DRY CREEK LONG-TERM WATERSHED STUDY: ASSESSMENT OF IMMEDIATE RESPONSE OF AQUATIC MACROINVERTEBRATES TO WATERSHED LEVEL HARVESTING AND THINNING OF STREAMSIDE MANAGEMENT ZONES

M.W. Griswold, R.T. Winn, T.L. Crisman, and W.R. White[1]

Abstract—Streamside Management Zones (SMZs) are meant to protect riparian habitat and the stream ecosystem. Benthic macroinvertebrates are recognized bioindicators of water quality in streams, typically occupying multiple trophic levels in these systems and providing food for vertebrates. Thus, it is important to understand the effects of harvest within and adjacent to the SMZ on macroinvertebrate assemblages. Benthic macroinvertebrates were sampled pre- and post-harvest from four first-order streams draining the Dry Creek watershed in southwestern Georgia. A multi-habitat sampling procedure was used. Macroinvertebrates were identified and compared using biotic indices. Comparisons were made between streams within the pre-harvest period and between the pre- and post-harvest periods to determine the effects of harvest. Differences in community structure were seen between pre- and post-harvest periods. These may have been influenced by environmental factors including, but not limited to, stream flow, water chemistry, and canopy cover.

INTRODUCTION

Riparian zones are important ecotones for aquatic systems, providing food for aquatic (e.g., organic matter and terrestrial insects) and terrestrial organisms (e.g., emerging aquatic adults), shade, temperature regulation, and woody debris (Nakano and others 1999). These factors determine stream community structure (Kiffney and others 2003) and produce heterotrophic systems dependent on allochthonous detritus. Furthermore, small headwater streams are closely linked to their terrestrial surroundings since they are relatively narrow and are usually shaded by the forest canopy (Cummins 1974, Hynes 1975, Vannote and others 1980). Headwater streams can account for 70 to 80 percent of total stream length and provide downstream areas with organic matter (OM), sediments, and nutrients (Kiffney and others 2003). Thus, it is important to understand how forest management strategies influence stream biota.

Forest harvest and removal of vegetation in the riparian zone reduce detrital input to streams. The extent of this reduction is influenced by the remaining canopy cover within the riparian buffer zones. Decreased canopy cover leads to increased light availability for stream biota and may increase primary productivity, thus changing typically heterotrophic forested streams into autotrophic ones (Fuchs and others 2003, Hartman and Scrivener 1990) dominated by macrophytes and periphyton (Kedzierski and Smock 2001, Noel and others 1986). This results in increased density, biomass and diversity of macroinvertebrates and can shift macroinvertebrate trophic structure from shredders to grazers (Fuchs and others 2003, Jackson and others 2001, Kedzierski and Smock 2001). The shift in habitat structure will result in changed patch quality as the availability of leaf packs decreases and cover in the form of macrophytes increases. Noel and others (1986) found that 50 percent of logged streams were covered by macrophytes, while unlogged reference streams only had 10 percent macrophyte cover. Instream habitat cover depends greatly on inputs and cover provided by the riparian zone. Thus, subtle changes in the composition and amount of riparian cover will influence available habitat for invertebrates. The objective of the study was to determine the immediate impacts of upland and streamside management zone (SMZ) harvest on aquatic macroinvertebrates and their basal resource.

MATERIALS AND METHODS

Site Description

The site is located in southwestern Georgia on International Paper's Southlands Forest within the Dry Creek watershed, which discharges to the Flint River. The study streams are first order, groundwater-influenced, low to medium gradient, and have sand-dominated substrate. In-stream habitat includes coarse woody debris, undercut banks, leaf packs, fine roots, and macrophytes. The four study watersheds (A-D) average 39 ha, 1.5 L/s average annual discharge, and 457 m channel length (Summer and others 2003). Watersheds A and B have gentle slopes and broader, meandering/braided channels, whereas C and D have steeper slopes with well-defined stream channels.

Study Design

The overall Dry Creek Study design includes elements of before versus after, control versus impacted (treatment), and upstream versus downstream comparisons. Watersheds A and D were designated controls and were left undisturbed throughout the study. Watersheds B and C were designated treatment watersheds, each receiving two silvicultural treatments. Treatment watersheds were harvested during fall 2003, leaving a SMZ. The SMZ along the upstream portion was left intact, while the downstream SMZ was partially harvested to reduce canopy cover by 50 percent. All harvesting followed Georgia BMPs (Georgia Environmental Protection Division 1999).

[1] Graduate Student, Howard T. Odum Center for Wetlands, University of Florida, Gainesville, FL, 32611; Partnerships Coordinator, International Paper, Memphis, TN, 38115; and Director and Graduate Student, respectively, Howard T. Odum Center for Wetlands, University of Florida, Gainesville, FL 32611.

Citation for proceedings: Connor, Kristina F., ed. 2006. Proceedings of the 13th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS–92. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 640 p.

Habitat Measurements

Eight 30-m fixed-distance sample stream reaches, two per watershed, were established. Three transects were established perpendicular to the stream within each reach for physical measurements including channel cross-sections, canopy cover, and percent cover of in-stream habitat. These habitat characterization data are not discussed in this paper but will be used in the later analysis.

Biological Measurements

Within each stream reach, 10 randomly selected locations were sampled for periphyton and macrophytes from June, 2001, to December, 2004, using a 0.25 m^2 quadrat. Following the method of Tett and others (1978), two petri dishes (17.34 cm^2) were inserted into the sediment at each sampling location. Chlorophyll a concentrations of periphyton in the sediment sample were measured using an ethanol extraction procedure (Sartory and Grobbelaar 1984) followed by spectrophotometric analysis. The contents of the second petri dish were dried at 60 °C, weighed, ashed at 500 °C, and reweighed for ash-free dry weight determination of instream organic matter. Macrophytes were sampled by cutting all vegetation at the sediment surface within a 0.25 m^2 quadrat. Macrophyte samples were rinsed and dried at 60 °C (Kedierski and Smock 2001) to determine dry weight.

Benthic macroinvertebrates were collected within sample reaches using a 500-µm-mesh D-frame net (0.3 m wide) every December and February from 2001 to 2005. A multi-habitat sampling procedure was used (Barbour and others 1999). Winter was selected because this is prior to the emergence of most species, and larvae are generally easier to identify because of their larger size. Within each reach, 20 sampling sweeps (i.e., disturbing habitat for 0.5 m, approximately 3.1 m^2) were made through major habitat types including sand, woody debris, fine roots, macrophytes, and leaf packs. The duration of sampling in each reach was timed to maintain a consistent sampling effort for all reaches. Samples were stored in 90 percent ethanol and transported to the laboratory where they were processed by washing organic debris (leaves and woody debris) with water into a 500-µm-mesh sieve. Invertebrates were sorted and identified, typically to genus, although *Chironomidae* were separated into *Tanypodinae* (predators) and non-*Tanypodinae* (collectors) for functional feeding group analysis.

RESULTS AND DISCUSSION

It is accepted that harvest will decrease leaf litter input to streams (Webster and Waide 1982) and will decrease canopy cover. However, there may be delayed reduction of leaf litter in the stream due to initial runoff from harvest and retention within the substrate. In this study, instream particulate organic matter remained similar between treatment and control watersheds for up to 1 year post harvest. A peak occurred the following autumn in the control but not in the treatment watersheds (fig. 1), confirming a reduction in detrital input. Furthermore, the open canopy allowed more light to reach the stream, resulting in increased primary productivity. Both macrophyte and periphyton biomass increased in the treatment watersheds after the harvest (fig. 2). Periphyton biomass only increased in the downstream portions of the watersheds where partial harvest had occurred. This suggests that more light is reaching the stream in the partial harvest SMZs. As noted elsewhere,

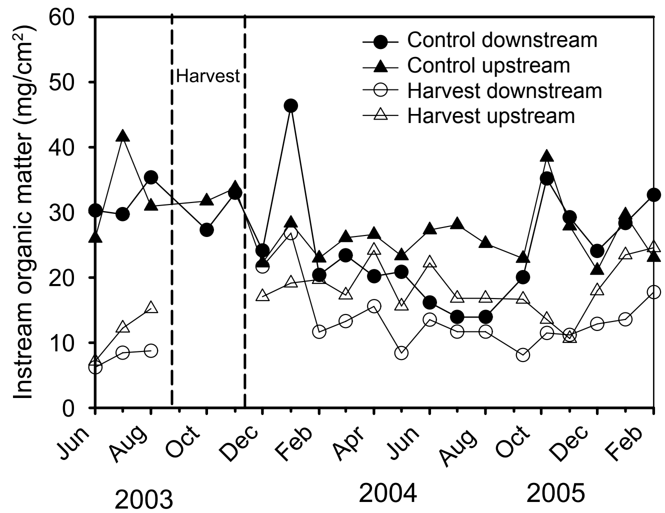

Figure 1—Average instream particulate organic matter (AFDW) before and after harvest. In harvested watersheds, the upstream SMZ was left intact and partial harvest occurred downstream.

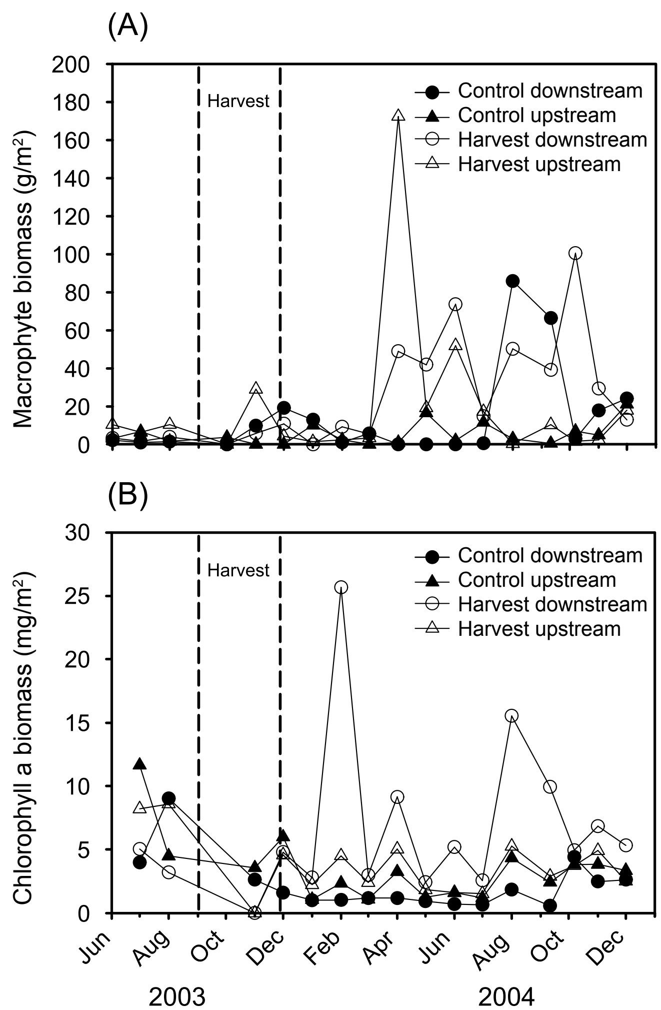

Figure 2—Average biomass of macrophytes (A) and periphyton (B) before and after harvest.

harvest is progressively shifting the stream from a heterotrophic to an autotrophic system (Fuchs and others 2003) even in the presence of the prescribed SMZ width.

The response of macroinvertebrates to harvest was more subtle and may lag behind the shift from a heterotrophic to autotrophic stream. Relative abundance and total taxa all increased in both the control and treatment sites postharvest (fig. 3). Such increases possibly reflect a recovery from drought in the region, as rainfall increased during the study period. However, Kreutzweiser and others (2005) found that removing 29 or 42 percent basal area from the riparian zone did not result in differences in macroinvertebrate abundance or richness immediately after harvest when compared to a reference site. Sedimentation may eliminate sensitive taxa; however, taxa richness may not change immediately after harvest if sedimentation is controlled. Stone and Wallace (1998) suggested that taxa richness may only be a useful metric for detecting disturbances such as organic pollution rather than shifts in relative abundance. Furthermore, any changes in richness may need to be examined on a longer time scale, as colonization by new species may not occur until the following spring and summer.

The most obvious change following harvest was the increased percentage of scraper insects in the treatment watersheds (fig. 4). Opening of the canopy and possible input of nutrients and/or detritus immediately following harvest created conditions for increased productivity. As elsewhere (Gurtz and Wallace 1984, Hawkins and others 1982, Wallace and Gurtz 1986), mayflies were the principal group displaying a response immediately after harvest. The mayflies encountered in this study (*Habrophlebiodes* spp.) feed primarily on periphyton (e.g., diatoms) and responded to increased levels of this resource. Wallace and Gurtz (1986) attributed such an increase to short generation time and high fecundity. However, samples in the current study were collected shortly after harvest (2 to 4 months), and increased abundance may have been due to resource tracking by this species. Benstead and Pringle (2004) found that mayflies are one of the few macroinvertebrate groups able to switch diets from terrestrial to algal carbon sources following forest harvesting.

Although shredders were expected to decrease with harvest, there were no differences between the treatment and reference sites (11 versus 9 percent, $P > 0.05$). Shredders may not be a good bioindicator in low-gradient sandy streams as their abundance is relatively low (Kedzierski and Smock 2001). They comprised at most 13 percent of total invertebrates in any stream in this study. Furthermore, instream particulate organic matter did not decline immediately following harvest. Leaf litter was retained within the substrate and may have been trapped in debris dams, providing habitat and food for shredders. However, many stream invertebrates may have relatively rigid diets (Benstead and Pringle 2004), limiting their distribution with decreased input of terrestrial carbon sources. Thus, as streams become more autotrophic, species diversity of specialists depending on allochthonous material, including collectors, is expected to decrease.

Riparian cover is an important factor affecting stream communities. It is expected that changes in cover and amount of vegetation will have profound effects on community structure. This study reports results from the first 2 to 4 months after

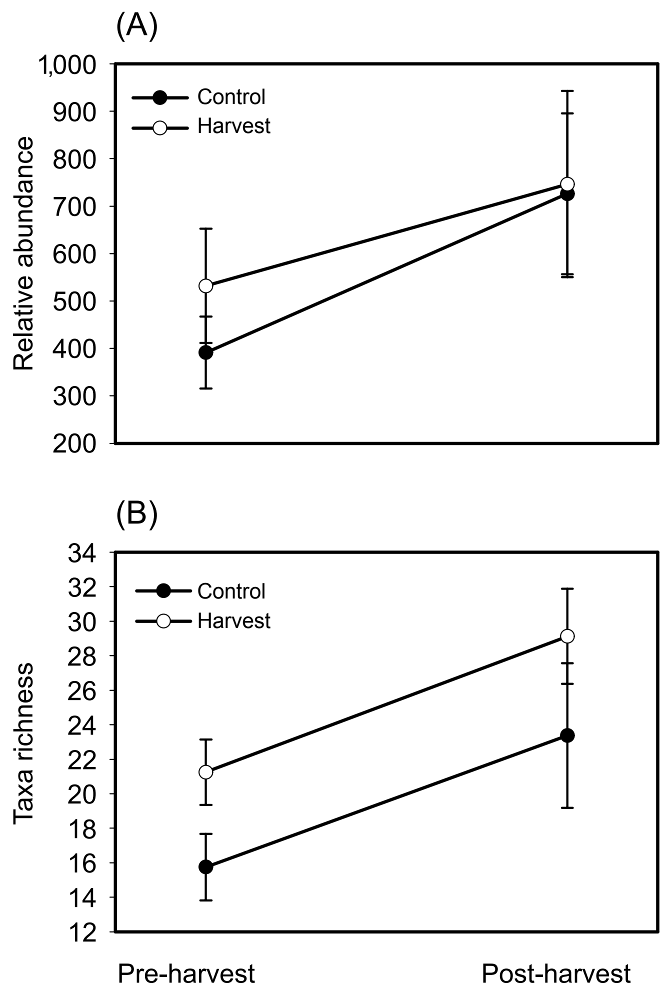

Figure 3—Mean relative abundance (per 20 dipnet samples) (A) and taxa richness (number of taxa) (B) pre- and post-harvest. Bars are ± 1 standard error. Data were combined for all dates before and after harvest.

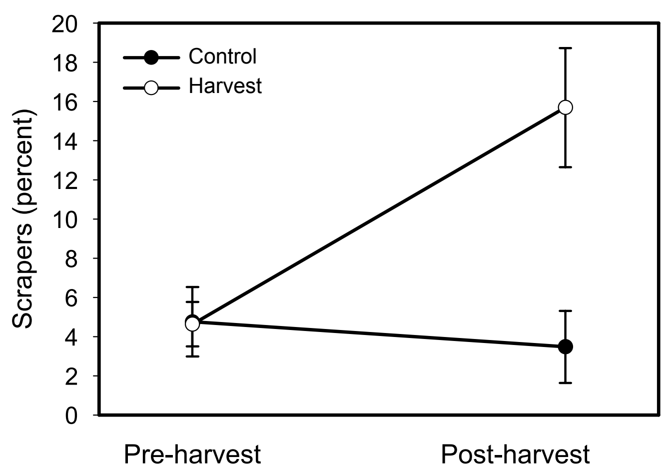

Figure 4—Mean percent of scrapers pre- and post-harvest. Bars are ± 1 standard error. Data were combined for all dates before and after harvest.

harvest, when community structure of invertebrates and the resource base are likely still in flux. Although there were no differences in intact and partially harvested SMZs, the abiotic and biotic structure of the stream are changing. Increased macrophytes in harvested watersheds are continuing and are expected to lead to changes in the macroinvertebrate community, further dividing the intact SMZ from the SMZ that was partially harvested.

ACKNOWLEDGMENTS

We thank the National Council for Air and Stream Improvement, National Fish and Wildlife Foundation, Howard T. Odum Center for Wetlands, and International Paper for funding and support of this project. Thanks goes to William B. Summer for assistance with instream habitat assessment. Previous versions of this manuscript were improved through comments from W. Wise and M. Miwa.

LITERATURE CITED

Barbour, M.T.; Gerritsen, J.; Snyder, B.D.; Stribling, J.B. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish. 2d ed. EPA 841-B-99-002. Washington, DC: U.S. Environmental Protection Agency, Office of Water. 339 p.

Benstead, J.P.; Pringle, C.M. 2004. Deforestation alters the resource base and biomass of endemic stream insects in eastern Madagascar. Freshwater Biology. 49: 490–501.

Cummins, K.W. 1974. Structure and function of stream ecosystems. Bioscience. 24: 631-641.

Fuchs, S.A.; Hinch, S.G.; Mellina, E. 2003. Effects of streamside logging on stream macroinvertebrate communities and habitat in sub-boreal forests in British Columbia, Canada. Canadian Journal of Forest Research. 33: 1408-1415.

Georgia Environmental Protection Division. 1999. Georgia's best management practices for forestry. 71 p.

Gurtz, M.E.; Wallace, J.B. 1984. Substrate-mediated response of stream invertebrates to disturbance. Ecology. 65: 1556-1569.

Hartman, G.F.; Scrivener, J.C. 1990. Impacts of forestry practices on a coastal stream ecosystem, Carnation Creek, British Columbia. Canadian Bulletin of Fisheries and Aquatic Sciences. 223: 1-148.

Hawkins, C.P.; Murphy, M.L.; Anderson, N.H. 1982. Effect of canopy, substrate composition, and gradient on the structure of macroinvertebrate communities in Cascade Range streams of Oregon. Ecology. 63: 1840-1856.

Hynes, H.B.N. 1975. The stream and its valley. Verhandlungen. Internationale Vereinigung fuer Theoretische und Angewandte Limnologie. 19: 1-15.

Jackson, C.R.; Sturm, C.A.; Ward, J.M. 2001. Timber harvest impacts on small headwater stream channels in the coast ranges of Washington. Journal of the American Water Resources Association. 37: 1533-1549.

Kedzierski, W.M.; Smock, L.A. 2001. Effects of logging on macroinvertebrate production in a sand-bottomed, low-gradient stream. Freshwater Biology. 46: 821-833.

Kiffney, P.M.; Richardson, J.S.; Bull, J.P. 2003. Responses of periphyton and insects to experimental manipulation of riparian buffer width along forest streams. Journal of Applied Ecology. 40: 1060-1076.

Kreutzweiser, D.P.; Capell, S.S.; Good, K.P. 2005. Effects of fine sediment inputs from a logging road on stream insect communities: a large-scale experimental approach in a Canadian headwater stream. Aquatic Ecology. 39: 55-66.

Nakano, S.; Miyasaka, H.; Kuhara, N. 1999. Terrestrial-aquatic linkages: riparian arthropod inputs alter trophic cascades in a stream food web. Ecology. 80: 2435-2441.

Noel, D.S.; Martin, C.W.; Federer, C.A. 1986. Effects of forest clearcutting in New-England on stream macroinvertebrates and Periphyton. Environmental Management. 10: 661-670.

Sartory, D.P.; Grobbelaar, J.U. 1984. Extraction of chlorophyll a from freshwater phytoplankton for spectrophotometric analysis. Hydrobiologia. 114: 177-187.

Stone, M.K.; Wallace, J.B. 1998. Long-term recovery of a mountain stream from clearcut logging: the effects of forest succession on benthic invertebrate community structure. Freshwater Biology. 139: 151-169.

Summer, W.B.; Jackson, C.R.; Jones, D.; Miwa, M. 2003. Characterization of hydrologic and sediment transport behavior of forested headwater streams in southwest Georgia. In: Proceedings of the 2003 Georgia water resources conference. Athens, GA: The University of Georgia, The Institute of Ecology. [Not paged].

Tett, P.; Gallegos, C.; Kelly, M.G. [and others]. 1978. Relationships among substrate, flow, and benthic microalgal pigment density in Mechums River, Virginia. Limnology and Oceanography. 23: 785-797.

Vannote, R.L.; Minshall, G.W.; Cummins, K.W. [and others]. 1980. River continuum concept. Canadian Journal of Fisheries and Aquatic Sciences. 37: 130-137.

Wallace, J.B.; Gurtz, M.E. 1986. Response of *Baetis* mayflies (Ephemeroptera) to catchment logging. American Naturalist. 115: 25-41.

Webster, J.R.; Waide, J.B. 1982. Effects of forest clearcutting on leaf breakdown in a southern Appalachian stream. Freshwater Biology. 12: 331–344.