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Vegetation responses to helicopter and ground based logging in blackwater floodplain forests

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Abstract

Logging in floodplains of low order, blackwater streams may damage existing seedlings and rootstocks, and create soil conditions that inhibit establishment and growth of regeneration after harvest. Removal of logs via helicopters has been advocated to minimize soil damage and facilitate rapid revegetation. We tested impacts of helicopter versus conventional skidder harvest systems on regeneration, woody plant community structure and biomass growth in three blackwater stream floodplains in southern Alabama. The helicopter treatment resulted in significantly greater woody plant density (19,900 versus 14,300 stems/ha by Year 8), but both treatments were well-stocked with commercially valuable species. By Year 8, treatment effects on density of individual species were generally not significant; however, density of *Cliftonia monophylla* was lower on skidder plots (p=0.001) and density of *Nyssa sylvatica* var. *biflora* was lower on helicopter plots (p=0.092). In both treatments, species richness within 0.004 ha regeneration plots declined slightly between pre- and post harvest, but the Shannon diversity and evenness indices remained essentially unchanged through 8 years after treatment. Post-harvest survival of *Acer rubrum*, *Cyrilla racemiflora* and *C. monophylla* rootstocks was significantly lower on the skidder plots. In both treatments, species dominant before harvest remained so afterwards. Species with the tallest sprouts in Year 8 were *Liriodendron tulipifera*, *Magnolia virginiana*, and *A. rubrum*. During the first 2 years after logging, aboveground biomass was greater in the helicopter treatment, but the difference was only significant in Year 1. We conclude that both harvesting methods had little effect on species composition. Skidding may result in a stand structure more favorable for commercial timber production; however, impacts of skidding on long-term productivity are not yet known. © 2000 Elsevier Science B.V. All rights reserved.

Keywords: Clearcut; Coastal plain; Diversity; Plant community structure; Regeneration

1. Introduction

Blackwater, oligotrophic ecosystems comprise more than half of the forested wetland area of the southeastern USA (Tansey and Cost, 1990). They typically have organic-matter rich soils and a species-rich forest community of evergreen and deciduous angiosperms and gymnosperms (Gemborys and Hodgkins, 1971, Walbridge and Lockaby, 1994). When located within narrow stream margins and small drainages, such systems have many species common to the adjacent uplands in addition to species adapted to wetter conditions (Gemborys and Hodgkins, 1971).

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Historically, tree harvests in blackwater systems were infrequent and focused on selective removal of the more valuable species such as slash pine (*Pinus elliottii* Engelm.) and yellow-poplar (*Liriodendron tulipifera* L.). Recent increases in use of hardwoods for paper production have accelerated rates of harvest and have made clearcutting economically viable. These trends have raised concerns about the ecological impacts of tree harvest in blackwater swamps. Soils are nearly always wet and therefore vulnerable to a high degree of site damage during logging (Aust et al., 1990; Walbridge and Lockaby, 1994). Harvest may cause rutting or accelerated nutrient release and soil movement, all of which may affect the regeneration of individual species, plant succession, and ecosystem productivity (Aust and Lea, 1992). To reduce such risks, helicopter removal of trees has been advocated (Jackson and Stokes, 1991; Aust, 1994). However, helicopter logging is expensive and dangerous (Conway, 1976, Stenzel et al., 1985). Benefits afforded by helicopters must therefore be documented before this practice can be recommended.

After a forest is clearcut, some of the rootstocks that survive disturbance develop stump or root sprouts. In most angiosperm forests, these sprouts are much larger than seedlings, and end up dominating the stand. However, the sprouting response is not uniform. Some species sprout more readily than others. Furthermore, the size and vigor of sprouts may be closely related to pre-harvest stump diameter (Johnson, 1977; Loftis, 1990). Influences of species, stump diameter and harvest methods on post-harvest sprouting are poorly understood for forested wetlands in blackwater floodplain ecosystems.

In a recent study, we determined that clearcutting in three Alabama blackwater systems: (1) briefly increased biological oxygen demand in the stream but had no impacts on other water quality parameters; (2) increased litter decomposition rate 2 years after harvest; (3) decreased water table height; and (4) had no measurable impact on rates of denitrification (Lockaby et al., 1994, 1997). In this paper, we report vegetation responses. Specifically, we contrast impacts of helicopter log removal versus the more traditional approach of rubber-tired skidders on: (1) 8-year trends in woody plant density and species composition; (2) 8-year trends in survival and growth of sprouts produced by advance regeneration; and (3) 2-year trends in aboveground production.

2. Methods

2.1. Study site and harvest treatments

Three floodplains of low-order blackwater streams were chosen within the Escambia River drainage basin in southwestern Alabama. At nearby Brewton, Alabama (approximately 30 km from study sites) mean annual temperature is 18.2°C, mean monthly temperatures range between 10.6 and 25.8°C, and mean annual precipitation is 165 cm (NOAA, 1992).

Soils are classified as Dorovan–Johnston Association which consists of Dorovan Muck, dystic, thermic typic Medisaprists, and Johnston series, coarse-loamy, siliceous, acid, thermic cumulic Humaquepts. These soils are highly to extremely acidic, typically saturated, and relatively low in available phosphorus (Lockaby et al., 1994). Additional site description can be found in Lockaby et al. (1994, 1997).

In each floodplain, four 0.4 ha treatment plots were established in the fall of 1990 within a planned 4 ha clearcut harvest area. Two plots were randomly assigned to a high impact and two to a low impact harvest system. The high impact treatment consisted of fellerbuncher logging on mats, helicopter removal of logs, and then simulated skidder removal of logs; i.e., a rubber-tired skidder dragged three logs (each 40–60 cm basal diam) for one pass across the entire plot. The low impact treatment utilized handfelling with chainsaws and helicopter log extraction. Within both treatments all stems more than 2 cm diameter at 1.4 m (dbh) were cut down to assure safety during the helicopter log extraction. Logging was completed between January and March 1991.

2.2. Pre-harvest vegetation

In the fall of 1990, overstory and midstory vegetation, and advance regeneration were inventoried within each treatment plot in each floodplain. For the overstory inventory, each tree more than 11.5 cm dbh was tallied by species and diameter in a 0.10 ha circular subplot located at the center of each treatment plot. In the midstory inventory, species and dbh were recorded for trees 3.8 to 11.4 cm dbh within a 0.04 ha, circular subplot located in the center of the overstory plot.

Advance regeneration, which included all stems but was predominantly trees less than 3.8 cm dbh, was inventoried within a 0.004 ha (10×4 m) rectangular, permanent plot established near the center of each treatment plot. Number of stems was recorded for each species.

2.3. *Response of total woody community to harvest treatments*

At the end of the first and second growing seasons after harvest (1991 and 1992), and in the middle of the eighth (1998), we re-inventoried the permanent 0.004 ha advance regeneration plots to tally number of individuals of each woody species. A plot of number of species over area sampled showed that our method of sampling was adequate to capture virtually all of the woody plant species in these ecosystems. However, because our small sample size (only twelve 0.004 ha plots) may provide inaccurate estimates of regeneration density in the entire area treated, we added two temporary 0.004 ha regeneration plots per treatment plot in the 1998 survey for a study-wide total of 36 regeneration plots. The expanded survey included four additional species of shrubs and trees (27 versus 23 species), but in both surveys, the 11 most abundant species were the same, and accounted for 92 to 93% of total stand density. Voucher specimens for most of the sampled species were deposited at the Virginia Tech herbarium, Blacksburg, Virginia.

Data from the permanent 0.004 ha regeneration plots were used to assess changes in density (number per ha). Density of each tested species and all species combined was relativized to initial density (i.e., [1991, 1992 or 1998 density−1990 density]/1990 density) and then analyzed using analysis of variance (ANOVA) with a randomized block model. There were three blocks (floodplains) and two treatments (helicopter versus skidder). Data were log transformed where needed to meet assumptions of equal variance across treatments. Our null hypothesis was that relativized density changes were the same in each treatment. We made these calculations and tests for all species combined, and for each of seven species (hereafter referred to as major species) that were either dominant in the overstory or midstory (*Magnolia virginiana*, *Nyssa sylvatica* var. *biflora*, *Acer rubrum*, *Cyrilla monophylla*, *C. racemiflora*, and *Ilex opaca*), or are especially valuable for forest products (*L. tulipifera*).

Species composition changes were assessed in three ways. First, using permanent plot data we tallied species richness, calculated Shannon diversity and evenness indices, and tested for treatment impacts on these values via ANOVA (Magurran, 1988). Second, the permanent plot data were used to calculate Bray–Curtis distance coefficients, D (Legendre and Legendre, 1998) between each helicopter plot and each of the skidder plots as follows:

$$D = 1 - \left(\frac{2W}{A+B}\right)$$

where W is the sum of the shared abundances (i.e., the minimum densities) for each species in the two plots being compared, A is the total abundance of all species in one of the plots, and B is the total for the other plot. Mean distances between plots were calculated for the pre-harvest survey (1990) and each of the re-inventories. Our null hypothesis is that helicopter and skidder harvests bring about the same changes in community composition, and thus, D values comparing helicopter versus skidder plots should be no greater after harvest than before. We tested this null hypothesis using a Wilcoxon two-sample test (Sokal and Rohlf, 1981). Finally, we used correspondence analysis (Legendre and Legendre, 1998) to compare community structure of plots in the skidder and helicopter areas before (1990) and 8-year after (1998) treatment. This analysis was conducted using data from both permanent and temporary plots, and the software program PC-ORD version 4.0 (McCune and Mefford, 1999) with no down-weighting of rare species.

2.4. *Response of large rootstocks to harvest treatments*

Individual stems of intermediate to large size (≥3.8 cm dbh) were selected within the central 0.10 ha of each treatment area prior to harvest. For each of the seven major species, 60 rootstocks were selected using a stratified random approach to insure a range in size classes and equal representation in the two harvest treatments. To facilitate relocation after harvest, trees were tagged and painted; and distance and azimuth from plot center were recorded. Stump diameter was measured to relate rootstock size to post-harvest responses.

At the end of the second growing season and in the middle of the eighth, each tagged tree was assessed for survival, the total number of sprouts, and the height and basal diameter of the dominant sprout. Effects of harvest treatment on survival were assessed using Fisher's exact test for all rootstocks combined, and within species. The influence of pre-harvest stump diameter on survival was assessed using logistic regression (Procedure CATMOD of the Statistical Analysis System; SAS Institute Inc., 1988). Finally, influences of harvest treatment and species on sprout height and number of sprouts per stump were assessed by two-way ANOVA treating floodplains as blocks.

2.5. Biomass responses

Within each of the harvested plots, all vegetation in ten 1 m^2 clip plots was harvested in late summer of 1991 and 1992 to assess aboveground biomass production. In both years, we collected all herbaceous plant material, all leaves from woody plants, and al l 1-year-old stem material from woody plants; older stem material on woody plants was excluded. Clip plots were 10 m apart along a linear transect located 3 m inside the outer edge of each harvest plot. In 1992, a different side of each harvest plot was used to avoid reclipping the same spot. The plant material was dried to constant weight at 60°C, and weighed. Harvest treatment impact on biomass was assessed for each year using ANOVA.

3. Results

3.1. Pre-harvest vegetation

Nyssa sylvatica var. *biflora* (see Tables 1 and 2 for taxonomic authorities) and *M. virginiana* clearly dominated the pre-harvest overstory, cumulatively comprising 69% of the total stem density and 77% of the basal area (Table 1). The midstory was dominated by *C. monophylla*, *I. opaca* and *A. rubrum* which together accounted for 60% of the density. Species harvested locally for pulp and timber products constituted 86% of the overstory stems, but only 36% of the density in the midstory.

Table 1
Mean community structure for three blackwater stream floodplain forests in southern Alabama

Species	Overstory		Midstory	
	Density (no./ha)	Basal area (m^2/ha)	Density (no./ha)	Basal area (m^2/ha)
Nyssa sylvatica var. *biflora* (Walt.) Sarg.[a]	191.9	4.42	22.7	0.17
Magnolia virginiana L.[a]	123.6	9.66	53.5	0.25
Acer rubrum L.[a]	37.9	1.47	121.5	0.58
Liriodendron tulipifera L.[a]	21.4	1.58	14.4	0.09
Ilex opaca Ait.	46.1	1.21	123.6	0.53
Pinus elliottii Engelm.[a]	4.1	0.97		
Quercus laurifolia Michx.[a]	15.6	1.48	6.2	0.02
Cliftonia monophylla (Lam.) Bitton ex Sarg.	7.4	0.15	131.8	0.41
Cyrilla racemiflora L.	4.9	0.13	76.2	0.34
Quercus nigra L.[a]	0.8	0.14	6.2	0.03
Persea borbonia (L.) Spreng.	4.1	0.11	12.4	0.03
Ilex coriacea (Pursh) Chapm.			51.5	0.10
Halesia diptera Ellis			4.1	0.01
Myrica cerifera L.			2.1	0.00
Osmanthus americanus (L.) Benth. & Hook F. ex Gray			4.1	0.01
Total	457.8	31.32	630.3	2.57

[a] Species harvested for forest products.

Table 2
Density (no./ha) of woody vegetation in permanently marked plots prior to harvest (1990) and 8 years after harvest (1998) of three Alabama floodplain forests[a]

Species	Pre-harvest (1990)		Post-harvest (1998)	
	Helicopter	Skidder	Helicopter	Skidder
Viburnum nudum L.	6542	38958	458	1500
Ilex coriacea (Pursh) Chapm.	9542	24458	1917	1500
Cyrilla racemiflora L.	12500	14333	3417	2333
Acer rubrum L.[b]	14708	11250	1792	583
Clethra alnifolia L.	9000	6917	2250	1417
Itea virginica L.	9750	5792	2167	1708
Cliftonia monophylla (Lam.) Bitton ex Sarg.	4792	2375	4458	958
Magnolia virginiana L.[b]	3375	1542	875	792
Ilex opaca Ait.	2792	1875	542	125
Aronia arbutifolia (L.) Ellis	583	2167	0	0
Nyssa sylvatica var. *biflora* (Walt.) Sarg.[b]	1500	958	458	1500
Liriodendron tulipifera L.[b]	1042	917	375	1583
Ilex glabra (L.) Gray	1167	42	292	42
Persea borbonia (L.) Spreng.	542	667	125	83
Quercus nigra L.[b]	875	250	250	0
Styrax americanus Lam.	417	208	208	42
Quercus laurifolia Michx.[b]	83	458	42	42
Myrica heterophylla Raf.	417	42	167	83
Vaccinium elliottii Chapman.	333	0	125	0
Pinus elliottii Engelm.[b]	333	0	0	0
Myrica cerifera L.	0	208	0	0
Cephalanthus occidentalis L.	0	42	0	0
Ilex cassine L.	42	0	0	0
Total	80335	113459	19918	14291

[a] Species listed in order of most to least abundant in 1990.
[b] Species harvested for forest products.

Prior to harvest, shrubs and trees in the 0.004 ha regeneration plots were very numerous in both the helicopter (80,300 stems/ha) and skidder (113,459 stems/ha) treatment areas. Of the 23 species encountered, the most abundant were *Viburnum nudum*, *I. coriacea*, *C. racemiflora* and *A. rubrum* (1990 survey, Table 2). With the exception of *A. rubrum*, these species are shrubs or small trees. All are shade tolerant which explains their success in the pre-harvest understory. Averaged over all regeneration plots, merchantable species comprised only 19% of the stems.

3.2. Response of total woody plant community

In the first year after harvest (1991), total stem density in the regeneration plots declined in both treatments (Fig. 1). Reductions were greater in the skidder treatment resulting in a significantly lower density relative to pre-harvest levels (Fig. 1). Total stem density continued to decline through 1998 in both treatments; however, differences in relativized abundance were not significant in 1992 and 1998 (Fig. 1). By the eighth growing season after harvest, density was 25% of pre-harvest level in the helicopter treatment, and 13% in the skidder treatment.

Individual species responded differently to the treatments. *Acer rubrum* followed the same trends as in total stems, declining throughout the study with significantly fewer stems in the skidder versus helicopter plots in 1991 only (Fig. 1). One other species, *I. opaca* was less abundant during all post-harvest inventories; however, in this species, density was temporarily greater in the skidder than in the helicopter plots (Fig. 1). In three species, *C. racemiflora*, *M. virginiana* and *C. monophylla*, the helicopter treatment appeared to stimulate density, but differences between treat-

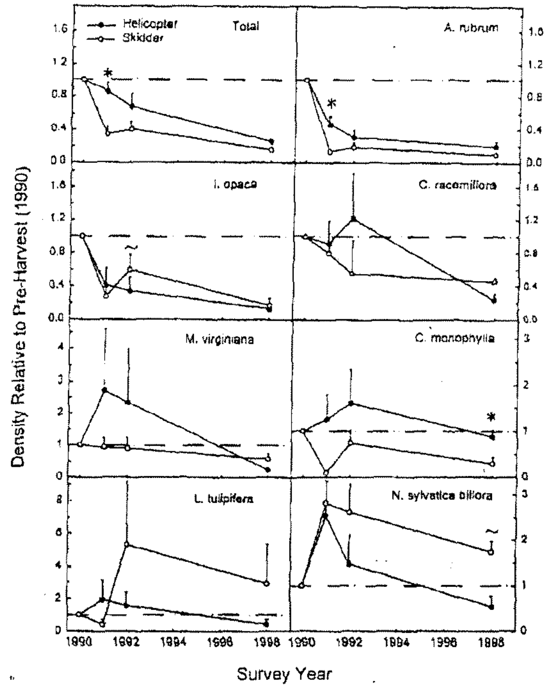

Fig. 1. Woody plant density response to handfelling and helicopter log removal versus fellerbuncher and simulated skidder log removal in three floodplains of low order, blackwater streams, Alabama. Bars are standard errors. *=treatment difference ($p<0.05$); ~=treatment difference ($p<0.10$).

ments were only significant for *C. monophylla* at the final survey. In *L. tulipifera*, density was quite variable, and no significant differences between treatments were noted. However, in skidder plots, *L. tulipifera* were five times more dense in 1992 than during pre-harvest (Fig. 1). *Nyssa sylvatica* var. *biflora* density was stimulated by both treatments, but more so in skidder plots.

Species preferred for commercial use, as a proportion of stand density, were favored in the high impact harvesting treatment, but not in the low impact treatment. In 1990, 25.7% of the stems in the helicopter plots belonged to commercially preferred species. This proportion dropped to 20.7%, 16.4% and then 17.5% in 1991, 1992, and 1998, respectively. In skidder plots, proportions of stems belonging to preferred species increased throughout the study from 13.5% in 1990 to 15.4% in 1991, 22.1% in 1992 and 31.3% in 1998.

After harvest, species richness tended to decline in both helicopter and skidder plots (Table 3). However, because evenness increased after harvest, so did the Shannon index of diversity. The Shannon diversity and evenness indices tended to be greater in helicopter plots, both before and after harvest, but differences were only significant in 1992 (evenness) and 1998 (Shannon index).

There was no apparent divergence in species composition caused by treatment. Mean Bray–Curtis distance between helicopter and skidder plots was 0.44 before harvest. This value increased to 0.49 in 1991, but then decreased back to 0.44 in 1992 and then to 0.43 by 1998. Differences between dates, including pre- and all post-harvest surveys, were not statistically significant (Wilcoxon test, $p=0.52$). Correspondence analysis showed a similar lack of divergence. Before harvest, plots in the helicopter and skidder plots appeared well-distributed along the first ordination axis, and somewhat less so along the second axis (Fig. 2, 1990). After harvest, plots in the two treatment

Table 3
Diversity indices prior to harvest (1990) and up to 8 years after helicopter and skidder logging (1998) for the woody plant community in blackwater river floodplains[a]

Survey date	Species richness		Shannon diversity		Shannon evenness	
	Helicopter	Skidder	Helicopter	Skidder	Helicopter	Skidder
1990	4.7a (1.1)	14.3a (0.8)	1.809a (0.101)	1.574a (0.151)	0.679a (0.036)	0.593a (0.055)
1991	2.5a (1.6)	11.0a (1.2)	1.861a (0.129)	1.559a (0.148)	0.752a (0.023)	0.655a (0.038)
1992	2.5a (1.6)	13.2a (1.0)	1.970a (0.107)	1.836a (0.070)	0.800a (0.023)	0.716b (0.023)
1998	11.5a (1.1)	9.7a (0.7)	1.971a (0.092)	1.643b (0.178)	0.819a (0.033)	0.720a (0.061)

[a] $n=3$ forests, standard errors are in parentheses. Within years and diversity indices, means with different small letters are significantly different ($p<0.05$).

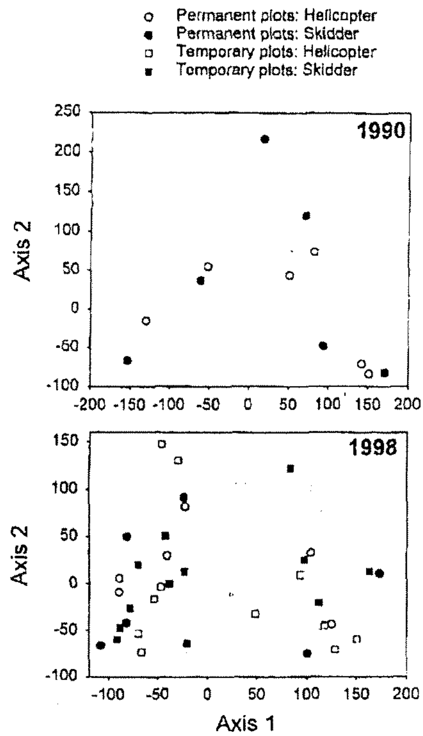

Fig. 2. Canonical correspondence analysis ordinations of vegetation in 0.004 ha sample plots before (1990) and eight years after (1998) helicopter and skidder harvest treatments in three floodplains of low order, blackwater streams, Alabama. R^2 for correlations of distance measures in the ordination (Euclidean distance) with distances measured in the original data sets (relative Euclidean distances) for axes 1 and 2 combined were 0.67 and 0.54, respectively, which implies that more than 50% of the variability in the original data sets was captured by the first two axes (McCune and Mefford, 1999).

types were well distributed across axes one and two (Fig. 2; 1998).

3.3. Response of tagged rootstocks

For all rootstocks combined, survival was significantly greater ($p<0.001$) in the helicopter treatment in both 1992 (86.3% versus 60.3%) and 1998 (70.1% versus 47.4%). However, substantial differences among species occurred. *Magnolia virginiana* was virtually unaffected by treatment (Fig. 3). No significant treatment impacts were noted for this species, or for *N. sylvatica*, var. *biflora*, *I. opaca* and *L. tulipifera* (Fig. 3). On the other hand, survival was significantly lower in skidder plots for *A. rubrum*, *C. racemiflora* and *C. monophylla* (Fig. 3). Poorest overall survival occurred in *N. sylvatica* var. *biflora* (Fig. 3).

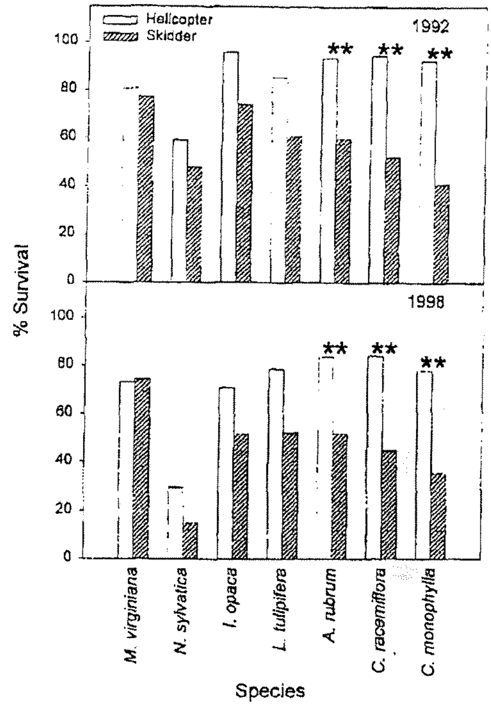

Fig. 3. Survival of rootstocks after helicopter and simulated skidder harvest systems in three floodplains of low order, blackwater streams, Alabama. **=treatment difference ($p<0.05$).

Larger stump sizes led to significantly poorer survival ($p<0.05$) in *A. rubrum* and *N. sylvatica* var. *biflora* in 1992 and for these two species plus *C. monophylla* in 1998. Stump size had no relationship with sprout survival in the remaining four species.

Mean number of sprouts per surviving stump declined between 1992 and 1998 in all species (Table 4). By 1998, *L. tulipifera* had the least number of sprouts per stump and *M. virginiana* the greatest. Larger stumps had more sprouts in 1998 in three species: *I. opaca* ($r=0.65$, $p<0.001$), *L. tulipifera* ($r=0.47$, $p=0.023$), and *M. virginiana* ($r=0.35$, $p=0.024$).

Mean height of the tallest sprout for each rootstock was significantly influenced by species ($p<0.001$) and logging treatment ($p<0.001$) in both 1992 and 1998. The interaction between species and treatment was

Table 4
Stump diameter prior to harvest and post-harvest sprout responses in 1992 and 1998 for randomly selected rootstocks in blackwater floodplain forests subjected to clearcutting[a]

Species	Range in stump diameters before harvest (cm)	Mean number of sprouts per stump		Percentage of rootstocks with dbh>3.8 cm in 1998
		1992	1998	
A. rubrum	6.4–92.0	11.9 (1.1)	8.2 (5.2)	56.4
C. monophylla	5.3–29.2	17.9 (3.1)	8.3 (0.8)	26.3
C. racemiflora	6.1–29.5	11.5 (1.2)	6.7 (0.7)	0.0
I. opaca	5.1–53.8	13.2 (1.0)	7.3 (1.0)	0.0
M. virginiana	7.6–127.0	11.5 (1.1)	11.0 (1.3)	85.7
N. sylvatica biflora	7.6–152.4	8.1 (3.4)	5.9 (1.2)	41.7
L. tulipifera	8.9–122.4	9.5 (1.4)	3.8 (0.4)	95.7

[a] n=37–59 per species; numbers in parentheses are standards errors.

nearly significant in 1992 (p=0.077), and not significant in 1998 (p=0.325). By 1998, the tallest species was *L. tulipifera* (Fig. 4). *Ilex opaca* and *C. racemiflora* were shortest. Mean heights were greater in the helicopter plots in 1992 (4.5 versus 2.7 m) and in 1998 (16.8 versus 15.4 m). One of the three floodplains had considerably shorter trees than the other two (10.1 m versus 18.0 and 19.6 m in the other two blocks in 1998); block effects were significant in both years (p<0.001).

3.4. Aboveground biomass production

At the end of the first growing season after disturbance, aboveground biomass was 4.3 times greater in the helicopter treatment (212.3 g/m^2) than in the skidder treatment (49.9 g/m^2). This difference was nearly significant (p=0.079). By the end of the next growing season, increases in biomass production in the skidder treatment narrowed the gap, with productivity in the helicopter plots only 1.3 times greater than in the skidder plots (416.3 versus 325.0 g/m^2).

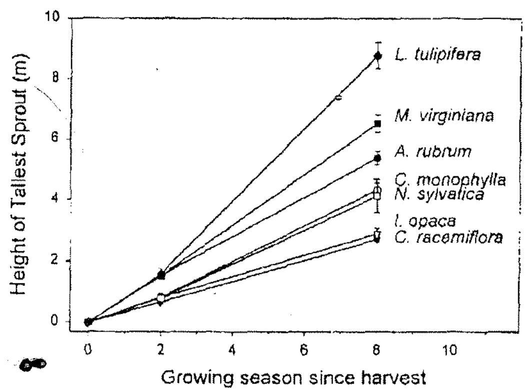

Fig. 4. Height of the tallest individual sprout from surviving rootstocks in three floodplains of low order, blackwater steams clearcut in spring 1991.

4. Discussion

Vegetation in our study system corresponds more or less to the zone II river swamp forest on sandy substrates of Christensen (1988), the sweetbay–swamp tupelo–red bay cover type of the Society of American Foresters (Larsen, 1980), and the wetter portion of the swamp forest soil moisture gradient described by Gemborys and Hodgkins (1971). It is also similar to vegetation in some non-alluvial forested wetlands such as peaty fresh water swamps (Penfound, 1952), and bayhead forests (Monk, 1966). Forests with the general physiognomy of our study area are widespread throughout the Atlantic and Gulf Coastal Plains of Southeastern United States.

The woody plant community in both helicopter and skidder treatments was relatively resistant to change when clearcut. Species that were very abundant before harvest remained so afterwards (Table 2). Species richness declined on the permanent regeneration plots (Tables 2 and 3); however, this decline was probably caused by the decline in total density (Fig. 1), which decreases the probability of encountering species on a limited number of plots of fixed size. Indeed, all species missing from permanent measurement plots in 1998 were present outside of the plots.

The resistance of our study system to disturbance is likely a consequence of the life history characteristics of the dominant species. Most are capable of regenerating by sprouting after fire (Burns and Honkala, 1990), an infrequent but important type of disturbance in floodplains of low order blackwater streams. Thus, when aboveground portions of the stand are removed by harvest, rootstocks sprout and replace the stand with one that is similar in species composition. In forest communities where the dominant species are not capable of resprouting, resistance to disturbance may be much lower (Halpern, 1988).

Helicopter and simulated skidder log removal had more or less equivalent impacts on the species composition of the woody plant community. The Bray–Curtis distance metric comparing helicopter with skidder plots was not significantly affected by harvest, and the ordination diagram showed considerable overlap between plots from helicopter and skidder areas (Fig. 2). Neither treatment eliminated any species or encouraged establishment of new species. There was a subtle difference, however, in the mode of regeneration favored by each method. The helicopter method favored sprout regeneration while the skidder treatment favored regeneration by seedlings. Two lines of evidence support this observation. First, rootstocks had poorer post-harvest survival in skidder plots (Fig. 3). Second, in a random survey of 405 stems immediately after harvest, seedling origin stems were proportionally more abundant on skidder plots (G test; p=0.027; Stokes, 1995).

Both harvest methods will produce stands with future economic value. Stem density for commercially preferred species (≥3480/ha in both harvest treatments) is well in excess of guidelines for full stocking in virtually any kind of forest (Carvel, 1988; Nyland, 1996). Although preferred species comprised a relatively small proportion of stems in both harvest treatments (<32%), they were in a very favorable crown position. Each of the three tallest species 8 years after harvest were preferred (Fig. 4). Part of this success may have been due to the felling of all stems 2.0 cm dbh and larger. Although this was done primarily for safety, such reduction of midstory and other unmerchantable stems is recommended (Meadows and Stanturf, 1997). In the absence of such cleaning, we would expect a greater dominance of shade tolerant species after harvest.

The skidder treatment may have some advantages for producing a commercially valuable stand. Skidding significantly suppressed sprouting in less desirable species (e.g., *C. racemiflora* and *C. monophylla*, Fig. 3), increased the proportion of preferred species (relative to pre-harvest conditions and to the helicopter treatment), and may have stimulated regeneration by seed in *L. tulipifera* and *N. sylvatica* var. *biflora*, two important commercial species (Fig. 1).

We were surprised that sprouting was not significantly influenced by stump diameter in several of the species tested. We expected that stump size would be inversely related to frequency of sprouting and survival of sprouts, as in other species (Solomon and Blum, 1967; Johnson, 1977; Sander et al., 1984; Loftis, 1990; Nyland, 1996). *Magnolia virginiana* was a particularly strong sprouter, producing large and vigorous sprouts on almost every stump (Fig. 4, Table 4), even those with large diameters. Furthermore, *M. virginiana* sprouting was virtually unaffected by logging treatment (Fig. 3). *Nyssa sylvatica* var. *biflora*, on the other hand, was a weak sprouter, with poor survival of rootstocks (Fig. 3), and slow growing sprouts (Table 4). Sprouting in *N. sylvatica* in other studies has been reported as vigorous (Burns and Honkala, 1990). The poor vigor we observed may be associated with the infertility of our study sites, especially the low phosphorus availability.

Biomass production was reduced on skidder plots relative to helicopter plots, at least during the first growing season after harvest. Sprout heights, which may reflect site productivity, were also lower in the skidder plots. Direct impacts of skidders on soil, and both direct (physical damage) and indirect (soil compaction) effects on rootstocks may have caused the lower production. We noted that soil compaction and subsequent surface ponding were much more prevalent in the skidder treatment, even 8 years after harvest. The long term effects of high versus low impact harvesting methods on biomass production are unknown. However, reductions in production caused by soil compaction in the skidder plots may be offset by having a greater proportion of merchantable species. Early results in this study are in keeping with those of Mader et al. (1989) who found that biomass converged in a *Taxodium–Nyssa* swamp, regardless of harvesting system. Harvesting systems using helicopters are expensive. Based on the results of this study, helicopter systems may be unnecessary in floodplains of low order blackwater streams. Fellerbunchers and skidders not only minimize harvesting costs (relative to handfelling and helicopters), they are also safer, encourage establishment of desirable regeneration, and reduce sprouting vigor of some less desirable species.

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