

## Evaluation of a Stream System after Clearcut Logging Disturbance in the Gulf Coastal Plain

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### ABSTRACT

We examined potential impacts of removal of timber, road construction, and military operations on a stream system at Fort Polk, Louisiana. In 1989, approximately 1,057 ha of upland pine and riparian hardwood timber were removed from the middle section of the Birds Creek watershed. In addition, roads were installed to facilitate vehicular passage during military exercises. Approximately 2.6-km of Birds Creek stream length occurred within the logged portion, which has been maintained as a cleared area since timber was harvested. During 2001-2005, we evaluated the assemblage structure of fishes and macroinvertebrates and the associated habitat at five sites in Birds Creek and five sites in an adjacent but unaffected stream, Whiskey Chitto Creek. Whatever the effects of timbering and construction, 12-yr after the disturbance the affected sites on Birds Creek contained heterogeneous habitats that supported rich and diverse fish and macroinvertebrate assemblages, not unlike those of the other sites on Birds Creek and the adjacent control stream.

### INTRODUCTION

In stream systems, the structure of biological communities is influenced by variation in stream habitat (Gorman and Karr 1978, Vannote et al. 1980, Minshall 1988, Williams et al. 2003), local watershed conditions (Richards and Host 1994, Wang et al. 1997, Meador and Goldstein 2003), and natural disturbances (Resh et al. 1988, Williams et al. 2002). Concomitant with this natural variation in environmental conditions are anthropogenic disturbances, like timber harvest practices, that also influence structure of stream communities (Campbell and Doeg 1989, Williams et al. 2002). These relationships, however, are complicated in that they vary across spatial, temporal, or taxonomic scale (Wiens et al. 1986, Angermeier 1987, Lammert and Allan 1999, Williams et al. 2002). For example, the impact of clearcutting on a stream system may depend on the geographic location and aerial extent of the impacted area, along with time since the disturbance occurred and the particular taxonomic group examined (Williams et al. 2002).

Over the last approximately 20 years, considerable effort has been invested in understanding how timber harvesting activities impact a variety of stream functions (Campbell and Doeg 1989). While the potential impacts of timber harvesting are well studied in Pacific Northwest streams (Hicks et al. 1991, Williams et al. 2002), fewer investigations have occurred in southeastern and gulf slope rivers and streams (Ensign and Mallin 2001, Williams et al. 2002, Davis et al. 2003). Previous work has demonstrated the potential impacts of clearcutting can include decreased dissolved oxygen (Binkley and Brown 1993), increased sedimentation (Waters 1995, Lockaby et al. 1997), alterations to stream geomorphology or wood inputs (Ralph et al. 1994, Poole et al. 1997, Gregory et al. 2003), increased nutrient levels and sunlight producing extensive

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algal blooms (Ensign and Mallin 2001), and short and long-term changes to fish and macroinvertebrate assemblage structure (Silsbee and Larson 1983, Campbell and Doeg 1989, Williams et al. 2002).

The timber industry is of major economic importance in the southeastern USA (Costanza et al. 1997), and many studies link reduction of forest cover to increased sedimentation and decreased habitat quality (Binkley and Brown 1993, Richards et al. 1996, Sutherland et al. 2002). Despite the importance of timber harvest in the lowland southeast (Warren et al. 2000, Williams et al. 2002), most studies related to forest cover removal have been conducted in western and eastern upland streams (Berkman and Rabeni 1987, Sutherland et al. 2002, Williams et al. 2002). Previous work that has examined impacts of timber removal on biotic communities in southeastern streams produced inconsistent results, with some authors finding little impact on biota (Wallace and Gurtz 1986, Rutherford et al. 1992, Rice et al. 2001, Williams et al. 2002). While lack of consistent findings in these stream systems may be related to variation in the spatial or temporal scales of individual studies, a recent study by Williams et al. (2002) indicated that the severe natural disturbance regime in many southeastern streams may limit the potential impacts of timber removal or other anthropogenic disturbances. Lowland, coastal plains streams in Louisiana have a harsher natural disturbance regime (Peckarsky 1983) than many eastern upland and western streams.

Nearly half of the streams and rivers in the USA are impacted by excess fine sediment (Judy et al. 1984), with major sources being agriculture, forestry, urbanization, mining, and road construction. Sedimentation is cited as the greatest threat to water quality in North American streams (Waters 1995). Because excess fine sediment is the most significant deleterious outcome from timber removal, it deserves special attention. Excess fine sediment in streams alters instream habitat and has caused shifts in fish and macroinvertebrate assemblages (Angradi 1999, Shaw and Richardson 2001, Sutherland 2002). Assessing the impact of excess sediment on stream ecosystems is difficult, however, in that there can be great variation in the distribution of fine sediments both spatially and temporally (Downes 1990). Also, most studies have focused on the behavior of few taxa or the effects of a single, high-level sediment pulse on assemblage structure (Gray and Ward 1982, Angradi 1999). Despite the high profile of sediment management in streams (Bryant 1995, Wood and Armitage 1997), few studies have examined the impact of a persistent disturbance, like sedimentation, on the structure of multiple taxonomic assemblages over larger spatial and/or temporal scales (Williams et al. 2002).

We examine the potential impacts of a 1057-ha clearcut on Birds Creek watershed in west-central Louisiana. Timber was removed in 1989 and the area has since been maintained as a cleared zone by mowing and burning for military training (U.S. Army, Fort Polk). We examined habitat and structure of fish and macroinvertebrate assemblages in Birds Creek and the adjacent, unlogged Whiskey Chitto Creek watershed. We predicted that clearcutting and subsequent maintenance as a cleared area would alter the biological communities and habitat (e.g., turbidity, total suspended sediment, substrate types) of Birds Creek.

## METHODS AND MATERIALS

### *Study area*

Birds Creek and Whiskey Chitto Creek are located on Fort Polk Army Base in Vernon Parish in west-central Louisiana, encompassing the easternmost low-gradient drainages of the western gulf slope. The watersheds of Vernon Parish are gently sloping to flat with loamy to clay soils, with pine forests in the upland areas and mixed hardwoods common along lowland areas. Within Fort Polk, the timber of the region are

fire climax communities. Riparian zones have only been harvested during construction projects and not for timber sales; thus, most stream segments contain a relatively unimpacted and climax forest riparian zone.

In 1989, the U.S. Army installed a multi-purpose range complex (MPRC) within Fort Polk. To construct the MPRC, over 1,057 ha of upland pine and riparian hardwood timber were removed from the middle section of the Birds Creek watershed (Fig. 1). In addition, roads were installed to facilitate vehicular passage during gunnery and maneuver exercises. The MPRC impacted 2.6-km of linear stream length on Birds Creek. Timber was cut all the way to stream banks resulting in highly altered riparian zones. Twenty-two sediment basins were constructed prior to timber harvest in an attempt to ameliorate potential impacts of sedimentation. The MPRC has been maintained since 1989 as a grassland training area. While riparian areas have recovered since initial disturbance, only a narrow band (<10-m in most places) of young trees, mostly pines, exists along Birds Creek in the MPRC.

We sampled five sites on Birds Creek (Fig. 1). Because pre-disturbance data for Birds Creek did not exist, we also sampled five sites on the adjacent Whiskey Chitto Creek for comparative purposes (Fig. 1).

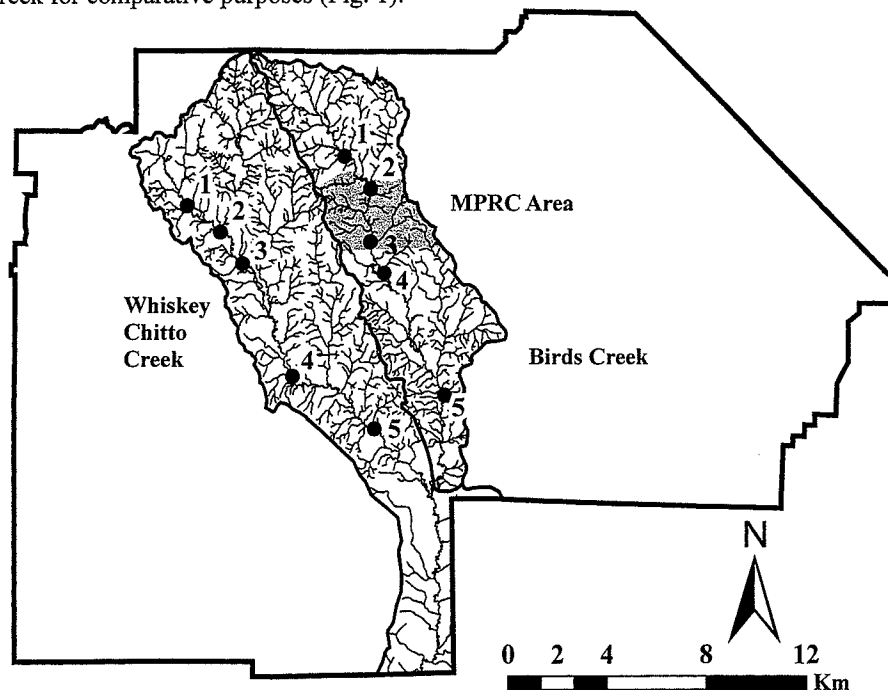


Figure 1. Map showing the Whiskey Chitto Creek and Birds Creek watersheds within the Ft. Polk Military Base. The boundaries of the MPRC area and the stream sampling sites also are shown.

#### *Sampling methods*

Biota was collected from ten representative sites, five sites per stream, seasonally (four times per year – typically June, November, January, and April) from June 2001 to April 2003. Each site consisted of a minimum 100-m reach containing mesohabitats (i.e., runs, riffles, and pools), substrate, and large wood in similar proportions to that of the entire stream. Each of the ten sites was divided into five subreaches for sampling of biota and habitat.

Fishes were sampled at each site by multiple-pass depletion electrofishing with a Smith-Root model-14 backpack electroshocker and seines. Block-nets were placed at the downstream end, midpoint, and upstream end of each site. Fishes were identified to species, enumerated, and released, except for voucher specimens, which were transferred to the Monroe Museum of Natural History at the University of Louisiana.

Four times a year we collected macroinvertebrates at each of the five subreaches, separately, within each site. We used a D-frame kick-net to sample all available microhabitats, both in flowing water and along the stream's edge. We also took a Surber sample in each of the five subreaches. Macroinvertebrates were preserved in 95% ethanol and rose-bengal stain and were sorted and identified to the family level.

We used a YSI Model 600 multiprobe meter to measure temperature, conductivity, pH, dissolved oxygen (DO), and turbidity for each sampling period at each site. We also collected water samples using a DH48 depth-integrating suspended sediment sampler. Samples were taken vertically throughout the water column and repeated for the complete cross section giving an integrated sediment sample for the wetted width. Samples were vacuum filtered through weighed glass-fiber filters, dried in a 103-105°C oven for 1h, and reweighed to determine total suspended solids (TSS; Bain and Hynd 1999). In addition, samples of substrate were taken with a grab sampler within each site during summer 2001 to estimate percent substrate composition. Samples were taken at the center point of each subreach and approximately 5-10 individual samples were collected across the channel depending on width. A sub-sample of the composited material was dried, sieved, and appraised as % gravel, sand, and silt. We measured current velocity (Marsh-McBirney Flowmate), depth, and width along five transects per reach to calculate discharge. Maximum depth also was recorded for each transect. At the same transects, we visually estimated bank stability as the percentage of bank that was not eroded. Visual estimates of cover, including large wood, detritus, and vegetation, also were made along each transect.

#### *Analytical methods*

To initially examine differences in the structure of fish and macroinvertebrate assemblages across sampling sites, we computed average richness and evenness. Richness was determined at the species-level for fishes and at the family-level for macroinvertebrates. Evenness (Buzas and Gibson's  $E$ , where  $E = e^H/S$ ;  $H$  is Shannon-Weiner diversity and  $S$  is species richness) was computed using the same level of taxonomic precision as richness (Hayek and Buzas 1997). We used analysis of variance (ANOVA) to test for differences in richness and evenness among sites. To examine differences in ecological function of these streams, we assessed proportional trophic composition of fishes and macroinvertebrates (Horwitz 1978, Merritt and Cummins 1996, Allan 1995, Matthews 1998) at each site. Seven groups of macroinvertebrates were excluded from further statistical analyses because our methods were not designed to collect them efficiently; they were Araneae, Bivalvia, Cladocera, Copepoda, Decapoda, Gastropoda, and Hydroida.

We summarized habitat characteristics for each stream across seasons and computed means for all measured variables. In addition, we calculated coefficient of variation (CV) for width, depth, and current velocity. T-tests (assuming unequal variances; Quinn and Keough 2002) were used to test for differences in individual habitat variables between Birds Creek and Whiskey Chitto Creek. Variables with missing or out of range values because of instrument error (i.e., DO and conductivity) were deleted from further multivariate analyses. We used the remaining variables in a principal components analysis (PCA) to measure temporal and spatial differences in habitat by stream, site, and season (Williams et al. 2005).

We used canonical correspondence analysis (CCA; ter Braak 1986) with the software package CANOCO (Version 4.5; ter Braak and Smilauer 2002) to examine relationships between fish and macroinvertebrate assemblages and habitat variables. Because of the large number of rare species, we used the option of downweighting rare species (ter Braak and Smilauer 2002). Significance of CCA was assessed with a Monte Carlo randomization procedure using 1,000 permutations (ter Braak and Smilauer 2002).

## RESULTS

Overall, richness and evenness of fishes showed significant differences among sites (Fig. 2; Richness  $F = 2.10$ ,  $P = 0.045$ ; Evenness  $F = 2.43$ ,  $P = 0.021$ ). Species richness was highest at Birds Creek Site 3 and evenness was highest at Birds Creek Site 2, both within the MPRC. Though not statistically significant (Richness  $F = 1.94$ ,  $P = 0.065$ ; Evenness  $F = 0.123$ ,  $P = 0.998$ ), richness of macroinvertebrates was higher in Birds Creek overall than in Whiskey Chitto Creek.

When macroinvertebrate assemblages were examined using trophic classification, all sites had similar proportion of functional feeding groups. The assemblages at all sites consisted of mostly collector taxa (Birds Creek average 78% +/- 0.84 SE; Whiskey Chitto Creek 81% +/- 2). Most fishes collected in this study were insectivores (Birds Creek 88.8% +/- 1.74; Whiskey Chitto Creek 92.2% +/- 1.8). The only site-level difference was a higher proportion of piscivores in Birds Creek Site 2 in the MPRC (average of 6%

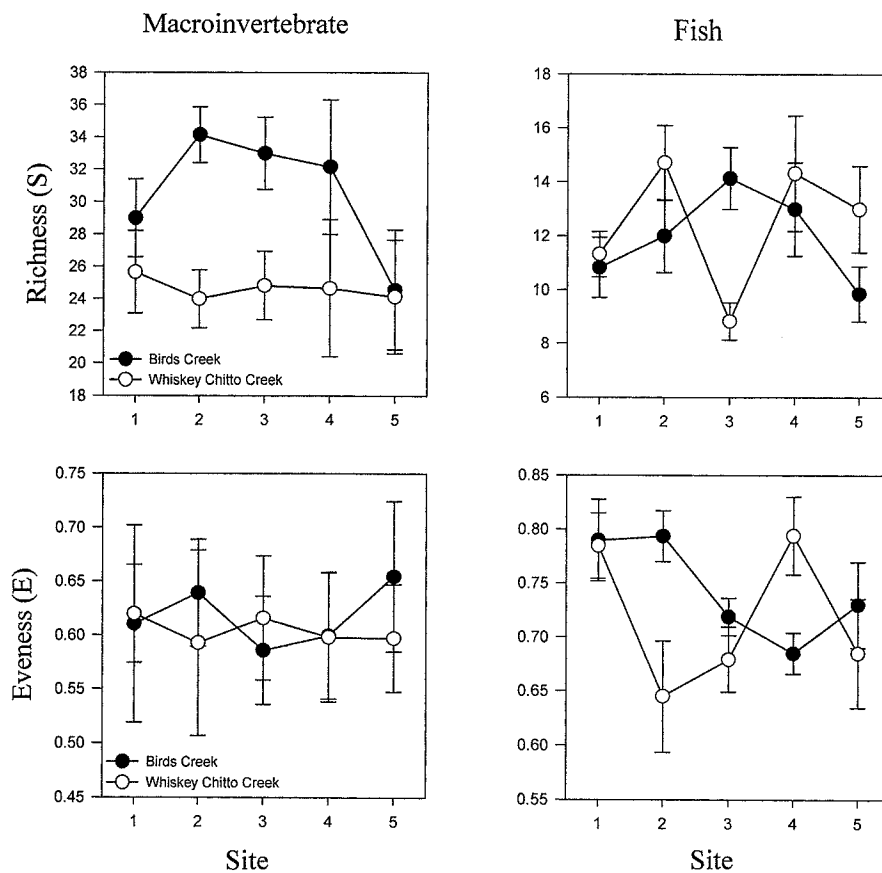


Figure 2. Mean (+/- 1 SE) richness and evenness (E) in Birds Creek and Whiskey Chitto Creek, June 2001 to April 2003.

versus overall average of 2.8% +/- 0.86), mostly attributable to higher relative abundances of centrarchid taxa.

Habitat conditions were similar in Birds Creek and Whiskey Chitto Creek across all sites. There were, however, some notable differences between the two streams. Mean (+/- SE) conductivity was statistically higher ( $t = 2.202, P = 0.002$ ) in Whiskey Chitto Creek ( $0.23 \pm 0.06 \text{ mS/cm}^3$ ) than Birds Creek ( $0.08 \pm 0.02$ ), as was TSS (Whiskey Chitto Creek  $10.3 \pm 1.86 \text{ mg/L}$ , Birds Creek  $6.4 \pm 1.04$ ;  $t = 2.01, P = 0.037$ ; Fig. 3). Though not statistically significant ( $t = 2.57, P = 0.179$ ), average fine substrate (i.e., silt)

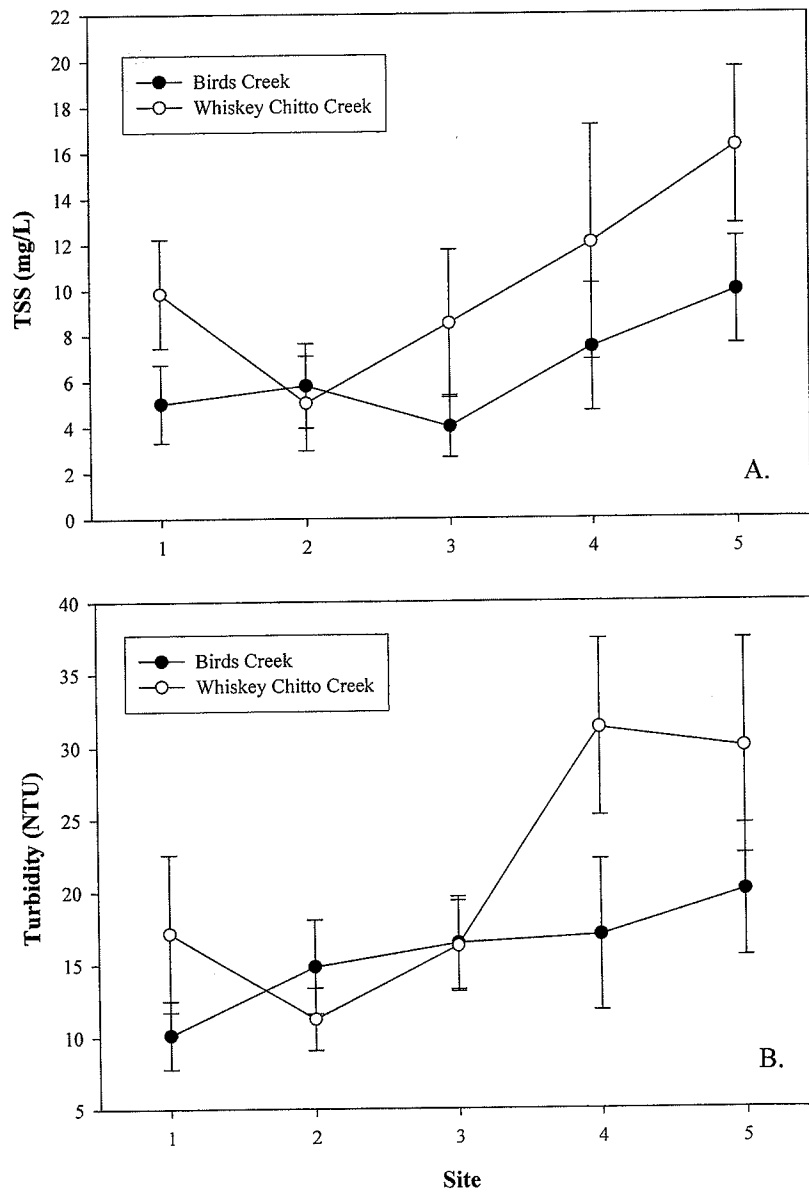


Figure 3. Mean (+/- 1 SE) amount of (A) Total Suspended Solids (TSS) in mg/L and (B) turbidity (NTU) at each site on Birds Creek and Whiskey Chitto Creek, June 2001 to April 2003.

was higher overall in Whiskey Chitto Creek (1.36% +/- 0.49) than in Birds Creek (0.55% +/- 0.16; Fig. 4). Birds Creek, on average, had more stable banks (70.1% stable, +/- 3.26) than Whiskey Chitto Creek (65.5% +/- 3.14;  $t = 2.00$ ,  $P = 0.046$ ).

Principal components analysis separated sites along two major habitat gradients (Fig. 5). The first PCA axis represented a stream size gradient with sites draining larger watersheds (i.e., downstream-most sites) loading positive on the first PCA axis associated with greater discharge, TSS, and depth. The second PCA axis primarily represented a seasonal gradient with discharge and TSS higher in the spring, corresponding with high rainfall events, and CV current velocity and percent cover corresponding with low rainfall events during the summer. Together the first two axes explained 48.7% of the overall variance and were considered statistically significant ( $P < 0.01$ ) when compared to a broken stick model (Jackson 1993, Williams et al. 2002).

Fish and macroinvertebrate assemblage association with habitat conditions was similar in Birds Creek and Whiskey Chitto Creek overall. Most sites occurred towards the middle of the CCA ordination biplots (Fig. 6), indicating that assemblages in the ten sites were similarly related to habitat variables. Some separation of sites was evident, however, with the most downstream Whiskey Chitto Creek site occurring along the top portion of the ordination associated with greater discharge, TSS, and maximum depth. The fish fauna at the lower Whiskey Chitto Creek site, for example, contained more riverine taxa like silvery minnow (*Hybognathus nuchalis*) and channel catfish (*Ictalurus punctatus*). Sites in the MPRC also separated from other localities, occurring towards the bottom of the ordination plot associated with greater bank stability, temperature, and CV of width and current velocity. Several fish species had their largest abundance in the MPRC, like Sabine shiner (*Notropis sabiniae*), tadpole madtom (*Noturus gyrinus*), freckled madtom (*Noturus nocturnes*), and many of the centrarchids [e.g., bluegill (*Lepomis macrochirus*), longear sunfish (*Lepomis megalotis*), redspotted sunfish (*Lepomis miniatus*), warmouth (*Lepomis gulosus*), and spotted bass (*Micropterus punctulatus*)]. In addition, three macroinvertebrate families were unique to the MPRC –

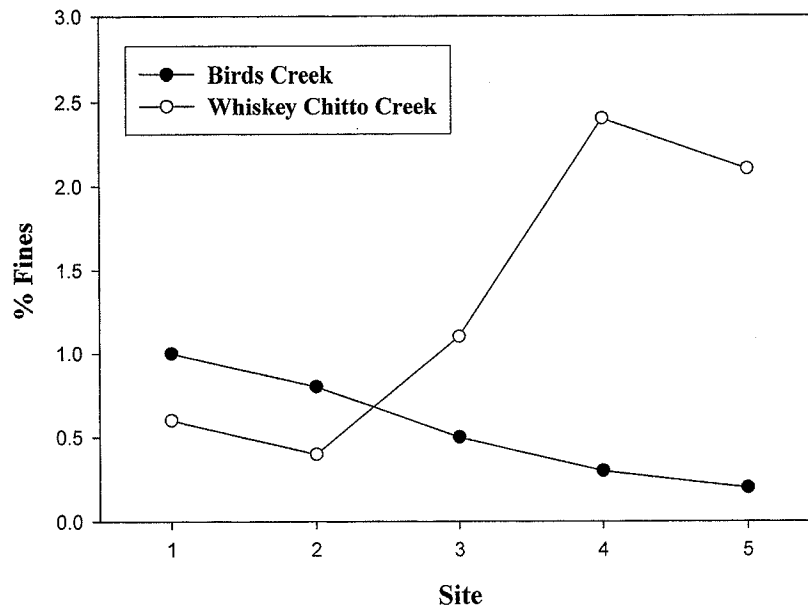


Figure 4. Percentage of fine sediments in substrate samples in June 2001 at Birds Creek and Whiskey Chitto Creek.

Phryganeidae (Trichoptera), Chrysomelidae (Coleoptera), and Braconidae (Hymenoptera); and two more had their largest abundance in the MPRC – Taeniopterygidae (Plecoptera) and Hydroptilidae (Trichoptera). Giant case makers (Phryganeidae), aquatic leaf beetles (Chrysomelidae), and braconid wasps (Braconidae) are associated with macrophytes in areas of slow current velocity (McCafferty 1998, Voshell 2002). Winter stoneflies (Taeniopterygidae) are a somewhat sensitive family that is associated with coarse substrates (e.g., leaf packs, wood jams, submersed plants), and micro-caddisflies (Hydroptilidae) are associated with macrophytes or filamentous algae (Wiggins 2000, Voshell 2002). These five macroinvertebrate taxa all were associated with macrophytes, and Birds Creek Site 2 was the only site that had a dense bed of macrophytes, probably related to the lack of canopy cover, throughout the entire period of the study. CCA analyses were statistically significant with Monte Carlo test for both macroinvertebrates ( $F = 1.466, P = 0.001$ ) and fishes ( $F = 1.644, P = 0.001$ ; Fig. 6).

#### DISCUSSION

Historically, streams in the southeastern United States experienced frequent disturbance from weather events and fire (Rebertus et al. 1993, Glitzenstein et al. 1995). Stream fish and macroinvertebrate species in gulf coastal plain streams have thus evolved tolerance to a broad range of stream conditions as a result of historical disturbance regimes (Stewart et al. 1976, Williams et al. 2005). As such, the impacts of anthropogenic disturbances like clearcutting are expected to differ from, and perhaps be less severe than, those in other geographic regions with less severe natural disturbance regimes. In addition, the high number of tolerant and/or generalist taxa in gulf coastal plain systems would seem to indicate high rates of recovery from disturbances (Conner and Suttkus 1986, Meffe and Sheldon 1990, Williams et al. 2002).

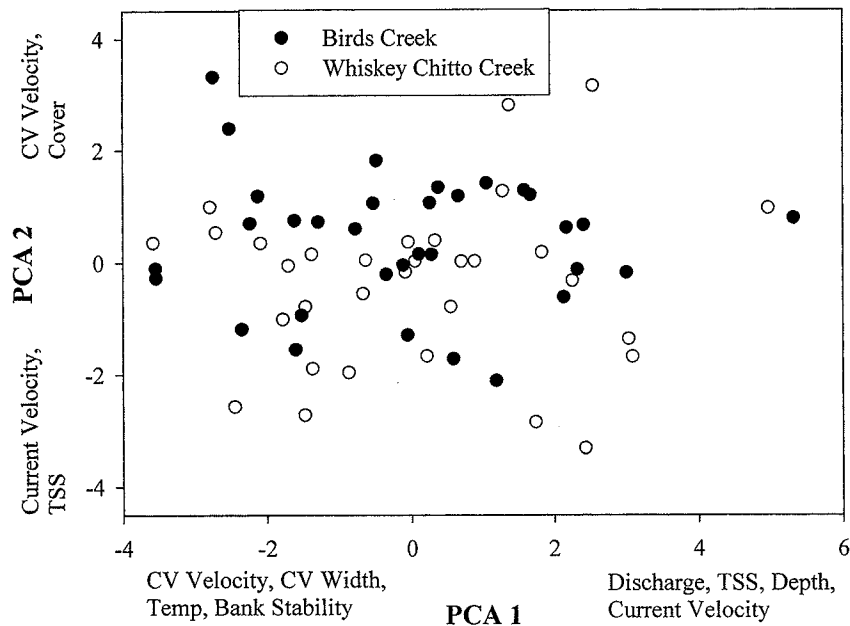


Figure 5. The first two axes from principal components analysis (PCA) on habitat data from Birds Creek (closed circles) and Whiskey Chitto Creek (open circles). Habitat variables loading high (eigenvectors  $< -0.2$  or  $> 0.2$ ) on PC axes 1 or 2 are indicated in the graph margins. Eigenvalue for axis 1 = 3.962 and for axis 2 = 1.886.



We found that the fish and macroinvertebrate assemblages in the MPRC at Fort Polk, Louisiana were not severely impacted by clearcutting that occurred in 1989 and subsequent maintenance efforts. In fact, the highest species richness occurred in the two sites associated with the MPRC. While higher richness certainly may not indicate a healthy ecosystem because of high numbers of exotic and tolerant native species (Scott and Helfman 2001), we found that some important taxa like the sabine shiner (the only fish species of concern in the watershed; Williams and Bonner 2006) had their highest

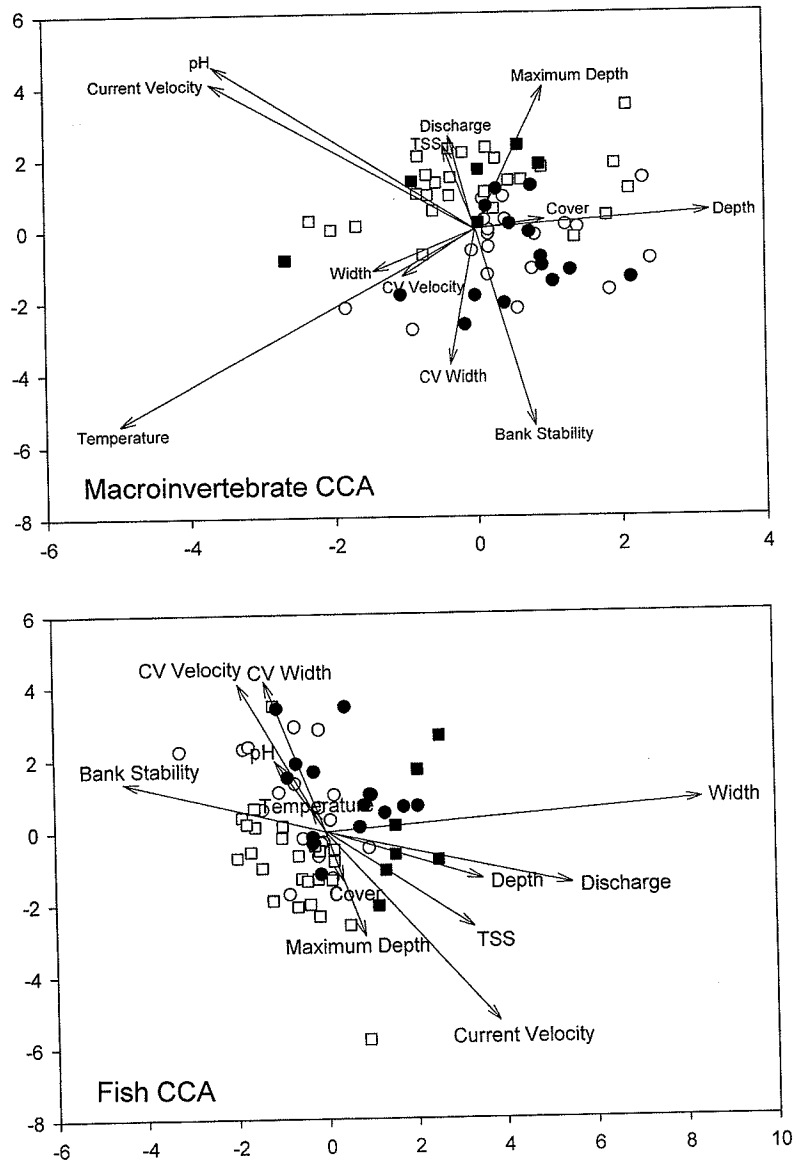


Figure 6. The relationship between environmental variables, sites, and fish and macroinvertebrate assemblages as shown by ordination diagrams of the first two CCA axes and centroids for sites (circles = Whiskey Chitto Creek; squares = Birds Creek). MPRC sites and Site 5 on Whiskey Chitto Creek are indicated with closed symbols.

abundances in the MPRC. Many other taxa of fishes and macroinvertebrates had their greatest abundances in the MPRC.

Other studies in the southeastern United States and elsewhere have shown similar patterns. Williams et al. (2002 and 2005) found that timber harvesting, both at the landscape/whole watershed and reach scales, associated with small streams in Arkansas and Louisiana did not have a significant effect on assemblages of fishes or macroinvertebrates. Likewise, Lockaby et al. (1997) found that clearcutting followed by natural regeneration and accompanied by good conservation measures had small and brief effects on water quality. In another study conducted in Louisiana, Kaller et al. (2005) found that seasonal and spatial variation, instead of timber removal, were most important in structuring macroinvertebrate assemblages. In Appalachian streams, Angradi (1999) also found greater variation among sites in macroinvertebrate assemblages than those shifts caused by excess fine sediment. In a study of fish assemblage response to timber harvesting in Malaysia, Martin-Smith et al. (1999) found few long-term changes in species composition or abundance and demonstrated resilience of assemblages to timber removal.

In general, sites in the MPRC had less silt, probably because of greater variability in flows and higher gradient. The two sites in the MPRC also generally had greater habitat heterogeneity and more well developed riffle-pool morphology. Differences in habitat associated with the MPRC likely relate to past and current disturbance events, as increased variability in flows has been shown to result from timber removal (Campbell and Doeg 1989). We also have observed that increased habitat complexity may relate to the lack of beaver activity, relative to other sites in these watersheds, although this trend was not quantified. Beaver activity in these systems tends to cause streams to be more sluggish with deeper pools (Naiman et al. 1988, Snodgrass and Meffe 1998). The increase in habitat complexity probably explains why so many fish species had their greatest abundance in the MPRC, including species characteristic of riffles and/or cover (madtom species, Ross 2001, Chan 1995), runs (Sabine shiner, Heins 1981, Williams and Bonner 2006), and pools (centrarchids, Ross 2001). Also, bank stability was higher on average in Birds Creek, probably because of grasses and primary succession woody invaders that established themselves in the less shaded MPRC. The invasion of these taxa would be directly related to timber removal along riparian areas in the MPRC. Some of these invading taxa have extensive root systems, and horizontal growth into the stream was evident at MPRC sites. In sites with more mature riparian forests, shading from trees kept grasses from growing along banks and erosion of soils was more evident around the roots of trees.

There may be theoretical reasons to expect greater diversity in the MPRC as well. Huston's dynamic equilibrium model (Huston 1979 and 1994) states that depending on the rate of competitive exclusion and population growth, diversity will peak at different levels of disturbance (Resh et al. 1988, Reice et al. 1990). Huston predicted that higher richness would be associated with disturbance if populations have high growth rates and density is reduced such that competitive exclusion does not occur. Stream communities in coastal plain streams are typically populated by wide-ranging generalists and endemic species that evolved under a dynamic natural disturbance regime, and many taxa exhibit high growth rates as would be predicted by Huston's model (Williams et al. 2002, 2003, and 2005). Another recent study (McCabe and Gotelli 2000) found that, although increased disturbance intensity and area decreased the abundance of macroinvertebrates, richness increased as a result of disturbance, which is consistent with the multi-trophic disturbance models of Wooten (1998). These models, while theoretical, provide some support for our findings. Adaptation to environmental change occurring through historical processes has produced taxa in gulf coastal plain streams that are tolerant of

abiotic disturbances (Conner and Suttkus 1986). Thus, many of the taxa in these streams have life histories that enable them to withstand frequent and more intense disturbances and adapt more easily to anthropogenic disturbances. In contrast, many of the taxa (e.g., salmonids) that inhabit streams in the Pacific Northwest are less tolerant of frequent abiotic disturbances, and in particular, timber harvesting (Chamberlin et al. 1991, Shaw and Richardson 2001).

While excess fine sediment may be the most serious threat to aquatic biota in streams (Judy et al. 1984, Waters 1995), systems with natural substrates of silt and sand may be less susceptible to these impacts. We found both taxonomic and trophic assemblages to be relatively unaffected by clearcutting and subsequent potential impacts of sedimentation. A recent study also examined impacts of military training activities (Quist et al. 2003) and found some contrasting results with our study, but the streams in their study were in the Flint Hills of Kansas and have larger substrates, are dominated by limestone, are spring-fed, occur in a grassland ecosystem, and have much less severe natural disturbance regimes than those in Louisiana (Peckarsky et al. 1983, Williams et al. 2005). Impacts of increases in sediment on fish and macroinvertebrate assemblages have been well documented in Pacific Northwest streams (Everest et al. 1987, Hicks et al. 1991, Shaw and Richardson 2001), but these effects are primarily related to fine sediments filling in the interstitial spaces of gravel (Hicks et al. 1991), which is not a concern in the sand-bottom streams that we sampled.

In summary, we found little evidence of faunal differences between a stream that experienced clearcutting and maintenance by the military and one that had not received such treatment. Sediment basins that were constructed as part of the MPRC installation seem adequate in reducing or eliminating any detrimental effects of sedimentation. Unfortunately, we do not have 12 years of data following the disturbance, and so we cannot determine the initial impacts of timber harvesting, nor can we ascertain when the system recovered most of its diversity, if ever it was lost. We have demonstrated, however, that in the system we studied stream fish and macroinvertebrate assemblages were resilient even with a shift in land use from forest to grassland.

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