Aquatic Invertebrate Responses to Timber Harvest in a Bottomland Hardwood Wetland of South Carolina

Darold P. Batzer, Bagie M. George, and Amy Braccia

Abstract: We used aquatic invertebrates to assess environmental impacts of timber harvest on a bottomland hardwood wetland in the Coosawhatchie River floodplain, Jasper County, SC. Two years (1998, 1999) of preharvest baseline data were collected during winter floods in three 11–13-ha tracts of wetland forest. The following autumn of 1999 one tract was completely clearcut. In a second tract the majority of the area was also clearcut, but three 0.2–0.6-ha islands of intact forest were retained (i.e., patch-retention treatment). The third tract remained intact and served as the control. We continued to sample invertebrates in the three tracts for another 2 years (2000, 2001) after harvests. Invertebrate communities in the clearcut tract differed significantly from previous baseline conditions in that habitat and also from the nearby control tract. The patch-retention tract induced a lesser response than the clearcut, suggesting that retention islands helped mitigate impacts. Timber harvest caused a decline in some invertebrate populations (Asellidae, Crangonyctidae, Planorbidae), but an increase in others (Culicidae). Overall invertebrate abundance and family richness was not affected by harvest, only community composition. Invertebrate change probably reflected a conversion of a fauna typical of forested wetland to one typical of herbaceous wetland. FOR. SCI. 51(4):284–291.

Key Words: Bioassessment, clearcut, logging, mosquito, swamp.

BIOTA ARE USEFUL FOR ASSESSING the environmental impacts of timber harvest on wetlands (Hutchens et al. 2004). Although regenerating forests support an array of wetland animals, and postharvest faunas can be just as diverse and productive as the original communities, it appears that pre and postharvest communities of birds, amphibians, and reptiles differ functionally. Removal of shade trees from wetlands opens the forest floor to sunlight, which stimulates the growth of herbaceous plants and algae, causing the formerly forested wetlands to take on the floristic characteristics of marshes (Perison et al. 1997, Gale et al. 1998) or wet meadows (Mitchell et al. 1995, Roy et al. 2000). Habitats in which food webs had been energetically based on leaf litter and woody debris become habitats with food webs based on herbaceous plants and algae. After harvest, a vertebrate fauna of forested-associated salamanders, arboreal reptiles, and interior forest birds is replaced by a marsh or meadow fauna dominated by frogs, ground-dwelling reptiles, and edge and meadow nesting birds (Clawson et al. 1997, Hurst and Bourland 1996, Moorman and Guynn 2001, Phelps and Lancia 1995, Perison et al. 1997, Harrison and Kilgo 2004). Because this vertebrate fauna is largely predaceous, mostly on invertebrates, their response may in part mirror a change in their invertebrate food. Invertebrates are the primary trophic link between plant primary production and higher animals in wetlands (Batzer and Wissinger 1996). However, despite this important ecological role, the response of aquatic invertebrates to the harvest of wetland forests has received scant attention (Hutchens et al. 2004).

This study was designed to assess the impacts of timber harvest on aquatic invertebrates in a bottomland hardwood wetland of South Carolina. We focused on identifying functional and ecological changes in the invertebrate community, rather than generating summary metrics (diversity or community-ratio metrics) traditionally used in wetland invertebrate bioassessment (Rader et al. 2001). We hypothesized that, after harvest, the aquatic invertebrate community would shift from a fauna typical of forested wetlands to one reflective of herbaceous wetlands.

Methods

Study Site

We tested our hypothesis in forested floodplain habitats of the Coosawhatchie River, Jasper County, SC (32°33′N, 80°54′W). The Coosawhatchie is a fourth-order blackwater river draining 1,000 km^2 of the South Carolina coastal plain. At the study site near the terminal delta, the floodplain was 1.6 km wide. Soils (Brookman series) had thick, black, loamy surface layers over dark gray, clayey subsoils (Burke and Eisenbies 2000). The floodplain typically floods beginning in winter and remains at least partially inundated into late spring. The initial study year was a wet El Nino year, and the site flooded extensively from Dec. 1997 through Apr. 1998. For simplicity, we refer to that event as the 1998

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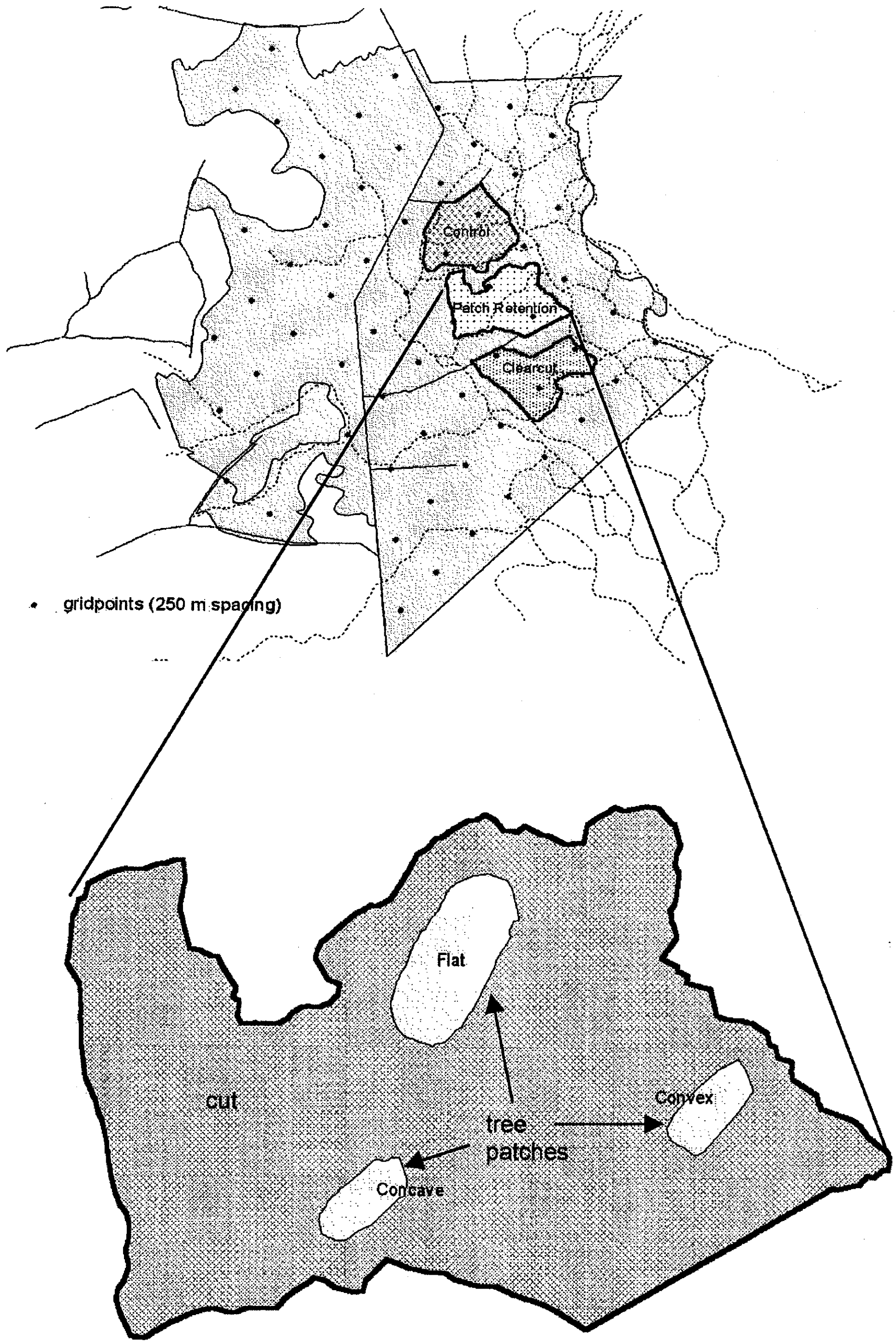

Figure 1. Map of the Coosawhatchie bottomland hardwood forest study site (Burke and Eisenbies 2000), showing the three treatment areas (uncut control, patch-retention clearcut, and completely clearcut). The enlarged inset shows the uncut forest patches that were left in the patch-retention area. Courtesy of Andy Harrison, USDA Forest Service, Center for Forested Wetland Research, Charleston, SC.

flood. The subsequent three winters were much drier and the wetland did not flood until Jan. or Feb. and was dry again by Mar. or Apr. Flooding those 3 years was mostly restricted to low-lying sloughs and channels. Vegetatively, the Coosawhatchie floodplain is classified as a bottomland hardwood forest (Sharitz and Mitsch 1993), and the major trees at the site include sweetgum (*Liquidambar styraciflua* L.), red maple (*Acer rubrum* L.), swamp tupelo (*Nyssa sylvatica* var. *biflora* [Walt.] Sargent), water tupelo (*Nyssa aquatica* L.), cypress (*Taxodium distichum* [L.] Rich.), and various oaks

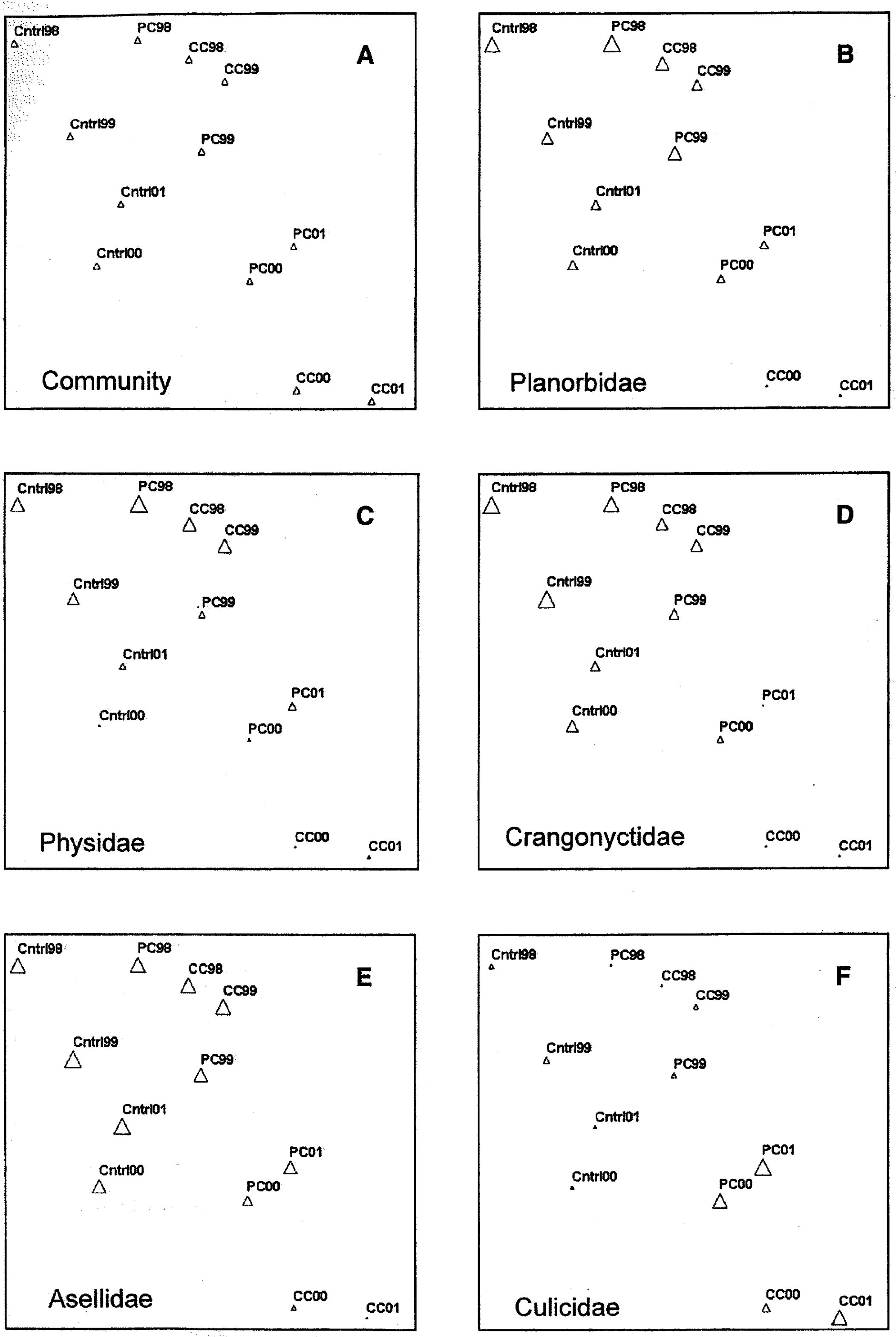

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Figure 2. Nonmetric multidimensional scaling (NMS) ordination of aquatic invertebrate communities in three wetland tracts of the Coosawhatchie River floodplain from 1998 to 2001. The three tracts included a nonharvested control treatment (Cntrl), a clearcut treatment (CC), and a patch-retention clearcut treatment (PC). In 1998 and 1999, all three tracts contained intact stands of trees. Harvests of CC and PC tracts occurred in autumn 1999, and invertebrate communities existing in 2000 and 2001 were living in modified habitat. (A) CC treatments in 2000 and 2001 (CC00, 01, lower right corner of ordination) and to a lesser extent PC treatments in 2000 and 2001 (PC00, 01) deviated from baseline conditions (CC98, 99; PC98, 99) and the Cntrl treatment (Cntrl 98, 99, 00, 01). The first two axes of the ordination accounted for 90.7% of variation (axis 1, 48.6%; axis 2, 42.1%), and the overall analyses had very low stress (7.5). The NMS ordination was based on Euclidean distance of $\log_{10}(x + 1)$-transformed abundance data. (B-F) These five graphs show the same ordination but indicate relative abundances for specific invertebrate families in each collection (triangle sizes are proportional to log-transformed relative abundances). The families shown each differed significantly ($P < 0.05$; Wilcoxon t-test) in abundance between harvested (CC00, 01; PC00, 01) and nonharvested (Cntrl98, 99, 00, 01; CC98, 99; PC98, 99) wetland, and were the organisms that contributed most to the overall ordination pattern.

(*Quercus* sp.). For additional descriptive data for this study site see Burke and Eisenbies (2000).

Experimental Timber Harvests

In 1997, three 11–13-ha portions of the Coosawhatchie floodplain forest were designated as control, clearcut, and patch-retention clearcut tracts (Figure 1). The control tract was intentionally located upstream of the clearcut and patch-retention tracts to better ensure that future harvest activities did not influence the control. After collecting baseline preharvest data in each tract in the 1998 and 1999 winter flood seasons (see below), experimental harvests were conducted the following autumn of 1999 using best management practices for the region. Where heavy equipment was used, machines were operated over beds of cut limbs to minimize impacts on soils; the site was unusually dry at harvest time, so impacts such as rutting and compaction were minor. In the clearcut tract, machine-mounted rotary saws were used to cut all stems. Merchantable stems were removed and nonmerchantable stems were left on the ground. The patch-retention clearcut tract was treated similarly to the clearcut, except two 0.2-ha and one 0.6-ha patches of trees were left undisturbed. One patch was located on flat terrain, the second in a concave depression, and the third on convex higher ground (Figure 1). The primary justification for leaving these patches was to promote subsequent forest regeneration and provide residual habitat for forest birds (Harrison and Kilgo 2004). However, patches also had the potential to harbor residual populations of invertebrates. Both the clearcut and patch-retention tracts were not replanted, but were allowed to regenerate naturally.

Invertebrate Sampling

Aquatic invertebrates were sampled twice during each winter flood. The first sample was collected within 2 weeks of the first flood event, which, depending on the year, ranged from Dec. to early Mar. This collection ensured that we sampled those invertebrates that aestivated in dry substrates and developed quickly after initial floodings (e.g., *Aedes* mosquitoes, crustaceans, some midges). A second sample was collected in Apr. or May of each year as the winter floods were receding, and this collection ensured that we sampled invertebrates whose populations built over the flood season or that colonized and reproduced after the wetland flooded. Because drought conditions began to develop in summer 1998, the site rarely flooded at times other than during these winter periods, and so aquatic invertebrates had few opportunities to develop outside of the designated sampling periods.

We used a D-frame net (30 cm diameter, 1-mm mesh) to collect aquatic invertebrates. This device has been shown to sample macroinvertebrates from wetlands efficiently and precisely (Cheal et al. 1993, Batzer et al. 2001). The 1-mm mesh was small enough to retain most macroinvertebrates, yet large enough to prevent net-clogging. Microinvertebrates (<1 mm) were not efficiently sampled, and thus we do not address those taxa in this study. An individual sample was collected by sweeping the net through a 1-m length of flooded habitat. The net base was scraped along the bottom substrate to collect organisms in surficial sediments, from submersed plant and litter substrates, and in the water column. For each sample event we selected a representative transect in each of the three study tracts and collected 10 subsamples with the net at randomly selected locations along those transects. The 10 subsamples per transect encompassed an area of about 3 m^2. After the patch-retention treatment was harvested, we stratified sampling there so that half of the sweeps were collected in the cutover area and half in the remnant forested islands. Samples were preserved in 95% ethanol and returned to the laboratory for processing. Invertebrates were identified to family or genus by using keys in Merritt and Cummins (1996) and Thorp and Covich (1991), and community compositions were quantified by relative abundance.

Statistical Analyses

Treatments were not replicated in this study, so we used both spatial variation among the three study tracts (control, clearcut, patch-retention) and temporal variation among years (preharvest 1998 and 1999, postharvest 2000 and 2001) to assess invertebrate response to timber harvest. A composite community sample for each tract in each year was developed by pooling the 20 subsamples collected yearly (2 sample dates per year, 10 subsamples per sample transect), generating 12 community profiles (3 tracts $\times$ 4 years). We used two multivariate tests, nonmetric multidimensional scaling (NMS) and cluster analyses (PC-Ord software, MjM software Design, Gleneden Beach, OR; McCune and Grace 2002), to assess dissimilarities among the 12 community profiles based on invertebrate relative abundances. Because the distance measure used in these tests can often affect the result (McCune and Grace 2002), we used both Euclidean and Sorensen distances. Euclidean distance assesses relative abundances and is most influenced by common taxa, whereas Sorensen distance assesses relative presence/absence and is most influenced by rare taxa. Before the analyses, data were $\log(x + 1)$-transformed as recommended by McCune and Grace (2002) for highly variable data with relatively few zero counts. We only interpreted results that were evident using both NMS and cluster analyses, regardless of distance measure. Because NMS is considered the method of choice for ecological community data (McCune and Grace 2002) and spatial relationships in ordination space are easily visualized by the procedure, we relied most heavily on NMS to contrast overall communities. However, because cluster analysis is useful for dividing multivariate data into discrete groups (McCune and Grace 2002), we used that technique to develop groupings for subsequent direct contrasts of individual invertebrate taxa. A nonparametric *t*-test (Wilcoxon rank-sum test) was used to contrast the abundances of individual invertebrate taxa between groups (or clusters).

Results

The invertebrate community at the Coosawhatchie study site (all tracts and years combined) was dominated by seven families: Chironomidae midge larvae (37% of the total), Asellidae isopods (22%), Planorbidae snails (12%), Physidae snails (10%), Crangonyctidae amphipods (5%), Culicidae mosquito larvae (4%), and Sphaeriidae clams (3%). Total taxon richness (number of different families) and total invertebrate abundance varied among study years, being higher in 1998 (19–21 families and 812–1,471 organisms per habitat) than during the following 3 years (11–18 families and 408–887 organisms per habitat). This temporal pattern probably developed because 1998 (preharvest) was unusually wet, whereas the subsequent 3 years encompassed an extended drought. In the high waters of 1998, flow-dependent riverine taxa (blackflies and mayflies) established in the floodplain, and they supplemented the core group of wetland invertebrates that occurred yearly (including the seven families listed above). Flow-dependent organisms occurred rarely in the more stagnant water conditions existing during the drought years of 1999 (preharvest), 2000 (postharvest), and 2001 (postharvest).

Despite annual variation in the invertebrate communities of the Coosawhatchie bottomland, the impact of timber harvest on invertebrates was clearly evident. Whether we used NMS (Figure 2) or cluster analyses, or whether the distance measure used was Euclidean (metric) or Sorensen (proportional), we found similar separation between the communities in harvested versus nonharvested tracts. The NMS ordination indicated that community compositions were similar among the three treatment areas during the preharvest baseline years of 1998 and 1999, although some annual variation was evident (Figure 2A). Communities in the control tract in 2000 and 2001 retained community compositions that were similar to that baseline condition (Figure 2A), indicating that ongoing drought conditions did not significantly alter overall community compositions. However, communities developing postharvest in the clearcut tract in 2000 and 2001 (Figure 2A) deviated strongly from their previous preharvest baseline condition in 1998 and 1999, and from the communities coexisting in the control tract during 2000 and 2001. The patch-retention clearcut tract also deviated from its baseline condition, and held an intermediary position in ordination space between the clearcut and control tracts during 2000 and 2001 (Figure 2A).

Cluster analyses also indicated a clear division between the communities in postharvest clearcut tracts in 2000 and 2001 and nonharvested wetland (control tract in 1998–2001, clearcut and patch-retention clearcut tracts in 1998 and 1999). That analysis also placed the postharvest patch-retention clearcut tract in an intermediary position, but indicated that patch-retention habitats shared more characteristics with the clearcuts than the nonharvested wetland habitats. Thus, we felt justified in combining the postharvest clearcut and patch-retention clearcuts into a single category of harvested habitats, and then directly contrasting the abundances of individual taxa between harvested ($N = 4$) and nonharvested wetland ($N = 8$). However, we were concerned that the unique conditions and invertebrate communities that developed in 1998, when only nonharvested wetland occurred, might skew results for the nonharvested category. To address this concern, we conducted comparative analyses both with and without 1998 data, and only considered a result valid if both testing procedures indicated that a significant difference existed between harvested and nonharvested wetland.

Four of the seven dominant invertebrate families differed between harvested and nonharvested wetland. Three families, Planorbidae (Figure 2B; Wilcoxon *t*-test, df = 11; $P = 0.002$), Crangonyctidae (Figure 2D; $P = 0.002$), and Asellidae (Figure 2E; $P = 0.004$), occurred in significantly higher densities in nonharvested than harvested wetland. In contrast, Culicidae (Figure 2F; $P = 0.002$) were more abundant in harvested than nonharvested wetland. Two of the remaining dominant families, Chironomidae ($P = 0.141$) and Sphaeriidae ($P = 0.280$), were similarly abundant in both habitat types. Contrasts involving Physidae were equivocal, with numbers of this snail being marginally higher in nonharvested than harvested wetland if all 4 years were considered (Figure 2C; $P = 0.045$), but not if data from 1998 were deleted ($P = 0.143$).

Family richness appeared to be somewhat lower in the harvested than nonharvested wetland (Wilcoxon *t*-test, df = 11, $P = 0.024$; Figure 3A), but not if the taxa-rich year of 1998 was dropped from the analysis ($P = 0.095$). In addition, the lowest richness detected in the study (11 families) occurred in the control tract in 2000, suggesting that any association between harvest and low taxa richness was weak. Probably because some taxa decreased while others

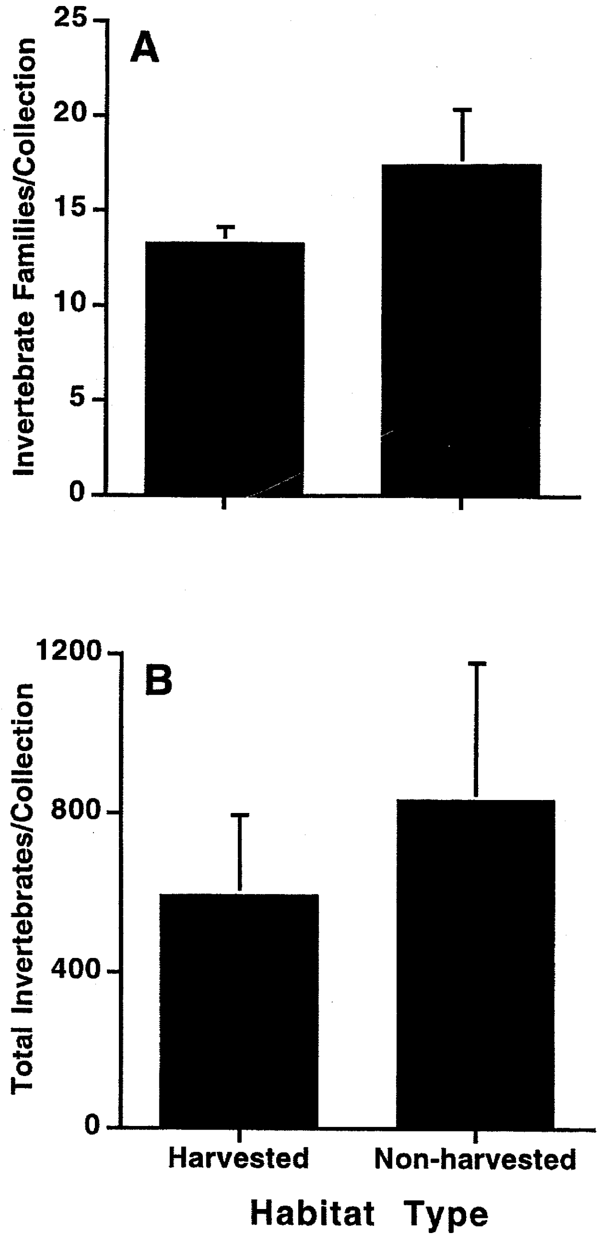

Figure 3. (A) Invertebrate family richness (±SD) collected per year (composite of 20 sweep net samples) from harvested and nonharvested wetland tracts in the Coosawhatchie River floodplain. (B) Total numbers of invertebrates (±SD) collected per year in harvested and nonharvested wetland tracts.

increased in response to harvest, and chironomid midges (the numerically dominant family) did not respond, overall invertebrate numbers were not different in harvested and nonharvested habitats ($P = 0.165$; Figure 3B).

Discussion

Timber harvest altered the structure of aquatic invertebrate communities on the Coosawhatchie floodplain (Figure 2). Some invertebrates were benefited by harvest (Culicidae), whereas others were negatively affected (Asellidae, Crangonyctidae, Planorbidae). Neither total invertebrate abundance nor family richness was affected by harvest, however. A productive aquatic-invertebrate community occurred in both harvested and nonharvested wetland.

The change in invertebrate community structure associated with timber harvest probably developed because ecological conditions changed functionally. Before harvest, the trophic basis of invertebrate production was leaf litter and wood. The asellids, crangonyctids, planorbids, and physids that flourished under those conditions were primarily detritivores, feeding on dead plant material and associated biofilms (Pennak 1989). After harvest, leaf litter was no longer an abundant food, and these organisms would have to shift their diets. In addition to food supplies, physical conditions for these invertebrates probably changed. Asellids, crangonyctids, planorbids, and physids do not have particularly well-developed desiccation resistance strategies (Pennak 1989), and before harvest-canopy shading and a moist layer of leaf litter probably enabled them to survive dry periods. After canopy-shading and leaf-litter input were largely eliminated, conditions during dry periods were probably harsh and stressful for these organisms. The drought conditions that existed after harvest might have exacerbated this problem. The degree of exposure during dry periods is known to strongly influence invertebrate community structure in wetlands (Wissinger 1999).

However, harvest might enhance habitat conditions for midges and mosquitoes. Increased levels of sunlight probably stimulated growth of algae, and midges and mosquitoes readily consume high-quality algal foods (Coffman and Ferrington 1996, Walker and Newson 1996). Harsh conditions during dry periods would not be a problem for the mosquitoes (*Aedes*, *Psorophora*) because they have highly desiccation-resistant eggs (Walker and Newson 1996). Many of the midges aerially colonized the Coosawhatchie bottomland after it flooded (larval densities tended to peak late in the hydroperiod), and so conditions during dry periods probably minimally affected them.

Harvesting wetland trees can affect site hydrology by influencing evapo-transpiration and interception rates (Sun et al. 2001). However, in this study, variation in hydroperiods (duration of wet phases) seemed an unlikely mechanism to induce the invertebrate community change associated with harvest. The study encompassed unusually wet (1998) and unusually dry (1999–2001) years. Invertebrate response to these extremes in hydrologic variation was detected by contrasting annual change in the control treatment. However, this change was modest in comparison to the change induced by harvest. Batzer et al. (2004) suggest that invertebrates that successfully exploit seasonal wetland habitats routinely experience pronounced change in hydroperiods, and they possess the flexibility to cope with such variation.

We cannot directly extrapolate our results to other systems because treatments were not spatially replicated. However, invertebrate response to harvest at the Coosawhatchie was consistent with changes observed elsewhere by other animal groups. What we observed in this study appeared to be a forested wetland invertebrate fauna being replaced after harvest by a seasonal marsh fauna. Amphibians exhibit similar changes. After harvest, an amphibian community dominated by salamanders is frequently replaced by one dominated by frogs and toads (Enge and Marion 1986, Phelps and Lancia 1995, Clawson et al. 1997, Perison et al. 1997). Invertebrates might influence amphibian replacement. Larval salamanders are visual predators that find large, actively swimming invertebrates, like asellid and crangonyctid crustaceans, to be easy prey (Wissinger et al. 1999). The midges that dominated the postharvest communities at the Coosawhatchie were small and sedentary, so they might be less obvious prey items for salamanders. Although active, swimming mosquito larvae were probably readily consumed by salamanders, mosquitoes develop very rapidly and are available for consumption for only brief periods. The frog larvae occurring postharvest are mostly algivorous, like mosquitoes and midges, and so trophic impacts of harvest on both groups may be similar, although probably not interdependent. However, because frog adults feed on adult insects, midge and mosquito larvae might eventually become vulnerable prey after emerging.

Change in the invertebrate community was also consistent with typical postharvest changes in avian faunas (Moorman and Guynn 2001, Harrison and Kilgo 2004). In forested wetlands, most birds live in the canopy and feed on insects (moths, caterpillars) that develop there. The snails and crustaceans living in the water below are not important foods for most of these birds, although on emergence, midge adults may fly up into the canopy and supplement the diets of canopy-dwelling birds. After harvest, the interaction between the aquatic invertebrate and avian faunas probably becomes more important. Most of the midges that developed in the harvested tracts were in the genus *Chironomus*, whose large, sedentary larvae are favored foods of dabbling ducks that sieve them from bottom substrates (Batzer et al. 1993). Additionally, as midges emerge, they become particularly vulnerable to ducks, ducklings, and swallows foraging at the water's surface (Murkin and Batt 1987). The insect-dominated fauna of the harvested wetland might have contributed an abundance of flying foods for birds and bats that find the open, meadow-type habitat a useful foraging area (Blake and Hoppes 1986). Large, mating swarms of midges or mosquitoes frequently develop over marshy habitats.

It was difficult to qualify invertebrate faunal change associated with timber harvest at the Coosawhatchie as being negative or positive, it simply reflected a change in ecological function. Significant progress has been made in preserving remnant bottomland wetlands. However, conversion of forested habitat into marsh habitat is a growing concern (Dahl 2000). Ensuring that harvested wetland successfully regenerates back into bottomland hardwood forest should probably be a priority (Roy et al. 2000). Our finding of increased mosquito production might be a negative influence associated with harvest because issues of public health could come into play. Fortunately, the floodwater *Aedes* and *Psorophora* mosquitoes found in this study are not considered important disease vectors, although they are aggressive human-biters.

The patch-retention management option may provide a means to mitigate some of the environmental change associated with clearcut harvest. Leaving remnant patches of intact, forested habitat scattered across clearcuts appeared to provide a refuge for invertebrates, and may reduce impacts of harvest on these organisms. There were also indications that recovery of some initially impacted invertebrate populations was rapid (see Figure 2E). Harrison and Kilgo (2004) studied bird responses to harvest in these same Coosawhatchie study sites and found that, as for the invertebrates, patch-retention reduced initial impacts of clearcut harvest on avian communities. Patch-retention may provide multiple ecological benefits for harvested wetlands, especially in cases where minimal intact wetland habitat persists in adjacent tracts, and the technique merits further investigation.

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