there were 7,831 projects on 1,470,998 acres enrolled in the Wetland Reserve Program. Through 2005, more than



USFWS

USDA Wetland Reserve Program

680,000 acres have been enrolled in Louisiana, Arkansas, and Mississippi (King et al. 2006).

HISTORICAL PERSPECTIVE

Historically, hardwood forest restoration was intended to create diverse forest habitat for wildlife and a sustainable timber harvest (Wilson and Twedt 2005). Unfortunately, most of the early restoration occurred opportunistically, resulting in isolated blocks of restored forest (i.e., little contribution to the reduction of forest fragmentation). Additionally, many of the restored sites had relatively low topography (i.e., flood-prone sites), coupled with a failure to properly match tree species with site conditions (Stanturf et al. 2001) that resulted in poor tree survival. These mismatches of tree species and site conditions are less frequent in current practice.

Despite high diversity of tree species in bottomland forests (Allen 1997), plantings on bottomland sites have historically focused only on a few species of slower-growing, hard-mast producing trees. The species selected for restoration are typically based on their mast-production, their seed dispersal method (e.g., heavy-seeded, poorly dispersed species were favored), and their value as timber. Indeed, one study (King and Keeland 1999) indicated that within the MAV >80% of all planted species have been oaks or sweet pecan, although the diversity of plantings has increased more recently.

Few guidelines exist regarding optimal planting densities (Lamb 1999). Historically, a density of 302 seedlings / acre (12 x12 ft spacing) has been used in most bottomland forest restoration in the MAV (King and Keeland 1999). Early restorations often employed direct seeding due to the low cost of acorns and sowing (Johnson and Krinard 1987, Haynes et al. 1995). However, unpredictable survival within direct seeded restorations (due to seed and/or planting qualities) has prompted greater reliance on planting bare-root seedlings despite greater cost.

DESIRED FOREST CONDITIONS

Forest restoration is the most important method by which we can achieve largely forested landscapes. However, reforestation has historically been extensive with an intent to "plant as many acres as possible," despite a lack of clearly defined site-specific objectives linked to succinct landscape objectives (Wilson et al. 2005). Although this approach may have been initially warranted, it fails to recognize important components of successful ecosystem restoration (e.g., succinct objectives linked to wildlife population response) (Young 2000). Obviously, the establishment of clearly defined focal areas and restoration priorities is necessary to effectively meet landscape


Brant Miller

Planting bare-root hardwood seedlings with a dibble bar


Lamar Dorris

An approximately 3-year-old reforestation site conservation objectives (see *Priority Wildlife Species and Habitat Objectives; Table 1*) (Llewellyn et al. 1996, Twedt et al. 2006). Over the last 5-10 years, conservation objectives have been used more effectively in prioritizing bottomland hardwood restoration (e.g., use of songbird, *Figure 4 and 5*, and black bear, *Figure 6,* decision support tools in the ranking of WRP).

Concurrently, the "one-size-fits-all" approach has often been used for restoration within sites, as evidenced by commonly planting few species (primarily oaks) at a standard density of 302 seedlings/acre (12x12 ft spacing). Evaluation of the subsequent development of these plantings suggests that many have failed to attain a diverse species composition or structural complexity, in the absence of additional site invasion by native species. Furthermore, it appears that many planted stems are unlikely to develop characteristics that will lead to quality timber production, thereby limiting forest management options to meet DFCs. Thus, site development following historical restoration methods appears unlikely to provide desired stand conditions (see *Management of Bottomland Hardwood Forests*) without additional silvicultural manipulations or extended periods of time. Below we articulate recommendations for bottomland restoration that target attainment of both desired landscape conditions and development of desired stand conditions.

LANDSCAPE SCALE CONSIDERATIONS

Many priority wildlife species are dependent upon large, forested landscapes that harbor contiguous bottomland hardwood forests. Thus, in general, our primary landscape conservation goal is to establish and maintain extensive areas of contiguous bottomland forest within distinct local landscapes (see *Priority Wildlife Species and Habitat Objectives*; Wilson et al. 2005).

Shauna Ginger

Black bear cub

Although small isolated, or long linear tracts may provide important wildlife habitat (e.g., as bear movement corridors), these sites are likely of lesser value to forest-breeding songbirds. An alternative management strategy for these sites may be to plant and maintain these areas in shrubby, early successional habitat (see below). Depending on topographic diversity, these sites may also be important for reptiles and amphibians. Both environmental and spatial variables influence amphibian assemblages (Parris 2004, Loehle et al. 2005) but Burbrink et al. (1998) noted that patch size was less important than topographic diversity for these species.

Sammy King

Mole salamander

When planning restoration at the landscape scale, sites with higher elevations should be considered as they have been underrepresented in previous restoration activities. As historic opportunity for restoration has largely been on flood-prone sites, higher elevation bottomland sites (e.g., ridges and second bottoms; *Figure 3*) have rarely been restored. Indeed, most extant bottomland forests in the MAV are on lower sites (Twedt and Loesch 1999, Rudis 2001b) whereas higher elevation sites remain in agricultural production. Functionally, higher sites provide unique habitat resources that are unavailable or limited on lower sites. For example, during major flood events, many forest interior species (e.g., ground foraging songbirds, deer, turkey, etc.) must find alternative habitat when displaced from flooded forests. Furthermore, higher elevation sites often have temporary, fishless wetlands that are important for many species of amphibians (Burbrink et al. 1998).


Michael A. Kelly
Cotton on higher elevation soils

Restoration of these higher sites should be a priority, but there are economic, social and political challenges. Economically, these sites are more productive agricultural areas and the costs of acquiring these sites will be considerably higher than marginally productive agricultural areas. Socially and politically, the loss of agricultural revenues from rural communities is a concern and will likely be met with resistance (R. Wilson, personal observation). Loss of farming activities can further impact rural communities as the need for services supporting this practice is diminished. The lag time between reforestation and forest harvesting can be hard on the local economies currently dependent on farming activities. These and other concerns must be appropriately addressed.

Opportunities may exist to gain substantial benefits from concurrent functions when they are considered in the selection process. These "secondary" functions can potentially enhance the success of restorations. For example, selecting sites for restoration that are known sediment sources or that are important sediment sinks may enhance the long-term condition of existing forests in a watershed.

Conversely, there may be conflicting landscape-based forest restoration objectives among priority wildlife species. For example, managers may have to choose between forest restorations or herbaceous moist soil intended for waterfowl. In these situations, the potential benefit of reduction in forest fragmentation will have to be balanced against maintaining non-forest habitat (e.g., moist soil units or managed agricultural areas) that benefit waterfowl and other waterbirds. The effect of landscape position on other wetland functions (e.g., carbon sequestration, water quality


USFWS
Aerial view of reforestation and hydrology restoration

enhancement) and other species of wildlife (e.g., amphibians and reptiles) should also be considered.

STAND LEVEL CONSIDERATIONS

Site Limitations

Forest composition within the MAV is highly correlated with hydrogeomorphic setting (Klimas et al. 2005). Thus, we suggest that forest restoration is likely to be most successful when restoration accounts for the effects of micro-topography, hydrology, soils and geomorphic setting on plant species composition. To that extent, most, if not all restoration sites have undergone hydrological changes/alterations. Although restoration of original hydrologic conditions may not be possible because of physical land use changes and/or socioeconomic constraints, restoring or emulating local hydrologic processes through re-contouring of lands or through active wetland management is encouraged. Flooding was and is a critical component of forested wetlands with ecosystem productivity and life cycles of many organisms linked to these hydrologic processes. Thus, restorationists should evaluate opportunities for hydrologic restoration or rehabilitation prior to selecting plant species for restoration.

Differences in soils and hydrology, among and within restoration sites, mandate that for optimal tree growth and survival, species selections must be compatible with site conditions (Baker and Broadfoot 1979, Patterson and Adams 2003, Lockhart et al. 2006). On sites with varied topography (e.g., ridge and swale), matching species with site conditions should result in increased heterogeneity of species and structure (Groninger 2005). However, on sites that are often inundated, soil with uniform topography or with homogeneous soils, planting only a few site-compatible species may be warranted.

Species diversity

The high diversity of tree species found within bottomland forests (Allen 1997) provides a great variety of wildlife habitat. However, previous restoration has focused on ensuring establishment of hard-mast producing trees, primarily oaks with the assumption that diversity would result from naturally colonizing light-seeded trees. Assessment of established restoration sites has indicated that diversity is often dependent upon distance from existing forest stands (Allen et al.


Bobby Keeland

Limited natural invasion in 7-year-old bottomland oak planting

1998, Battaglia et al. 2002, Twedt 2004, Wilson and Twedt 2005).

Due to limited natural invasion, including a greater diversity of tree and shrub species in reforestation plantings (i.e., mixed-species plantings) is important for successfully attaining long-term conservation goals. Mixed-species plantings have numerous benefits including greater diversity and broader temporal availability of mast and insects, greater structural diversity, higher timber quality and yield, increased non-timber and timber products, improved soil health, enhanced natural regeneration, and increased carbon sequestration (B. Lockhart, U.S. Forest Service, personal communication).

Restored forests that are diverse in woody species provide benefits to priority wildlife by distributing food and shelter resources across space and time. A stable supply of insects is important for the diverse assemblage of forest dwelling bats, all of which are insectivorous. Most migratory birds forage primarily on insects rather than mast during spring and summer, and nestlings are provisioned almost exclusively with insects, especially caterpillars. Many of these caterpillar species exhibit preferences among host tree species (Twedt and Best 2004). Thus, in forests that are depauperate in tree species diversity, some caterpillar species may be rare or absent. Furthermore, the abundance of insects and species-specific fruit (mast) production vary temporally. Black bears have an omnivorous diet that shifts in space and time to exploit available food sources (Stransky and Roese 1984, Rode and Robbins 2000, Benson and Chamberlain 2006). Thus, species rich forests buffer temporal variability resulting in a more stable supply of insects and mast.

A multitude of woody species also provides many growth forms and phenologies that provide varied and seasonally dynamic structural niches. Mixed-species stands also allow for greater structural diversity, and often at a much faster rate than would occur with plantations of primarily heavy-seeded species (Twedt 2004). Mixed species stands can create interspecific competition that can improve timber quality, particularly of oaks, and increase management options in the future (Oswalt and Clatterbuck 2006, Lockhart et al., 2006).

Restorations that incorporate fast-growing tree species promote rapid colonization by silvicolous birds (Twedt et al. 2002, Hamel 2003). For example, eastern cottonwood interplanted with oaks on appropriate sites have proven to be successful in achieving rapid development of vertical structure and providing economic benefits to landowners (Twedt and Portwood 1997, Gardiner et al. 2004, Twedt and Best 2004). Sweetgum interplanted with oaks have also been recommended for providing more rapid development of forest structure. In early stages, sweetgum will outgrow the oaks, but at about 25 years the oaks will attain dominance within planted stands (Lockhart et al. 2006). Additional conceptual models of compatible bottomland species, targeting improved timber quality of oaks, have been proposed for use in establishing multi-species restorations (B. Lockhart, unpublished manuscript).

Bob Strader

Oak/cottonwood interplanting

Although mixed-species plantings are recommended on most sites, another method used to provide rapid height development of trees is to plant plantations exclusively of fast-growing hardwood trees. Plantation forests have been successfully used to achieve diverse forest conditions (Keenan et al. 1997, Lamb 1998). Plantations facilitate forest succession in their understories through modification of both physical and biological site conditions, changing light, temperature and moisture conditions at the soil surface (Lugo 1997). These changes enable germination and growth of seeds transported to the site by wildlife and other vectors (Parrotta et al. 1997, Joslin and Schoenholtz 1998). That these physical changes occur within the understory implies that plantation trees have rapid development of a forest canopy. Diversification of these forests can be further hastened by "under-planting" a mixture of slower-growing and understory tree species, shrubs and vines (Twedt and Portwood 1997, Gardiner et al. 2004), although Allen et al. (2006) identified limitations to this approach (e.g., reduced survival and growth due to low light conditions). As such, these species should be included in the initial planting stock.

Regardless of how achieved, to ensure rapid colonization of a restored site by priority wildlife, trees with rapid growth characteristics must occur on the reforested site. Although there remains a perception that forest diversity, particularly colonization of light-seeded species, will result from natural colonization, it is often necessary to plant several species to ensure species diversity on restored sites. Flooding (i.e., overtopping seedlings) impacts natural colonization of trees but colonization may be restricted by distance from existing seed sources or harsh site conditions (e.g., drought) for seed establishment. When restoration sites are far (>660 ft) from seed sources, natural colonization by woody species may be sparse (Allen 1990, McCoy et al. 2002, Twedt 2004, Wilson and Twedt 2005).

There is no set number of species to be planted per field or project. Forest restoration within some ecosystems, such as rainforests in Australia (Tucker and Murphy 1997) and thamnic forests in Texas (Twedt and Best 2004), have successfully planted up to 80 species at densities of up to 1,215 stems/acre to promote restoration of diversity. While large numbers of species would be beneficial in many areas, in some cases, such as an old baldcypress brake, it might be appropriate to plant only one or two species, baldcypress and button bush. Conversely, in a field with ridge and swale topography, it might be appropriate to plant numerous species. Species found within adjacent forests can be used to guide species selection (i.e., reference sites) for restoration within site limitations. If non-traditional species are candidates for restoration, limited past demand may reduce the availability of planting stock. Thus, land managers may need to communicate planting stock needs with nurseries well in advance (more than one year) of anticipated planting dates.

Stem Density

Some forest resource managers have determined that the planting rate used by most agencies, 302 trees per acre, is sufficient to create habitat beneficial to silvicolous birds (Wilson et al. 2005). However, Stanturf et al. (2001) suggested that the standard currently used to define restoration success, 125-225 trees per acre at or before the third year after planting, is not sufficient to produce commercial timber and recommend survival of 250-450 trees/acre. Historically, it has been assumed that natural colonization of light-seeded species will ensure restored forests are both diverse and stocked at densities more than 250 trees/acre. However, as with diversity, natural colonization cannot be relied upon to produce densely stocked stands when sites are far (>660 ft) from existing forests (Allen 1990, Allen et al. 1998, McCoy et al. 2002, Twedt and Wilson 2002, Twedt 2004). Thus, planting at higher densities may be required to initiate stands at high densities.

High densities of trees and shrubs provide benefits to wildlife by rapidly achieving "forest-like" habitat conditions. Furthermore, these dense, shrub-like habitats often provide important food sources for priority wildlife, in the form of soft, fleshy fruits and small hard seeds. Wunderle (1997) found that sites with greater availability of perches, structurally complex vegetation and food (fruit and insects) resources attract seed dispersers, thereby increasing within site diversity. Some birds of management concern (e.g., Bell's vireo, orchard oriole [*Icterus spurius*], and painted bunting) preferentially breed in shrub-scrub habitats provided by "thickets" of invading trees,

An example of a reforestation tract that has been naturally invaded by other trees

whereas other priority wildlife species use these thamnic areas for post-breeding cover and foraging (Kilgo et al. 1999, Vega Rivera et al. 1999). In areas where species using shrubby habitat are high priority, mangers are encouraged to maintain thamnic habitat through periodic manipulation of vegetation (e.g., burning, disking, chaining or mowing).

Densely stocked stands promote early canopy closure, encouraging vertical development of trees. In addition to the positive correlation between tree height and colonization of sites by silvicolous birds, high sapling densities stimulate development of dominant or emergent trees within stands due to the "shepherd tree" effect that inhibits lateral growth while encouraging apical growth (Gómez-Aparicio et al. 2004, Lockhart et al. 2006). Emergent trees within a multilayered forest canopy provide preferred nest and perch sites for some priority bird species (Hamel 2000).

However, densely stocked stands that allow little sunlight penetration to the forest floor generally harbor few priority wildlife species. Indeed, wildlife would benefit from silvicultural treatments that introduce disturbance and increase structural heterogeneity even in relatively young restored forests. Unfortunately, such silvicultural treatments are not commercially viable and thus are unlikely to occur. A potential alternative to commercial operations is via the acquisition of shared harvesting equipment (e.g., a feller-buncher) capable of felling small diameter trees. Although the cost of such equipment could likely not be justified by a single management area, harvesting units that are regionally based and jointly operated may be feasible.


Lamar Dorris

Lack of vegetation within closed canopy reforested stand

Impediments to increasing density of woody species on restored bottomland sites are both logistic and economic. Increasing the density of planted seedlings markedly increases the cost of restoration. For example, moving from 12-foot spacing (302 seedlings/acre) to eight-foot spacing (680 seedlings/acre) more than doubles the planting stock and labor required for restoration. On the other hand, an increase to 435 seedlings/acre (10-ft spacing) only increases the cost by about 50% and may provide a much preferred basis for attaining DFCs. Although initial costs are higher, planting higher densities of seedlings will likely improve timber quality (e.g., merchantability), as well as enhancing wildlife habitat.

To minimize costs in some situations, the planting rate can be reduced along field margins within 100-660 feet of adjacent forests, where increased rates of natural colonization is likely. Another alternative to reduce costs is the use of direct seeding. Seeds of woody plants cost a fraction of seedlings and can be planted with relatively little time and expense (Allen et al. 2001). Furthermore, Twedt and Wilson (2002) suggested that wildlife (birds) benefit more from direct seeding acorns than from restorations of planted oak seedlings, owing to increased species and structural diversity attained within these sites. Additionally, some land managers have found that direct seeded acorns survive periods of drought or prolonged flooding whereas planted seedlings suffered high mortality under these adverse conditions. However, there are also disadvantages of direct seeding: (1) direct seeding has been proven reliable only for large seeded species, such as oaks, (2) development of direct seeded oaks is generally slower than that of planted seedlings, and (3) rodents may eat sown acorns reducing survival (Savage et al. 1996).

Other woody species and cane have been successfully restored by directly sowing seeds (Holt 1998a, 1998b, Snell and Brooks 1998, Camargo et al. 2002). Unfortunately, little information is available on the methodology or success of directly sown non-hard mast seeds on bottomland sites (Herman et al. 2003, Lof et al. 2004), although Gagnon (2006, Appendix 2) provides recommendations for cane restoration. Allen et al. (2001) and Twedt (2006a) indicated that direct-seeding of light-seeded species has been largely unsuccessful in the MAV. Where successful restorations from direct seeding have been reported, success has often been contingent upon control of weed competition (Herman et al. 2003, Twedt and Best 2004). Weed control also benefits growth of planted trees (Ezell 1995, Ezell and Catchot 1998, Rey Benayas et al. 2005). However, weedy cover can provide beneficial habitat for many wildlife species during these early forest developmental stages. Regardless, limited financial resources and lack of personnel have prevented weed control on most restoration sites. Because of their inability to provide weed control (or other pre-commercial silvicultural treatments; see *Management of Bottomland Hardwood Forests*), many managers are reluctant to risk increased tree mortality by planting species that are susceptible to weed competition. Considerable challenges remain to ensure germination and successful establishment of diverse forests via direct seeding.

When high tree densities can be obtained, caution should be exercised as the resultant dense canopy cover within the maturing forest diminishes its suitability for many wildlife species. Thus, it is advisable to mix densely planted areas with sparse or unplanted areas. One option is to plant small areas or only part of a restoration site with fast growing tree species. These areas of rapid vertical growth potentially serve as ornithochory foci (Werner and Harbeck 1982, McClanahan and Wolfe 1993, Robinson and Handel

1993) that may result in increased diversity and density of trees, but this has not been experimentally proven in the MAV (B. Keeland personal communication). Similar areas of rapid vertical growth may be achieved by isolated trees (Guevara and Laborde 1993), small clumps of trees (Toh et al. 1999, Twedt, 2006b), or linear strips (Twedt and Portwood 2003). Even so, colonization by other woody species at these sites can be slow (Wunderle 1997) and survival poor (Toh et al. 1999), thus necessitating the need for multi-species plantings.

Few guidelines exist as to the relative planting densities of species within multi-species restorations. Historically, restoration has focused on long-lived, commercially valuable species. Even when planted at relatively low densities, intraspecific competition among these species may result in mortality of many of the planted individuals. Conversely, planting of multiple species promotes interspecific competition that results in improved stand development and enhanced wildlife habitat. This approach risks the possibility that some species may be overtopped by faster growing species but many of these species (e.g., oaks) can normally persist and eventually out-compete the faster growing pioneer species (Clatterbuck and Hodges 1988, Johnson and Krinard 1988, Lamb 1998, Lockhart et al. 2006). Moreover, specific mixed species plantings that combine early and late successional species or shade-tolerant and shade-intolerant species have been recommended for quality timber development and wildlife habitat (Ashton et al. 2001, Lockhart et al. unpublished manuscript).

SUMMARY AND RECOMMENDATIONS

Landscape Perspective

> Future conservation efforts should clearly articulate goals and objectives that directly link habitat restoration and habitat needs of priority wildlife.

Following direction provided by restoration objectives, existing decision support tools can be used to focus restoration so as to promote population sustainability of priority species. These support models exist, or are in development, for forest birds, hydrogeomorphology, and natural flooding. We encourage development of additional science based, biologically driven, landscape oriented models for other priority wildlife, particularly the threatened Louisiana black bear. Not only will clear articulation of goals and objectives guide restoration decisions, it will facilitate improvement of restoration efforts through evaluation of both programmatic and ecological success. These results can then be used to adjust management prescriptions via adaptive management.

Site Limitations

As discussed previously, forest distribution and composition are strongly linked to both the geomorphic setting and its associated hydrology. Furthermore, much of the MAV has undergone significant, hydrologic alterations due to flood control activities (e.g., levees) and farming practices (e.g., land-leveling). In an attempt to keep our "eye on the prize," restoration activities should strive to restore local hydrology and topography via re-contouring of land-leveled fields and the promotion of natural hydrologic events.

Due to the comprehensive nature of *A Guide to Bottomland Hardwood Restoration,* we made a conscience decision to not address the many facets of site preparation, handling and storage of seeds/seedlings, etc., from the onset (refer to Allen et. al. [2001] for more detail). However, recent research and anecdotal observations in the use of no-till sub-soiling techniques and chemical weed control warrant further discussion. The use of sub-soiling (aka "ripping") has been shown to increase both growth and survival of planted species, as well as to facilitate planting efforts (And Ezell, personal communication). In addition, the use of post-planting weed control (first-growing season), through the use of chemical applications, has also been shown to increase both growth and survival


Jon Wessman

Subsoiling (aka "ripping") prior to tree planting of planted species (Andy Ezell, personal communication) via reduced competition for resources (i.e. water). As such, we recommend that all restored sites be sub-soiled before planting and that post-planting chemical weed control during the first-growing season be considered where applicable (i.e., when weeds are presumed to be a problem).

Promotion of Vertical Stratification and Horizontal Structural Heterogeneity

Vertical stratification and increased horizontal heterogeneity within restored sites is only possible over time and with maturation of woody vegetation. As such, it seems somewhat premature to include these elements as objectives for initial restorations. However, attainment of desired stand conditions (see *Management of Bottomland Hardwood Forests*) is our ultimate objective regardless of the length of time it takes to be achieved. Thus, initial restoration decisions should target desired forest conditions, including increased species richness and greater structural diversity. Managers should bear in mind that increased diversity of species (including faster growing trees), higher densities of stems, and varied planting strategies (e.g., leaving patches, circa 1-2 acres, unplanted), not only represent a sound initial restoration strategy but also contributes to improved habitat conditions within maturing forests.

Recommended Planting/Survival Rates

To facilitate natural stand development processes (e.g., inter-specific competition) and to increase wildlife habitat, we recommend increasing the initial planting rate to 435 seedlings per acre (10 ft spacing), recognizing that 680 seedlings per acre (8 ft spacing) would be even better. On most sites, hard mast species, including multiple species of oak, sweet pecan, and other hickories (*Carya* spp.), should represent 30% to 60% of planted trees. These proportions are based on three assumptions: (1) that oak-hickory was part of the previous forest composition, (2) that >30% oak composition is needed to ensure an adequate abundance of oak in future stands to maintain high merchantability, thereby enhancing future management options, and (3) that sufficient hard mast production will occur for resident wildlife species (e.g., black bear, white-tailed deer [*Odocoileus viginianus*], wild turkey [*Meleagris gallopavo*], squirrels [*Sciurus* spp.], as well as for migratory waterfowl (e.g., mallard and wood duck). The remaining 40% to 70% of the planted trees should represent a mixture of light seeded, soft mast, and fast growing species (e.g., red maple, persimmon [*Diospyros virginiana*], elm, green ash, sweetgum, sugarberry, blackgum, American sycamore and black willow) that would naturally occur on the site. Other trees that are native to many sites, such as honey locust (*Gleditsia triacanthos*), ironwood (*Carpinus caroliniana*), swamp dogwood (*Cornus drummondii*) and boxelder (*Acer negundo*) should not be forgotten from the mix of available species.

Although wildlife managers on public lands are not striving for commercial products, planting appropriate species mixtures (Lockhart et al. unpublished manuscript) may promote development of merchantable timber and increase management options. Achieving stocking rates of >300 trees per acre three years post-planting, including 75-180 hard-mast producing trees per acre, will also promote these objectives. To increase density of trees, naturally colonizing species should be encouraged. Once established, species composition within these stands can be altered using prescribed silvicultural management. Not only does natural colonization increase species diversity and stem density, these benefits are incurred at essentially no additional cost. This cost savings can be enhanced through judicious planting, wherein locations within restoration sites that are likely to have considerable colonization (e.g., near forest edges) are not planted or selectively planted at lower densities.

LDWF

Eastern wild turkeys