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DRY CREEK LONG-TERM WATERSHED STUDY: BUFFER ZONE PERFORMANCE AS VIABLE AMPHIBIAN HABITAT

Brooke L. Talley and Thomas L. Crisman¹

Abstract—As bioindicators, amphibians typically require both terrestrial and aquatic habitats to complete their life cycles. Pre-timber-harvest monitoring (December 2002 through September 2003) of salamander and frog (*Hylidae*) populations was conducted in four watersheds of Decatur County, GA. Post-timber-harvest monitoring (December 2003 through September 2004) continued in the same watersheds (two reference and two treatment watersheds). Coverboards were used to monitor adult salamanders, larval salamanders were surveyed with dipnet sweeps, and frogs were monitored with vertical PVC pipes. Six salamander species (*Desmognathus apalachicola*, *Eurycea cirrigera*, *E. guttolineata*, *Notophthalmus viridescens*, *Plethodon grobmani*, and *Pseudotriton ruber*) and five frog species (*Hyla chrysoscelis*, *H. cinerea*, *H. femoralis*, *H. squirella*, and *Pseudacris crucifer*) were recorded. Pre- and post-harvest data were compared to assess changes in amphibian population structure. Larval salamanders were the only amphibian group to decline 1 year after timber-harvest. Further long-term examination of amphibian populations should be continued to assess potential delayed responses to site disturbance.

INTRODUCTION

Timber- and partial-harvest in uplands and within Streamside Management Zones (SMZs) may cause significant aquatic and terrestrial habitat alteration. Alterations include, but are not limited to, stream hydrology and water quality. Physiological adaptations of many amphibians make them vulnerable to ecosystem stress following perturbation because of specific aquatic and terrestrial microhabitat requirements (Welsh and Ollivier 1998).

In previous studies, increased sedimentation from land management activities (including timber-harvest) led to reduced amphibian abundance (Welsh and Ollivier 1998). In the southern Appalachians, Petranka and others (1993) found that adult terrestrial salamander abundance declined in clear-cut plots compared to mature forest stands. In general, stream-dwelling organisms like macroinvertebrates and fishes have been more frequently studied for timber-harvest response compared to amphibians (Welsh and Ollivier 1998). Natural variation in magnitude and frequency for amphibian populations can make it difficult to identify fluctuation causes, including timber-harvest (Blaustein and others 1994, Pechmann and Wilbur 1994, Stebbins and Cohen 1995).

This study examines how interannual amphibian populations fluctuate in intact and harvested watersheds. As biological indicators of aquatic and terrestrial environments, salamander and frog (*Hylidae*) responses following timber harvest can provide valuable information on how Georgia Best Management Practices (BMPs) may affect biotic structure within these watersheds.

Study Site

Southlands Experimental Forest of International Paper occurs within the Coastal Plain physiographic province, in Decatur County, GA (30°47'30" N and 84°37'30" W), approximately 16 km south of Bainbridge, GA (fig. 1). First-order perennial streams draining four neighboring watersheds (termed A, B, C, and D) were studied (fig. 1).

Located in the Dry Creek watershed, study streams flow into Dry Creek (a second-order stream) and eventually into the Flint River (upstream from the Apalachicola River). In-stream habitat composition included coarse woody debris, undercut banks, leaf packs, fine roots, and pools. Streams were ground-water-fed with sand-dominated substrates. Of deeper incision,

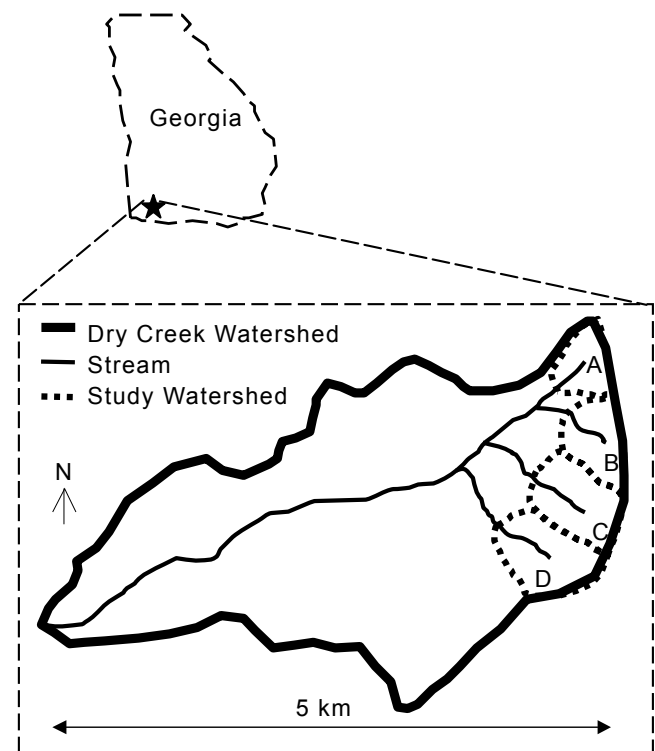


Figure 1—Dry Creek watershed location and experimental watershed layout (30° 47'30" N and 84°37'30" W). Watersheds A, B, C, and D are composed of headwater streams and mixed hardwood forests. Watersheds A and D were not harvested, while watersheds B and C were clearcut from September through November, 2003, according to Georgia Best Management Practices.

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streams C and D channels were adjacent to steeper slopes than streams A and B (Jones and others 2003, Sumner and others 2003) (fig. 1).

METHODS

Two reaches per watershed were studied (1 = downstream reach, 2 = upstream reach), for a total of eight sample reaches (A1, A2, B1, B2, C1, C2, D1, and D2). Multi-year data were collected to examine amphibian seasonal differences (Pough and others 1998) and interannual differences in natural populations, with monitoring at monthly intervals. Pre-harvest sampling occurred over 10 months (December 2002 through September 2003) in all streams. Watersheds B and C were harvested following sampling in September, 2003, a process that lasted 3 months. Post-harvest data collection resumed in December, 2003, and continued through September, 2004. Sampling techniques employed to capture amphibians included dipnet sweeps (for larvae salamanders), coverboard shelter attractants (for adult salamanders), and vertical PVC pipe shelter attractants (for frogs).

To sample all potential microhabitats within the stream, the flat surface of a standard D-frame dipnet ($V \cong 0.02 \text{ m}^3$; dimensions: 0.3-m opening, 0.5-m length, 1,000- μm mesh) was swept along the bottom of the stream and under incised banks. For each sample reach, 20 dipnet sweeps were performed, each approximately 1-m long. Dipnet sampling occurred upstream from stationary hydrologic flumes. Captured larvae were counted, identified to species, and released into the stream reach where captured.

Coverboards were used as shelter attractants for terrestrial and semi-aquatic salamanders by mimicking conditions found under naturally occurring surface objects (Houze and Chandler 2002). In this study, coverboards were used to assess adult salamander species richness and dispersal distance into surrounding uplands. Coverboards, cut from 1.9 cm untreated plywood sheets into 60 by 60-cm squares, were placed along transects perpendicular to stream channels toward adjacent uplands. Eight coverboards were placed in designated habitat zones for a given sample reach (4 coverboards on either side of the stream, 256 total). The four habitat zones were designated as (1) streamside, (2) riparian, (3) midslope, and (4) upland, with increasing distance from the stream. Salamanders found under coverboards were identified to species and counted, noting specific coverboard position.

Vertical polyvinyl chloride (PVC) pipes (5.1-cm diameter and 60-cm height-above-ground) were used for frog monitoring. PVC pipes act as shelter attractants by shielding inhabitants from extreme wind and temperature, thereby providing moist refuge (Wyatt and Forsys 2004). One sampling pipe was installed at each coverboard location (256 total pipes). Frogs inhabiting the artificial habitat were identified to species, counted, and specific PVC pipe was noted.

Statistical analyses utilized Jandel SigmaStat 2.0[®]. Catch per Unit Effort (CPUE) data were analyzed with normality, equal variance, Mann-Whitney Rank Sum, and t-tests. All statistical analyses were considered significant with $\alpha = 0.05$.

RESULTS AND DISCUSSION

Amphibian species richness data were collected monthly during pre- and post-harvest surveys. Although species richness varied between watersheds and years, this was not analyzed for any effect due to timber-harvest. Because capture data were not adjusted for detection probabilities, amphibian abundance could not be estimated (see Dodd and Dorazio 2004, Schmidt 2003, 2004). To make capture data comparable for pre- and post-harvest surveys, CPUE values were calculated based on the number of individuals captured per number of experimental units. Larval salamander CPUE values were calculated by dividing total capture by 1,600 (160 sweeps per month by 10 months), except for months when site conditions prohibited data collection. For adult salamander and frog surveys, CPUE values were determined by dividing capture values by 640 (64 coverboards/PVC pipes per habitat zone by 10 months). All amphibian Catch per Unit Effort (CPUE) data were tested for normal distribution using the normality test ($\alpha = 0.05$) to determine further statistical analyses required.

Larval Salamanders

Two larval salamander species were detected, *Eurycea cirrigera* and *Pseudotriton ruber*. When calculating CPUE values, larval species capture-data were combined to examine overall trends in population dynamics instead of specific species patterns.

CPUE distributions passed the normality test (reference streams: $P = 0.129$, treatment streams: $P = 0.444$) and the test of equal variance for normally distributed populations in treatment streams ($P = 0.246$) but not reference streams ($P = 0.010$). Because CPUE distributions failed the test of equal variance for reference streams, Mann-Whitney Rank Sum Tests were performed. Statistical analyses showed no significant median value differences between pre- and post-harvest larval salamander populations in reference streams ($P = 0.734$) (fig. 2a). Results of t-test statistical analyses for treatment streams showed significant differences between CPUE values for pre- and post-harvest larval salamanders ($P = 0.032$) (fig. 2b).

Because larval salamander CPUE values in reference watersheds were not significantly different between sampled years, the differences detected in treatment watersheds were probably not due to natural variation. Instead of reflecting true timber-harvest effects, differences between CPUE values may be in response to other abiotic differences, such as temperature (Lucas and Reynolds 1967). Typically, early stages of amphibian development (i.e., larvae) are more sensitive to temperature changes than in later stages (Stebbins and Cohen 1995). Temperature change could have been caused by timber harvest (from tree canopy changes), but this was not examined in this study.

Although larval amphibian populations fluctuate between years and seasons, the resultant change in treatment watersheds was not apparent in reference watersheds. Because all four watersheds are in close proximity, abiotic factors that could potentially affect amphibian populations (e.g., temperature, rainfall) should be similar under natural conditions. Therefore, the difference detected after timber-harvest was likely a reflection of site-disturbance. Potential changes in abiotic factors of treatment watersheds should be examined further for their relationship to larval salamander populations.

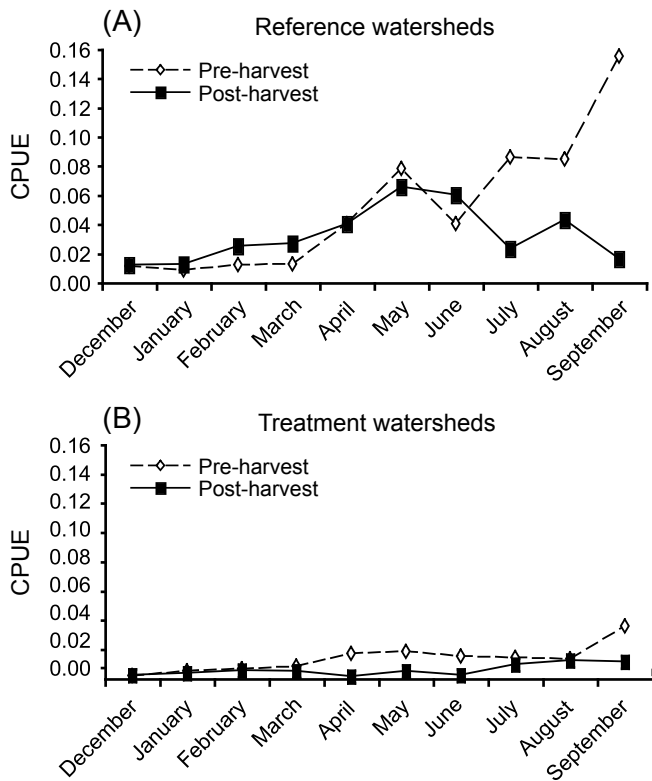


Figure 2—Larval salamander CPUE for (A) reference and (B) treatment watersheds. CPUE values are based on number of individuals captured with dipnet sweeps. Pre- (December 2002 through September 2003) and post-harvest (December 2003 through September 2004) CPUE values indicate differences in treatment watersheds. This statistical difference is not seen in reference watersheds. CPUE = catch per unit effort.

Adult Salamanders

Adult salamander species richness was comprised of six species: *Desmognathus apalachicola*, *Eurycea cirrigera*, *E. guttolineata*, *Notophthalmus viridescens*, *Plethodon grobmani*, and *Pseudotriton ruber*. Both *E. cirrigera* and *Plethodon grobmani* were detected throughout all watersheds during pre- and post-harvest surveys. The presence of other salamanders varied between sampling years and watersheds. To further examine how adult salamander population dynamics responded to timber harvest, CPUE values combined capture data of all adult salamander species.

In reference and treatment watersheds, CPUE distributions passed the normality test (reference watersheds: $P = 0.374$, treatment streams: $P = 0.551$) and the test of equal variance for normally distributed populations (reference: $P = 0.882$, treatment: $P = 0.684$). Results of t-test statistical analyses showed no significant differences between CPUE values for pre- and post-harvest adult salamanders in all four watersheds (reference: $P = 0.579$, treatment: $P = 0.931$) (fig. 3).

In both reference and treatment watersheds, adult salamander CPUE exhibited no significant change from the first year of sampling to the second. However, the population increased in reference watersheds from the first year of sampling to the second (fig. 3a); this was not detected in treatment watersheds (fig. 3b). Instead, the latter decreased in CPUE values from pre- to post-harvest surveys. In pre- and post-harvest surveys,

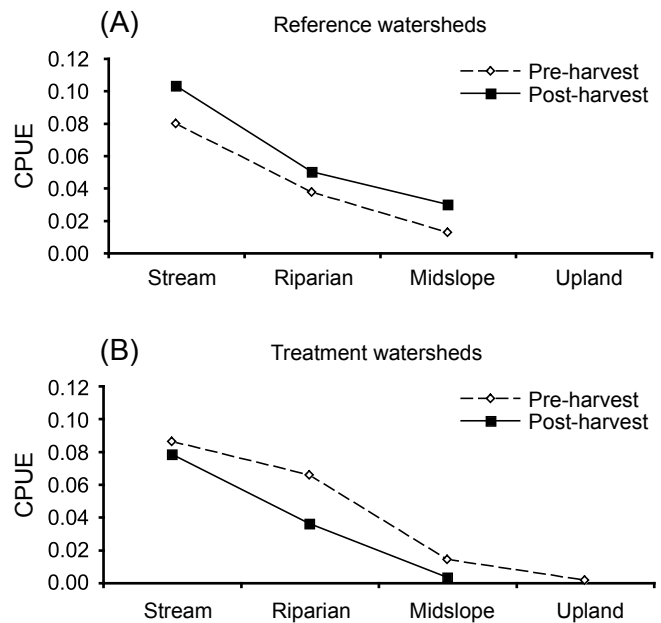


Figure 3—Adult salamander CPUE for (A) reference and (B) treatment watersheds. CPUE values are based on number of individuals captured beneath coverboards at each habitat zone. Pre- (December 2002 through September 2003) and post-harvest (December 2003 through September 2004) CPUE values are not statistically different. Treatment watershed CPUE values are lower after timber harvest, while experimental watershed CPUE values increase during the second survey year. CPUE = catch per unit effort.

salamanders preferred streamside habitat zones compared to those farther upland, regardless of timber-harvest treatment.

Because these watersheds are similar in morphology and located within the same larger watershed system (Jones and others 2003, Summer and others 2003), population dynamics would be comparable. Since CPUE trends differ between reference and treatment watersheds, overall interpretation should not be based only on statistical results. Adult salamanders can display delayed responses to site disturbance and other abiotic changes. Therefore, long-term examination in population structure should be continued.

Frogs (*Hylidae*)

Five hylid frog species (*Hyla chrysoscelis*, *H. cinerea*, *H. femoralis*, *H. squirella*, *Pseudacris crucifer*) were detected. All frog capture values were used for CPUE determination to monitor changes in population dynamics.

For both reference and treatment watersheds, CPUE distributions were normal (reference watersheds: $P = 0.652$, treatment watersheds: $P = 0.725$) but failed the test of equal variance (reference: $P = 0.002$, treatment: $P = 0.04$). Because of this failure, Mann-Whitney Rank Sum Tests were performed and showed no significant median value differences between pre- and post-harvest frog populations in reference and treatment watersheds (reference: $P = 0.114$, treatment: $P = 0.886$) (fig. 4).

Evaluating frog population dynamics through CPUE values indicated no differences between pre- and post-harvest surveys. In general, CPUE values showed a dramatic increase in reference watersheds for all habitat zones (fig. 4a). Although CPUE values increased in treatment watershed stream and

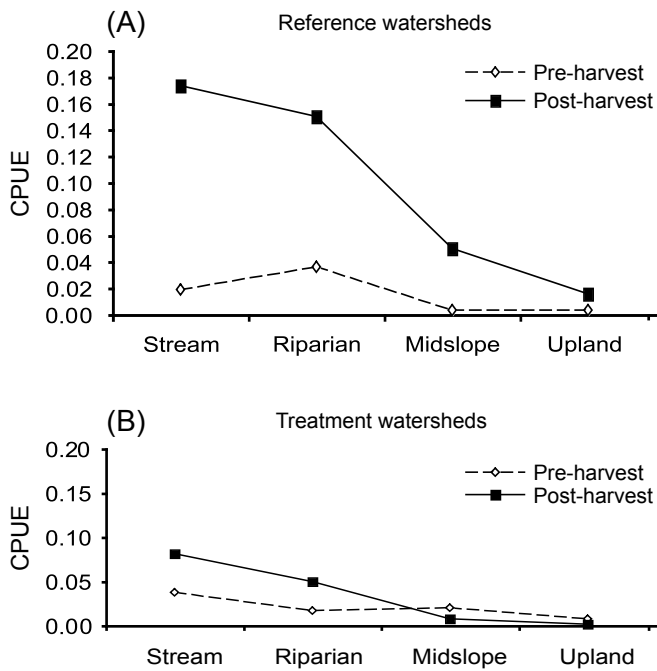


Figure 4—Frog CPUE for (A) reference and (B) treatment watersheds. CPUE values are based on number of individuals found within vertical PVC pipes. Pre- (December 2002 through September 2003) and post-harvest (December 2003 through September 2004) CPUE values showed no significant difference between survey years in reference and treatment watersheds. CPUE = catch per unit effort.

riparian habitat zones during post-harvest surveys, overall frog CPUE did not respond with the degree of change seen in reference watersheds.

Frog activity and PVC pipe inhabitation changes seasonally (Zacharow and others 2003). Because PVC pipe sampling techniques are a relatively new sampling device, changes in frog CPUE values could be influenced by sampling technique effectiveness. Therefore, data interpretation should be conservatively analyzed regarding population dynamics.

CONCLUSIONS

Larval salamanders in treatment streams displayed the only amphibian population change after timber-harvest. The effect of timber-harvest was likely reflected in larval salamander populations because they live within streams, where site-disturbance changes are likely to occur quickly. Although adult salamander and frog populations did not change significantly after timber-harvest, overall changes in populations may not respond to site disturbance with as much immediacy as larval salamander populations. Therefore, future studies should examine long-term effects of timber-harvest on amphibian populations, information which could help predict degree of viability remaining in harvested habitat.

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