

Reforestation of Bottomland Hardwoods and the Issue of Woody Species Diversity

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Abstract

Bottomland hardwood forests in the southcentral United States have been cleared extensively for agriculture, and many of the remaining forests are fragmented and degraded. During the last decade, however, approximately 75,000 ha of land—mainly agricultural fields—have been replanted or contracted for replanting, with many more acres likely to be reforested in the near future. The approach used in most reforestation projects to date has been to plant one to three overstory tree species, usually *Quercus* spp. (oaks), and to rely on natural dispersal for the establishment of other woody species. I critique this practice by two means. First, a brief literature review demonstrates that moderately high woody species diversity occurs in natural bottomland hardwood forests in the region. This review, which relates diversity to site characteristics, serves as a basis for comparison with stands established by means of current reforestation practices. Second, I reevaluate data on the invasion of woody species from an earlier study of 10 reforestation projects in Mississippi, with the goal of assessing the likelihood that stands with high woody species diversity will develop. I show that natural invasion cannot always be counted on to produce a diverse stand, particularly on sites more than about 60 m from an existing forest edge. I then make several recommendations for altering current reforestation practices in order to

establish stands with greater woody species diversity, a more natural appearance, and a more positive environmental impact at scales larger than individual sites.

Introduction

Bottomland hardwood forests were a prominent feature of the southcentral United States landscape at the time of European settlement. These forests extended along nearly the entire lower Mississippi River alluvial valley (LMRAV) from near the Gulf of Mexico to southern Illinois and on floodplains of smaller river systems throughout the region (Fig. 1). During the last century, however, a large amount of the original acreage has been lost. In the LMRAV, the approximately 9.8 million ha of bottomland hardwood forest present originally were reduced by 1978 to 2.2 million ha (MacDonald et al. 1979). In east Texas, approximately 63% of the original acreage has been lost (Frye 1987).

The primary cause of bottomland hardwood loss has been conversion to agricultural production: approximately 96% of the cleared land in the LMRAV has been devoted to this purpose (MacDonald et al. 1979; U.S. Department of the Interior 1988). Conversion on such a large scale was made possible by the massive flood control projects that began in earnest following disastrous floods in 1912, 1913, 1916, and especially 1927; these projects lowered the risk of flooding on millions of hectares of land with rich agricultural potential (U.S. Department of the Interior 1988; Newling 1990). Additional losses have been caused by construction of flood control and navigation structures, surface mining, and urban development (Brabander et al. 1985; Frye 1987).

While the alternative uses of bottomland hardwood forest land are important economically, the degree of forest clearing has become excessive from both an environmental and an economic perspective. The remaining bottomland hardwood resource is severely fragmented (Rudis 1995), and many of the remaining patches are in a degraded condition due to poor timber management practices or hydrologic alterations. The cumulative environmental impacts of this large-scale loss and degradation have been addressed specifically for bottomland hardwoods by Gosselink and Lee (1989) and for forested ecosystems in general by Harris (1984), Wilcove et al. (1986), and Saunders et al. (1991).

Recognition of the scale and effects of the loss of bottomland hardwoods in the region has resulted in a considerable degree of interest in their restoration. Serious effort at restoration began in the early 1980s, when agencies such as the U.S. Fish and Wildlife Service, the Louisiana Department of Wildlife and Fisheries, and the Mississippi Game and Fish Commission began reforesting former agricultural lands on their refuges (Haynes & Moore 1988; Savage et al. 1989; Newling

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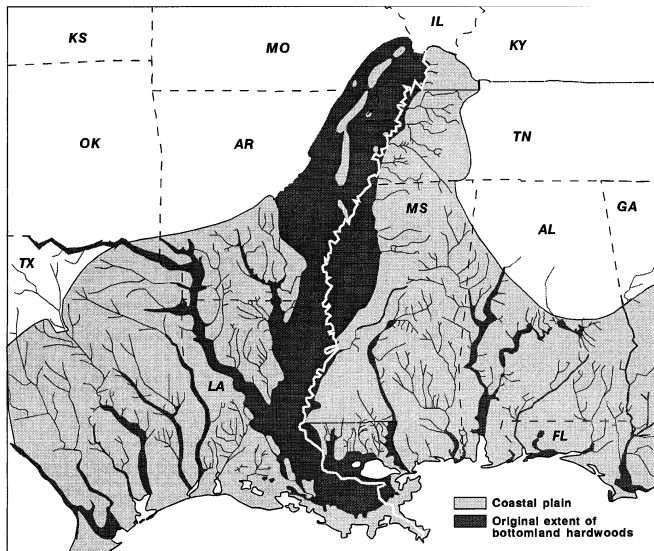


Figure 1. The approximate distribution of bottomland hardwood forests in the southcentral region of the United States (adapted from Putnam et al. 1960).

1990). The pace of reforestation picked up rapidly with the establishment of the Conservation Reserve Program and the Wetlands Reserve Program, federal programs that provided payments to private landowners who planted trees on former agricultural fields. The combined efforts of the agencies and federally sponsored programs have resulted in the planting (or contracting for planting) of approximately 75,000 ha of bottomland forests in the southern United States since 1985; acreage on the same or larger scale may be planted in the coming decade (R. Haynes, U.S. Fish and Wildlife Service, personal communication).

A typical reforestation project on a recently abandoned agricultural field in the region involves the establishment of one to three overstory tree species, usually oaks—especially *Quercus nuttallii* (Nuttall oak), *Q. phellos* (willow oak), *Q. nigra* (water oak), and *Q. pagoda* (cherrybark oak)—either by planting 1-year-old bare-root seedlings or by direct seeding of acorns. The rationale behind planting oaks is that they are mostly shade intolerant, have limited natural dispersal, and have high wildlife and timber value. Also, it is generally assumed that establishment of other woody species will occur through natural dispersal and that a mixed-species forest will develop over time. The techniques commonly used to reforest former agricultural lands have been described by Allen & Kennedy (1989), Newling (1990), Sharitz (1992), and Strader et al. (1994).

Given the severe funding and manpower limitations faced by government agencies and most private landowners, the current approach may be the most effective



Figure 2. An approximately 20-year-old stand of *Quercus pagoda* (cherrybark oak) in the Red River Wildlife Management Area, Louisiana. The rows are still clearly visible, which is typical of many reforested sites in the region. February 1987.

way to achieve long-term restoration. The near-mono-culture condition and plantation appearance of many of the reforested sites established to date (Fig. 2), however, are sources of concern. Although it can be argued that in the long-term reforested stands now dominated by one or a few species will eventually become more diverse, there may be cost-effective ways for forests with greater woody diversity and a more natural appearance to be established initially. My major objective here is to explore this possibility. In order to address the issue more completely, I first briefly summarize some existing descriptions of woody plant species diversity in natural bottomland hardwood forests and then discuss some of the few data available for the region on woody species invasion into reforested sites. Because oaks are such an important component of virtually all reforestation

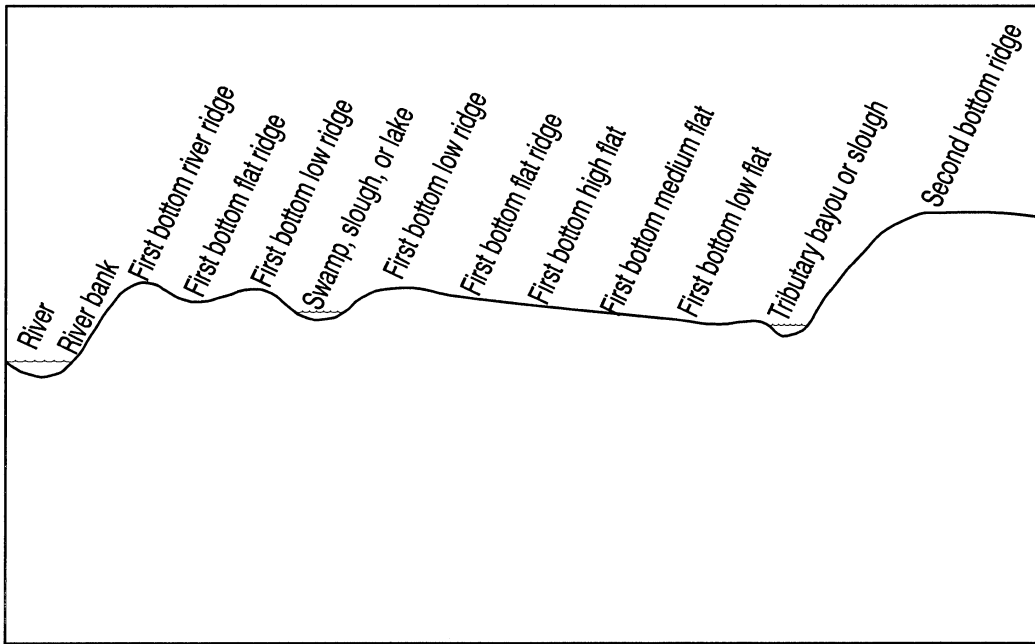


Figure 3. Depiction of floodplain topography from Tanner (1986); most current reforestation is taking place on first bottom flats and lower ridges.

projects in the region, particular attention is paid to their relative importance in the studies reviewed below.

Woody Species Diversity in Natural Bottomland Hardwood Forests

Bottomland hardwood forests in the southcentral region have a degree of woody species diversity (species richness) that compares favorably with the most diverse upland forests in the United States (Marks & Harcombe 1975). Much of the diversity can be attributed to the substantial (albeit subtle) variation in topography typical of southern floodplains, which results in a variety of sites with differing soil characteristics and flooding regimes (Shelford 1954; Bedinger 1981; Wharton et al. 1982; Shankman 1993). This variation in topography has been characterized in different ways, including systems such as those depicted in Fig. 3 and an approach that divides the floodplain into "ecological zones" (Huffman & Forsythe 1981; Wharton et al. 1982).

Several studies have been conducted on the relation of topography to woody species diversity in the southcentral United States. I first summarize studies of this type for old-growth bottomland stands, followed by studies of mature forest with some history of relatively recent human disturbance. In the final part of this section, I summarize studies of stands in early successional stages. In addition to the individual studies described below, several publications list woody species found in bottomland hardwood forests and provide information on the types of sites in which they are found (Putnam et al. 1960; McKnight et al. 1981; Wharton et al. 1982).

One study of an old-growth stand is that of Tanner (1986), who reported on work he conducted in the late 1930s on a tract along the Tensas River in Madison Parish, Louisiana. He described the tree species composition on eight site types ranging from a "swamp, slough, or lake" with near-permanent flooding to a "second bottom ridge" that rarely flooded (Fig. 3). Tanner sampled a total of 12 ha across the eight site types, and in his paper he mentioned 32 tree species that occurred on at least one type. None of the site types defined by Tanner had less than six species; the most recorded for one site type was 15. Nearly one-third of the species occurred on only one site type: eight were found only on second bottom ridge (i.e., least frequently flooded) sites, and two were found only in a swamp, slough, or lake (i.e., sites with frequent to near-permanent flooding). Oaks accounted for more than 75% of the trees greater than 30 cm diameter at breast height (dbh) on the second bottom ridges and first bottom low flats. Oaks typically accounted for 20–30% of the largest trees on all other site types except the swamp, slough, or lake type, which was almost entirely dominated by *Taxodium distichum* (baldcypress) and *Nyssa aquatica* (water tupelo). *Liquidambar styraciflua* (sweetgum), *Celtis laevigata* (sugarberry), *Fraxinus pennsylvanica* (green ash), and *Ulmus americana* (American elm) were also important components of the overstory on most of the site types except the highest and lowest.

Nixon et al. (1977) analyzed woody species composition in an old-growth forest along the Neches River in east Texas. They divided the floodplain into only two site types, ridge areas and more frequently flooded low flats. Of a total of 48 species found, 47 occurred on

ridges and 35 on low flats, indicating a large degree of species overlap between site types. In contrast to most studies from the LMRAV, midstory species such as *Carpinus caroliniana* (ironwood) and *Ilex decidua* (deciduous holly) were very prominent in this forest, especially on the ridges. In the overstory (>30 cm size classes), however, oaks—mainly *Quercus nigra*, *Q. laurifolia* (laurel oak), *Q. lyrata* (overcup oak)—accounted for at least 46% of all trees. The oaks were especially prominent on the ridges, although overcup oak was described as somewhat more abundant in flats, and both *Q. nigra* and *Q. laurifolia* also occurred on some flats. Species that were more common on the flats than on ridges included *F. pennsylvanica*, *T. distichum*, and *Acer rubrum* (red maple).

The highest woody species richness reported for a bottomland hardwood forest in the region comes from the northernmost portion of the LMRAV, where the ranges of many northern and southern species overlap. On Horseshoe Lake Island in southern Illinois, Robertson et al. (1978) found 62 woody species in an old-growth, apparently virgin stand and 59 species in a second-growth stand that has been relatively undisturbed since the early 1900s. Oaks as a group were important components of both stands: a total of 12 species was found and together they accounted for at least 24% of the total basal area of the old-growth stand and 31% in the second-growth stand.

Huffman (1980) characterized the woody vegetation in five relatively uniform and undisturbed (but not old-growth) stands within the Ouachita River Basin of southern Arkansas that varied in flood duration and timing. The number of woody species (excluding vines) found on the sites ranged from 15 to 20, with oaks accounting for four to six of the total on any one site. The number of tree species in the overstory size classes (>15 cm dbh) ranged from 3 to 10, with all but one site having at least six species. Oaks accounted for more than one-third of the basal area of trees greater than 15 cm dbh in four of the stands, with the total proportion in these four stands ranging from 39% to 69%. In the one site where oaks were not prominent, the stand was dominated by *L. styraciflua*, *Ilex opaca* (American holly), and *Carya* spp. (hickories).

In a Louisiana swamp located along the Mississippi River, Devall (1990) found 22 tree species in six community types. The number of species occurring in each community type ranged from 5 to 10, with all but one type having at least 7 species. In addition to the tree species, Devall recorded 11 species of vines. This study reports one of the lower numbers of oak species recorded (4). Furthermore, oaks were a prominent component of only two of the community types, and even then they accounted for only 16% and 14% of the total importance value. Sites where oaks might generally be more prominent were instead dominated by species

such as *C. laevis*, *F. pennsylvanica*, *L. styraciflua*, and *Carya illinoensis* (pecan). Although the study area has been relatively undisturbed for decades, the unusually small importance of oaks may still be partly an artifact of its history, which includes both farming and selective logging. Also, the study site is located in an area along the Mississippi River that has no man-made levees. It therefore may have a high rate of sediment deposition, which is more favorable to other species than to oaks (Hodges 1995).

Few studies have reported on woody species composition during the earliest stages of old-field succession in bottomland forests of the region. There is considerable information available on successional patterns of newly formed land (i.e., point bars along rivers) and in clearcut areas (Shelford 1954; Bowling & Kellison 1983; Krinard & Johnson 1986; Johnson & Krinard 1988; Kennedy & Meadows 1993; Shankman 1993). The very different conditions (e.g., soil characteristics and opportunities for seed dispersal) on these types of sites, however, limit the utility of these studies as a basis for assessing potential species composition on old agricultural fields.

The general pattern that seems to exist on old-field sites is one of slow establishment of woody species, especially for overstory tree species. DePoe and Pritchett (1986) found only one tree species, *Diospyros virginiana* (persimmon), on a 3.2-ha plot near Monroe, Louisiana, during the first year following abandonment, and those seedlings were dead by the end of the growing season. Five years later, only six woody taxa were recorded for the same area, and only two of these (*D. virginiana* and *Fraxinus* spp.) were potential overstory species (Battaglia 1991).

Old-field succession studies in the LMRAV (Bonck & Penfound 1945; Hopkins & Wilson 1974; Bazzaz 1975) suggest that the following pattern may be typical. During the first year following abandonment, the site is dominated by annuals; from the second to tenth year, perennial herbs (e.g., *Solidago* spp.) dominate; shrubs, vines, and—to a lesser degree—tree seedlings begin to establish themselves as early as the first year and gradually increase in number and size such that they become dominant at around the tenth year; after about 25 years, the site looks like a young forest, with a mixture of large shrubs and small trees. This pattern may be somewhat altered for low sites that are frequently flooded well into the growing season, where annuals may persist as dominants for a longer period (Battaglia 1991).

The degree of flooding that occurs on an old-field site may have a substantial impact on the rate and type of woody species establishment. Hydrochory (seed dispersal by water) has been shown to be an important mechanism for species such as *T. distichum* and *Nyssa aquatica* (Schneider & Sharitz 1986), although it is worth noting that most seeds were trapped by features such as

Table 1. Mean number and height of planted and invader trees in the 10 stands evaluated in west central Mississippi by Allen (1990).*

Species	Stand Number									
	1	2	3	4	5	6	7	8	9	10
Mean Number per Hectare										
Planted Oaks	741	657	613	781	494	497	2265	395	252	215
Invader Species										
<i>Liquidambar styraciflua</i>	14329	49	3	7	7	141	32	12	3	5
<i>Fraxinus pennsylvanica</i>	247	12	64	17	17	99	15	3	235	0
<i>Ulmus</i> spp.	141	37	5	126	54	5	0	67	10	49
<i>Platanus occidentalis</i>	413	0	3	20	0	0	0	0	0	0
<i>Diospyros virginiana</i>	7	42	158	15	12	25	20	17	20	59
<i>Acer negundo</i>	12	3	0	37	0	0	0	7	185	72
<i>Celtis laevigata</i>	106	7	3	5	32	0	3	0	12	0
<i>Gleditsia</i> spp.	17	5	0	7	27	7	42	0	0	0
<i>Quercus nigra</i> / <i>Quercus phellos</i>	37	0	0	0	0	3	0	3	0	0
<i>Acer rubrum</i>	0	0	7	5	0	0	0	3	3	5
<i>Carya aquatica</i>	0	0	0	0	0	5	10	0	0	0
<i>Quercus lyrata</i>	0	0	0	0	0	0	12	0	0	0
Other trees	7	5	0	10	0	302	25	0	0	5
Shrubs	3	0	0	27	3	72	111	3	7	5
Total Invaders	15319	160	243	276	152	659	270	115	475	200
Average Height (m)										
Planted Trees	5.84	2.95	2.81	4.76	2.64	1.39	2.10	1.24	1.02	0.54
s.e.	0.11	0.05	0.06	0.07	0.07	0.02	0.02	0.04	0.07	0.05
Invaders	4.03	2.25	1.76	2.60	1.55	1.47	1.41	1.18	1.97	0.51
s.e.	0.02	0.12	0.08	0.13	0.09	0.03	0.04	0.06	0.07	0.04

*Stands 1–5 were established with seedlings, stands 6–10 by direct-seeding.

logs and cypress knees, which are generally absent on recently abandoned agricultural fields.

Woody Species Diversity on Reforested Sites

Few studies of any type have been conducted on bottomland hardwood reforestation projects on old agricultural fields. Possibly the only published study to address woody species invasion into old-field sites reforested with oaks is that of Allen (1990).

In that paper, I assessed (1) the stocking and growth of trees planted or direct-seeded on 10 sites in west-central Mississippi and (2) the degree of woody species invasion. Five of the sites (stands) were established by planting 1-year-old bareroot seedlings, and the other five were established by direct seeding, which was done with a modified soybean planter that sows seed in furrows. In all cases the planted seed or seedlings were one to three of the most commonly planted oak species in the region (*Q. nutallii*, *Q. nigra*, *Q. phellos*, and/or *Q. pagoda*). All 10 stands were between 4 and 8 years old. I sampled the stands by using from one to three transects, along which I established 0.02-ha circular plots every 30 m; between 5% and 22% of the area of each stand was sampled. A more detailed description of the sites, species planted, and methods can be found in Allen (1990).

At first it may appear that both the number and the diversity of invaders were reasonably high (Table 1), with the notable exception of heavy-seeded species of genera such as *Quercus* and *Carya*. Further analysis of the data, however, raises some concern about the future composition of most of the stands.

One source of concern is the average height of the invading species compared to that of the planted species. For the stands established with seedlings (stands 1–5), the planted species were on average 1.36 m, or 58% taller than the invading trees (Table 1). Because the stocking of planted trees was consistently high on the five sites established with seedlings (657 seedlings/ha), it seems reasonable to conclude that growing space is limited and that many of the invading trees will therefore be overtopped and shaded out by the planted trees, especially seedlings of species relatively intolerant of shade.

The results were considerably different for the five stands established by direct seeding (stands 6–10), in which the stocking was much less consistent and the average height of the oaks much shorter (Table 1). For these five stands, the invading species were on average 0.05 m taller than the planted trees, and the prospects for a diverse stand after crown closure appear better.

An even more critical concern is the differential invasion that occurs across large fields. The stands in Table 1 with high amounts of invasion were either long, nar-

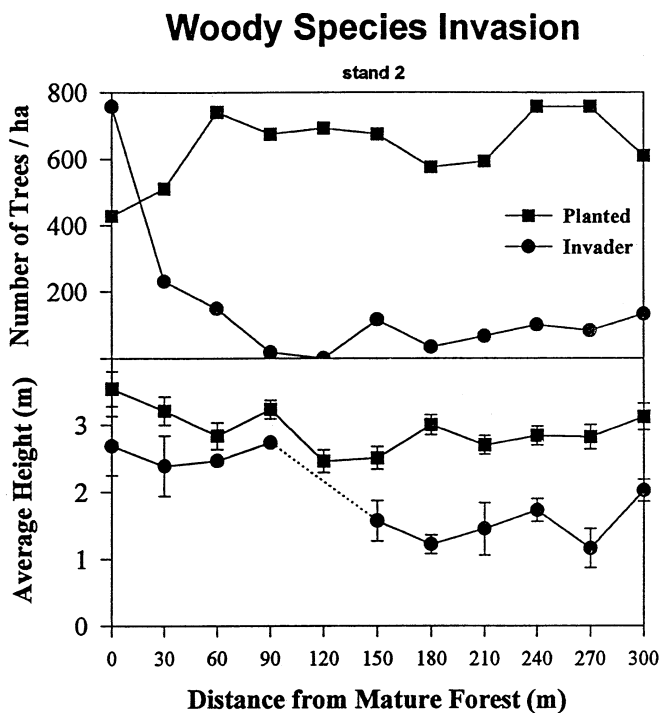


Figure 4. Average numbers (trees/ha) and heights (m) of planted and invader trees in relation to distance from a mature forest. Data for stand 2 in Allen (1990).

row stands bordering on mature forest (stand 1) or stands closer to square in shape that are bordered on at least two sides by mature forest (stands 4, 6, and 9). The other stands were more isolated from mature forest, and the average number of invader trees was lower and the distribution considerably more uneven.

In Fig. 4, the average number and height of both planted and invading trees in stand 2 are depicted in relation to distance from a stand of mature forest, which borders the southern end of the reforestation site. It is apparent that most of the invasion is limited to the first 60 m from the forest edge. Also, the average height of the invading trees is considerably shorter beyond the 60-m point, dropping from an average of 2.60 to 1.56 m. This suggests that trees may have taken longer to invade the more distant parts of the stand and that the planted trees have a more significant head start in growth.

Although I am aware of no other published studies on woody species invasion on sites reforested with oaks, this issue is becoming an increasing concern among researchers and practitioners of reforestation in the region. For example, little invasion of woody species has been observed in the reforested stands established on the Ouachita Wildlife Management Area near Monroe, Louisiana, even in those that flood frequently and that were planted as long ago as 1985 (L. Savage, Louisiana Department of Wildlife and Fisheries, personal communication).

It is clear, however, that natural invasion will occur on reforested sites if conditions are suitable. Evidence for this was presented recently by Shear et al. (1996), who compared woody species composition on old-field sites in western Kentucky planted between 1942 and 1945 with nearby naturally regenerated and mature forests. While they found that invading light-seeded species were prominent in the planted stands, their sites were somewhat atypical in that they were small (<2 ha) and were planted either with *Taxodium distichum* or *Chamaecyparis thyoides* (Atlantic white cedar), the latter of which had 100% mortality.

Discussion

Implications for Reforestation Projects. Reforestation projects implemented to date have been generally effective in establishing target tree species (Allen 1990; Sharitz 1992). It is becoming increasingly apparent, however, that, at least for the first 40 to 50 years after planting, many of these reforested stands will have the appearance of plantations and will be near-monocultures of oaks. This seems to be most likely on sites that have good survival, are not thinned, and are located more than approximately 60 m from mature forest.

Oaks are clearly a major component of most bottomland hardwood forests throughout the region. Also, they generally invade old-field sites at a slower rate than many other species (Table 1; Shear et al. 1996). The emphasis of current restoration projects on planting oaks therefore appears to be justified to a large degree. This is not to say, though, that other species should not be planted, that the composition of the oak species currently planted should not be changed, or that other concerns about current oak planting practices should not be raised.

At least two courses of action are possible. First, current practices could be continued and land managers could operate from the (probably safe) assumption that, as the original planted stand begins to break up, other species will begin to occupy a significant proportion of the site. Alternatively several relatively small modifications could be made to current practices to establish a forest with a more natural appearance and greater species diversity.

To make the reforested site look more natural, the most obvious modification would be to discontinue the practice of planting trees in rows (Fig. 5b). Planting in rows is easier with mechanical planters and also serves to make monitoring or post-planting weed control easier to carry out. In practice, however, weed control is seldom done, and adequate techniques for monitoring randomly distributed seedlings are available. When possible, driving mechanical planters in an irregular pattern would be desirable. Also, the use of broadcast

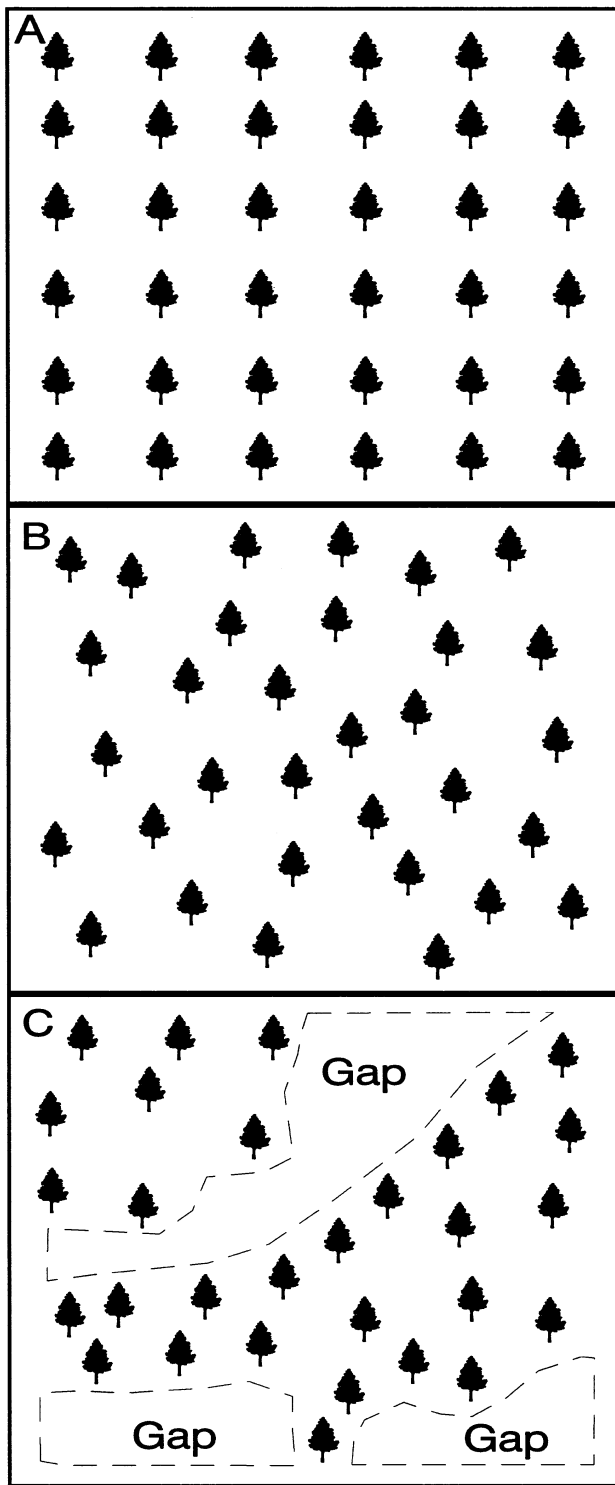


Figure 5. The typical pattern of tree planting in the southcentral United States (a), an alternative method using random spacing (b), and an alternative method using a system of group plantings and gaps (c).

seeders rather than the commonly used row planters should be encouraged for direct seeding, especially when light-seeded species are sown. Some of these practices are beginning to be applied by agencies such as the Louisiana Department of Wildlife and Fisheries (K. Ribbeck, personal communication) and the U.S. Fish and Wildlife Service (Haynes et al. 1995). I am aware of no demonstrated biological justification for planting more randomly, but there is no reason why aesthetics should not be considered to be an important part of restoration, especially on public lands and in cases where it does not add significantly to the cost of the project.

Some planting practices are likely to encourage the establishment of other woody species while at the same time saving money. Simply planting fewer trees may be one of the most effective practices. By planting fewer trees on every acre, crown closure will be achieved more slowly and more growing space will be available for natural invasion. Planting seedlings at a spacing of roughly 3×3 m requires 1111 trees per hectare; increasing the space to 5×5 m requires only 400 trees per hectare. If only a third of the 400 seedlings planted per ha survive to maturity, the stocking of oaks will exceed that found in the overstories of most of the natural stands reviewed in this paper.

A practice that may prove effective is to continue to plant oaks at the usual spacing but leave numerous unplanted gaps (Fig. 5c). Planting the oaks close together should encourage height growth and hasten crown closure, ensuring that an adequate number survive to maturity, while gaps allow room for other species, especially shade-intolerant ones such as *Liquidambar styraciflua* and *Gleditsia* spp. (Putnam et al. 1960), to become established.

Direct seeding is considerably less expensive than planting seedlings (Bullard et al. 1991), and I believe it is also more likely to result in a diverse forest. Direct seeding has been shown to result in slower initial growth than planting seedlings (Allen 1990; Wittwer 1991), which lengthens the time to crown closure and should therefore afford invaders a greater opportunity to become established. Direct seeding also tends to result in more-variable stocking rates than planting seedlings (Allen 1990), which may actually prove to be an asset for producing a more diverse forest.

Heavy-seeded species not typically planted, such as *Carya aquatica* on wet sites and *C. cordiformis* (bitternut hickory) and *C. ovata* (shagbark hickory) on higher sites, should be given more consideration in future projects. Oak species other than those currently emphasized should be planted, such as *Quercus lyrata* on the lower sites, *Q. michauxii* (swamp chestnut oak) on higher sites, and *Q. laurifolia* or *Q. obtusa* (diamondleaf oak) on sites immediately to the east and west of the LMRAV. Based on the composition of the natural forests found in the

studies reviewed here, I suggest that land managers attempt to develop stands in which oaks comprise 30–50% of the overstory trees.

Natural invasion should not be relied upon entirely for other species, even those dispersed primarily by wind. Examples of species that should be planted more often (when appropriate for the site) are *Diospyros virginiana*, *Platanus occidentalis* (American sycamore), and *Ulmus* spp., especially those other than *U. americana*. Midstory and shrub species should be planted where they form significant components of the natural forest. Also, a formerly important component of bottomland forests now almost completely overlooked in reforestation projects are canebrakes, dominated by *Arundinaria gigantea* (river cane or switchcane).

Future Research Needs. Researchers interested in supporting bottomland hardwood restoration efforts are faced with a familiar dilemma: information needs to be provided to land managers quickly, yet long-term studies are needed to provide much of the information needed. In this case, the risk of providing information too slowly is that researchers will be left behind and that another 75,000 ha or more of forest may be established over the next decade in a suboptimal fashion.

Clearly, more studies on the establishment of invading woody species in reforested stands are needed. There are some data available and a growing amount of anecdotal evidence, but more information is needed before land managers can feel justified in altering current reforestation practices to promote additional species.

The development of chronosequences for the study of secondary succession would be valuable. Because many farmers have gone bankrupt or reduced the number of acres they farm, it may be possible to locate more fallow fields than was possible in the past—although, with some notable exceptions (Shear et al. 1996), most will be 10 years old or less. Better information on the rate of natural invasion and the species most effective in dispersing naturally is critical for determining which species to include in reforestation projects. In particular, there seems to be little information on secondary succession in the central part of the lower Mississippi Valley, where much of the reforestation is taking place.

While new, long-term studies designed to evaluate techniques such as those suggested in the previous section are essential, we also need to take full advantage of existing reforestation projects, many of which are approaching or have reached the crown closure stage. We need to accumulate case studies of these older stands and seek to relate the woody species diversity found to (1) method of reforestation (site preparation and seedlings versus direct seeding), (2) distance and direction from mature forest, (3) flooding regime, (4) soil type, and (5) post-planting management practices (e.g., thinning).

Experimental manipulations of some of these older stands, such as thinning or enrichment plantings of additional woody species, could provide valuable guidance for management of existing reforestation sites.

I have primarily been concerned with site-specific aspects of reforestation. We also clearly need, however, to examine the larger-scale aspects of reforestation as currently practiced in the region. We need, for example, to begin addressing the impacts of large-scale reforestation on the genetic composition of tree populations. Information on the existence of substantial, localized, intraspecific genetic variation could have important implications for the movement of planting stock. The effects of fragmentation on gene flow should also be considered.

Finally, we need to address landscape-scale questions related to reforestation. The potential may exist for researchers to provide considerable guidance to land managers on the location of reforestation projects. Properly locating reforestation sites may help reverse past patterns of forest fragmentation, provide effective corridors for the movement of wildlife, and help maintain regional biodiversity (Dunn et al. 1993). Some promising efforts have been made in this direction, primarily for the Tensas basin in northeastern Louisiana (Gosselink & Lee 1989; Gosselink et al. 1990; Craig et al. 1992). Every effort should be made to expand such work to the entire region.

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