

## The Influence of *Lepidostoma* (Trichoptera: Lepidostomatidae) on Recovery of Leaf-litter Processing in Disturbed Headwater Streams

MATT R. WHILES, J. BRUCE WALLACE AND KEUN CHUNG

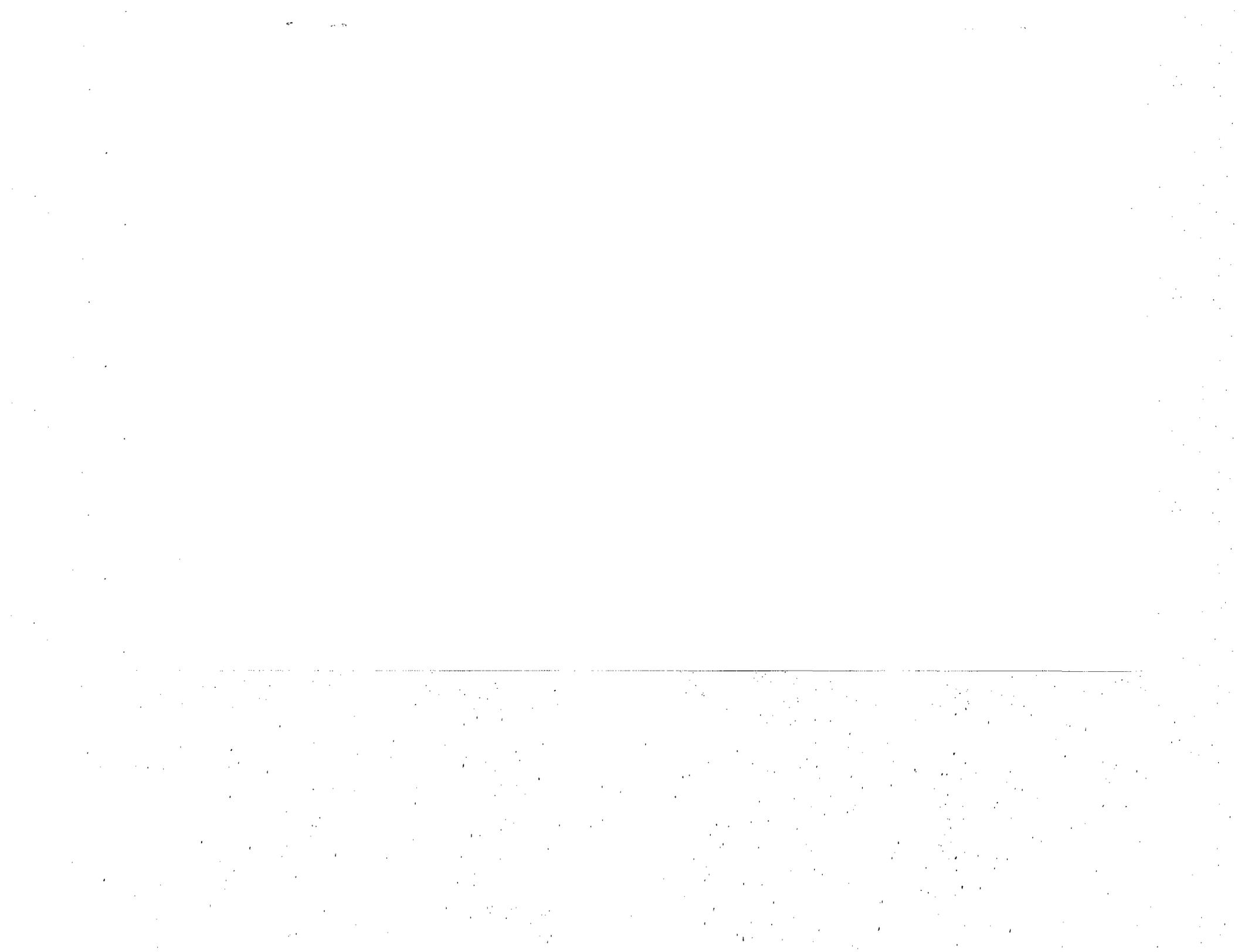
*University of Georgia, Department of Entomology, Bioscience Building, Athens 30602*

**ABSTRACT.**—Two headwater streams draining catchments 53 and 54 (C53 and C54, respectively) at the Coweeta Hydrologic Laboratory in western North Carolina were treated with insecticide in 1980 (C53) and 1986-1988 (C54). During recovery periods in both streams, *Lepidostoma* spp. were abundant, early colonizers. Densities of *Lepidostoma* in litterbags and benthic samples collected from recovering streams were substantially higher than in untreated streams (pretreatment and reference streams). During treatment years, leaf-litter processing rates were severely reduced relative to pretreatment and reference streams. In contrast, litter processing rates in C53 and C54 during recovery were faster than those in untreated streams (pretreatment and reference). Rhododendron is one of the most refractory leaves commonly found in Coweeta streams; however, percent increase of rhododendron processing rates from treatment periods to recovery was greater than that of more labile red maple. Laboratory feeding experiments were performed in order to examine and quantify use of rhododendron and red maple litter by *Lepidostoma* larvae. In the laboratory, *Lepidostoma* consumed significantly more rhododendron (1.062 mg AFDM/mg AFDM body wt/day) than red maple (0.479 mg AFDM/mg AFDM body wt/day) ( $P = 0.001$ ). Results suggest that changes in the taxonomic composition of stream macroinvertebrate communities due to disturbance may have significant effects on ecosystem processes for at least 2 yr after termination of disturbance. *Lepidostoma* colonize rapidly following disturbance, enhancing the restoration of vital ecosystem processes such as litter decomposition in Coweeta streams.

### INTRODUCTION

Allochthonous leaf litter inputs are the primary energy source in forested headwater streams (Cummins *et al.*, 1983; Webster *et al.*, 1983). This coarse particulate organic matter (CPOM) is generally retained until it is converted into fine particles (FPOM) (Naiman and Sedell, 1979; Wallace *et al.*, 1982a; Cummins *et al.*, 1983) and/or respired via microbial activity (Cuffney *et al.*, 1990). Feeding activities of invertebrate communities, particularly shredders, are important in conversion of CPOM into FPOM (Cummins, 1974; Cummins and Klug, 1979; Cuffney *et al.*, 1990). Shredders typically have low assimilation efficiencies (Fisher and Likens, 1973) coupled with high ingestion rates (Cummins, 1973; Golladay *et al.*, 1983; Webster, 1983), enhancing their importance in conversion of CPOM to FPOM. Fine particles generated by the feeding activities of shredders are easily entrained and used by other invertebrate groups (*e.g.*, collector filterers and gatherers) (Short and Maslin, 1977; Anderson and Sedell, 1979; Grafius and Anderson, 1979).

Experimental reduction of shredder densities in headwater streams has been shown to reduce rates of litter breakdown (Wallace *et al.*, 1982b; Cuffney *et al.*, 1990) and generation of FPOM (Cuffney *et al.*, 1990). Recovery of these processes is linked to recovery of shredder and other macroinvertebrate densities (Wallace *et al.*, 1986; Wallace *et al.*, 1991). Thus, recovery of macroinvertebrate communities, particularly shredders, following disturbance restores a vital component of energy flow, and a link between transport and storage in forested headwater streams.



Rates and patterns of macroinvertebrate community recovery are dependent upon physical aspects of a disturbance (*see* reviews by Yount and Niemi, 1990; Niemi *et al.*, 1990), as well as characteristics of the resident biota (*see* review by Wallace, 1990). Taxa with short generation times and high vagility are often the fastest colonizers (*e.g.*, Gray and Fisher, 1981; Fisher *et al.*, 1982; Molles, 1985). These early colonists can lead to fairly rapid functional recovery (re-establishment of functional group densities and biomass). However, taxonomic differences among invertebrate communities compared to pre-disturbance conditions may persist until taxa with poor vagility and/or longer life cycles can colonize (*e.g.*, Wallace *et al.*, 1986; Whiles and Wallace, 1992).

Wallace *et al.* (1986) observed functional recovery of the macroinvertebrate community (1 to 2 yr) before taxonomic recovery [up to 5 yr (Lugthart and Wallace, 1992)] following a 1-yr insecticide-induced reduction of macroinvertebrate densities in a headwater stream. In a similar study following a 3-yr insecticide treatment of a different stream, Whiles and Wallace (1992) observed a similar recovery pattern (functional recovery preceding taxonomic recovery). In both cases, taxa with high vagility, relatively short life cycles and/or long periods of adult presence (*e.g.*, *Lepidostoma* spp., Chironomidae) dominated the macroinvertebrate communities during recovery. *Lepidostoma lydia* Ross and *L. griseum* (Banks), univoltine caddisflies with relatively long flight periods, appeared to dominate the shredder communities in both streams during recovery.

Unusually fast litter processing rates, particularly for rhododendron (*Rhododendron maximum* L.), coincided with high densities of *Lepidostoma* (Wallace *et al.*, 1986; Whiles and Wallace, 1992). In undisturbed streams at Coweeta, rhododendron is one of the more refractory litter species (Webster and Waide, 1982; Wallace *et al.*, 1986; Cuffney *et al.*, 1990).

The main objective of this study was to examine the mechanism by which litter decomposition rates increase in Coweeta streams during recovery from insecticide treatment. Predisturbance, recovery and untreated reference stream data from two different insecticide studies were used to examine the relationship between high densities of *Lepidostoma* spp. and unusually fast leaf litter decomposition rates (particularly rhododendron) during recovery periods. In addition, a laboratory feeding experiment was performed in order to quantify use of rhododendron and more labile red maple (*Acer rubrum* L.) litter by *Lepidostoma* larvae.

#### STUDY SITE

The study streams are small headwater streams located at the Coweeta Hydrologic Laboratory (U.S. Forest Service) in the Blue Ridge Province of the southern Appalachian Mountains, North Carolina (detailed descriptions of the Coweeta basin are presented by Swank and Crossley, 1988). All three streams (Catchments 53, 54, 55 = C53, C54, C55) drain mixed hardwood forest dominated by oak (*Quercus* spp.) and hickory (*Carya* spp.), and are heavily shaded by a dense understory of rhododendron. Physical characteristics of the streams (*e.g.*, elevation, temperature, discharge) are similar (Lugthart and Wallace, 1992).

#### METHODS

*Insecticide treatments.*—During 1980, C53 was treated seasonally for 1 yr (=four treatments) with the insecticide methoxychlor [1,1,1-trichloro-2,2 bis (paramethoxyphenol) ethane], and C54 served as a reference stream. During later treatment of C54 with methoxychlor (1985–1988; seasonally for 3 yr = 12 treatments), C55 served as a reference stream. Treatments consisted of spraying the entire length of the stream (flume to spring seep) with

hand sprayers at a delivery equivalent to a concentration of 10 mg/liter (based on discharge at flume).

*Litter decomposition.*—Leaf-litter processing rates of rhododendron and red maple were measured in plastic mesh litter bags (36 × 20 cm with 5-mm mesh opening). Litter bags filled with 15 g dry mass leaf material (40–60 bags of each leaf species) were placed in study streams early in the year (Dec.–Feb.). Replicates of four or five bags of each litter species were collected during the year (1- to 2-mo pickup intervals for approx. 1 yr). Contents of each bag were washed, oven-dried, weighed, ashed, and re-weighed to obtain estimates of ash-free dry mass (AFDM) remaining on each pickup date. See Wallace *et al.* (1986) and Cuffney *et al.* (1990) for complete details on pickup schedules and processing methods. During treatment of C53, litter processing rates were measured in C53 and C54 (untreated reference). During treatment of C54, litterbags were placed in C54, C55 (untreated reference) and C53 (long-term recovery).

Annual litter processing rates (days to 5% AFDM remaining) were obtained from untreated streams [C53: 1985–1990; C54: 1980, 1982, 1985; C55: 1985–1990 (n = 15)], treated streams [C53: 1980; C54: 1986–1988 (n = 4)] and recovery periods [C53: 1982; C54: 1990 (n = 2)]. Averages and standard errors were calculated for the untreated and treatment period litter processing rates. Processing rates during recovery were based on average values of 2nd-yr recovery. Because only two recovery values were available, no standard error was calculated for the recovery value.

*Lepidostoma densities.*—Densities of *Lepidostoma* were determined in litterbags (same as described previously for litter processing) and benthic samples. Litterbags were used during both treatment studies and subsequent recovery periods (in treated and reference streams). In the laboratory, all macroinvertebrates were removed from litterbags, identified, counted, and measured (total body length in mm) under a dissecting microscope with a graduated stage. Details on processing of litterbag macroinvertebrates are described by Wallace *et al.* (1986). Annual average densities and 95% confidence intervals of *Lepidostoma* larvae in litterbags were calculated for pairwise comparison of densities (C54 vs. C55) during pretreatment, treatment and recovery. Differences were considered significant ( $P < 0.05$ ) when 95% confidence intervals did not overlap, a conservative test for difference (Zar, 1984). In order to protect for multiple comparisons, 99% confidence intervals were used when litterbag densities in C54 recovery were compared to those of both C55 reference and C54 pretreatment.

During pretreatment, treatment and recovery periods of C54 (1984–1991), benthic samples were used to estimate macroinvertebrate densities in C54 and C55 (untreated reference). Monthly (C54) and bimonthly (C55) benthic samples were taken at randomly selected locations along each stream in mixed substrate [four benthic corer samples per sampling date (400 cm<sup>2</sup> each)] and bedrock outcrop habitats [three rock scrapings per sampling date (225 cm<sup>2</sup> each)]. In the laboratory, all macroinvertebrates were removed from substrates, counted, identified and measured under a dissecting microscope with a graduated stage. Average *Lepidostoma* densities were weighted for the percent composition of bedrock outcrop and mixed substrates in each stream [(*Lepidostoma* density on bedrock outcrop × % bedrock outcrop in stream) + (*Lepidostoma* density in mixed substrates × % mixed substrates in stream)]. Whiles and Wallace (1992) give complete details on benthic sampling procedures.

Averages and 95% confidence intervals of benthic *Lepidostoma* densities were calculated for December 1988–June 1989, July 1989–December 1989, and January–April 1990. Densities from individual samples were log (log X + 1) transformed prior to calculations, and transformed back for determination of significant differences [nonoverlapping 95% confidence intervals ( $P < 0.05$ )] (Zar, 1984).

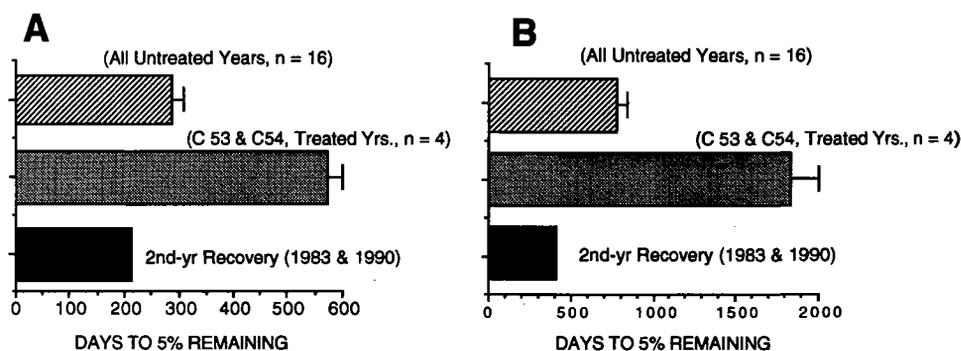


FIG. 1.—Average decomposition rates (days to 95% loss  $\pm$  SE) in untreated streams ( $n = 16$  annual litterbag series), treated streams ( $n = 4$  annual litterbag series), and 2nd-yr recovery streams ( $n = 2$  annual litterbag series) for (A) red maple and (B) rhododendron (note different scales)

*Laboratory feeding experiments.*—Before feeding experiments, late instar *Lepidostoma* larvae and conditioned rhododendron and red maple leaves were collected from C54. In the laboratory, leaf disks were cut from conditioned leaves with a cork borer (ca. 2 cm diam) and wet weight of disks measured. AFDM conversion factors for each leaf species were determined by drying and ashing a subsample of six disks of each leaf species.

Six disks of rhododendron (ca. 0.5–0.7 g total), red maple (ca. 0.1–0.2 g total), or three disks of each type (choice experiment) were placed in plastic mesh (250  $\mu$ m mesh opening) containers (ca. 100 cc) with five *Lepidostoma* larvae. Containers with leaf disks and no larvae were used as controls. Containers were randomly placed in 11.5  $\times$  6.5  $\times$  3.5 cm plastic boxes filled with C54 stream water (ca. 2.5 cm deep) and aerated with air stones. Boxes were maintained in an incubator at 12 C for 10 days.

After 10 days, larvae and leaf disks were removed for measurements. Wet weight of leaf material remaining was determined and converted to AFDM using the previously determined AFDM conversion factors. Lengths of *Lepidostoma* larvae (total body length in mm) were measured using a dissecting microscope with a graduated stage. AFDM of larvae in each chamber was estimated using length-weight regressions constructed with larvae from the same stream. Feeding rates (mg AFDM leaf material removed/mg AFDM body weight/day) were estimated for each container. For choice experiments (rhododendron and red maple in same container), feeding rates were determined for each leaf type separately.

Feeding experiments (10 days each) were performed on 17 March, 8 April and 24 September 1991. Before each experiment, new *Lepidostoma* larvae and conditioned leaf material were collected from C54. AFDM conversions for each litter type were calculated before each experiment. A total of 10 replicate feeding rates were collected for each of the following: rhododendron only, maple only, and choice between rhododendron and red maple. Feeding rate data were analyzed for statistical differences (rhododendron vs. red maple) by analysis of variance (ANOVA).

## RESULTS

*Litter decomposition.*—During treatment of streams, rhododendron and red maple processing rates (average days to 95% loss) were substantially slower (1832 and 573 days, respectively) than in untreated periods (778 and 283 days, respectively) (Fig. 1). Average processing rates for both rhododendron (407 days) and red maple (215 days) were faster

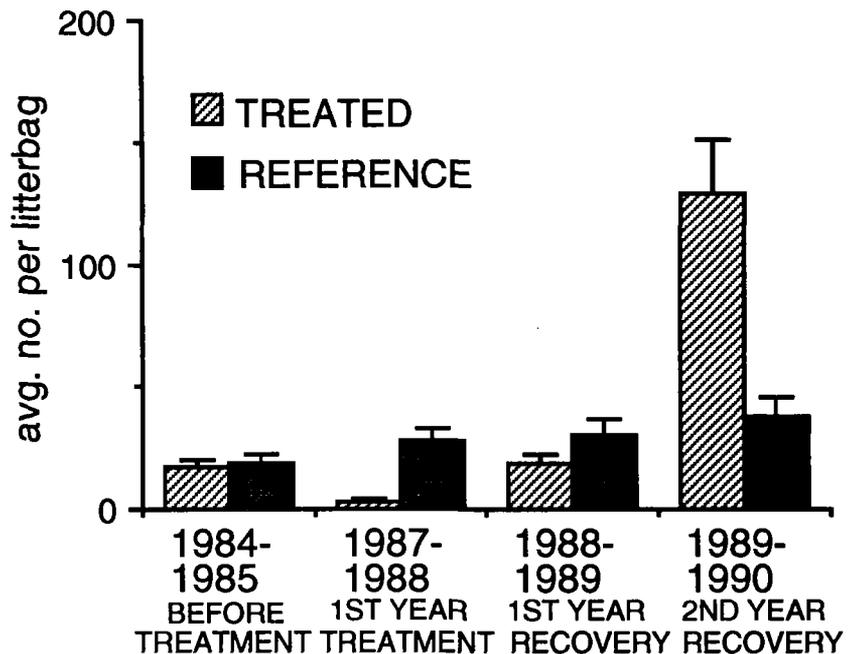


FIG. 2.—Average litterbag densities (number/litterbag  $\pm$  95% CL) of *Lepidostoma* spp. larvae during pretreatment (1984–1985), treatment (1987–1988), 1st-yr recovery (1988–1989) and 2nd-yr recovery (1989–1990) of C54 (treated) and C55 (untreated reference)

than treated or untreated years during 2nd-yr recovery (Fig. 1). However, percent increase of rhododendron processing rates from untreated years to that of 2nd-yr recovery (ca. 48%) (Fig. 1B) was twice that of red maple (ca. 24%) (Fig. 1A).

*Lepidostoma* densities.—Annual average density of *Lepidostoma* in C54 and C55 litterbags was similar before treatment of C54 (ca. 16–18 individuals per litterbag in each stream) (Fig. 2). During treatment of C54, *Lepidostoma* density in litterbags decreased in C54 to two individuals per litterbag (annual average), whereas *Lepidostoma* density in C55 litterbags remained similar to the pretreatment year (Fig. 2). However, by 2nd-yr recovery, density of *Lepidostoma* in litterbags was significantly higher in C54 (128.9) than C55 (37.8) and C54 pretreatment ( $P < 0.01$ ) (Fig. 2).

Benthic density of *Lepidostoma* steadily increased in C54 throughout the 1st yr of recovery (Fig. 3). After 1 yr of recovery, *Lepidostoma* density was substantially higher in the recovering stream than the reference stream (ca. 550 indiv.  $m^{-2}$  vs. ca. 20 indiv.  $m^{-2}$ , respectively, in December 1989) (Fig. 3). During the 2nd yr of recovery, benthic density in the recovering stream (398.7 indiv.  $m^{-2}$ ) was significantly higher than that of the reference stream (30.85 indiv.  $m^{-2}$ ) for averaged January–April 1990 samples ( $P < 0.05$ ).

*Laboratory feeding experiments.*—In the laboratory, *Lepidostoma* larvae consumed significantly more rhododendron litter (mean  $\pm$  SE = 1.062  $\pm$  0.10 mg/mg body wt/day) than red maple litter (0.479  $\pm$  0.10 mg/mg body wt/day) ( $P = 0.001$ ). However, no significant differences in feeding rates were observed when *Lepidostoma* were given a choice between rhododendron (0.565 mg/mg body wt/day  $\pm$  0.12 SE) and red maple (0.286  $\pm$  0.06 mg/mg body wt/day) ( $P = 0.203$ ). During all trials, leaf disks of both types in control chambers

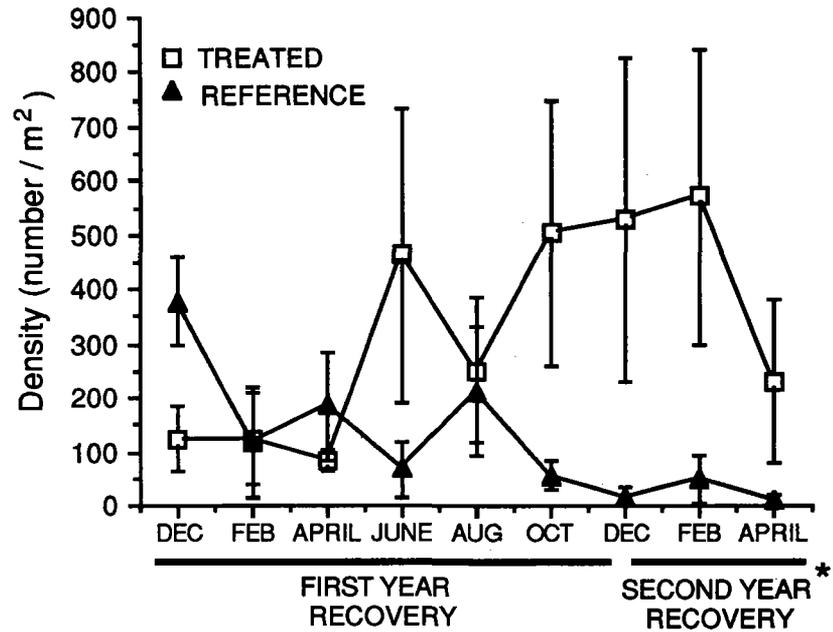


FIG. 3.—Bimonthly habitat weighted average benthic abundances (number indiv. m<sup>-2</sup> ± SE) of *Lepidostoma* spp. larvae in the treated (C54) and reference (C55) streams during 1st 17 mo of recovery (December 1988–April 1990). Asterisk indicates significant difference between pooled C54 and C55 densities during 2nd yr recovery of C54 ( $P < 0.05$ )

either remained unchanged or gained weight after 10 days (mean change for maple = +0.95 mg/chamber ± 0.64 SE; rhododendron = +3.1 ± 1.16 mg/chamber).

#### DISCUSSION

Results of this study demonstrate that a single taxon can have a substantial effect on recovery processes. During recovery from disturbance, *Lepidostoma* colonize rapidly, contributing to rapid recovery of the shredder functional group and litter processing (including refractory species), thus restoring a vital component of energy flow in Coweeta streams. Additionally, this study demonstrates the importance of recognizing that functional recovery in stream systems is not always indicative of taxonomic recovery.

Recovery of other invertebrate groups (e.g., collector-filterers) is dependent upon the recovery of litter processing and associated FPOM generation (Wallace *et al.*, 1986; Wallace *et al.*, 1991). In headwater streams at Coweeta, *Lepidostoma* are apparently a very important component of the recovery process. *Lepidostoma* are at relatively low densities in undisturbed Coweeta streams (Figs. 2, 3), and account for only a small fraction of shredder abundance and biomass (Lugthart and Wallace, 1992; Whiles and Wallace, 1992). However, during recovery periods in C53 and C54, when densities of other shredders were low, *Lepidostoma* dominated the shredder community (Wallace *et al.*, 1986; Whiles and Wallace, 1992). Other common leaf shredders at Coweeta are not as successful at rapid colonization because of shorter flight periods [e.g., *Pycnopsyche* (Trichoptera)], longer life cycles [e.g., *Fattigia* (Trichoptera), *Peltoperla* (Plecoptera)] and/or poor vagility (Lugthart and Wallace, 1992; Whiles and Wallace, 1992).

Rhododendron is one of the most abundant evergreen components of the Coweeta forest

(Day *et al.*, 1988). In undisturbed Coweeta streams where *Lepidostoma* densities are not high, rhododendron is one of the more refractory litter species (Webster and Waide, 1982; Wallace *et al.*, 1986; Cuffney *et al.*, 1990). Nitrogen levels of rhododendron leaves are lower than many other common trees at Coweeta (Webster and Waide, 1982). In addition, rhododendron leaves possess a thick cuticle (Martin and Juniper, 1970). The combination of these characteristics is likely responsible for the normally slow decomposition rate of rhododendron leaves in Coweeta streams. In contrast, red maple litter is quite labile; rates of decomposition for red maple during undisturbed conditions are often less than half that of rhododendron (Fig. 1). Although no clear preference was evident, *Lepidostoma* larvae were able to use rhododendron at a significantly higher rate than red maple in the laboratory. Lower nutritional quality and refractory nature of rhododendron leaves may have necessitated higher ingestion rates.

*Lepidostoma* larvae have been observed feeding on other refractory litter species such as conifer litter in Oregon streams (Grafius and Anderson, 1980) and white pine in Coweeta streams (Whiles and Wallace, pers. observ.). It is likely that *Lepidostoma* are important contributors to post-disturbance recovery of litter decomposition and associated ecosystem processes in a variety of stream systems. As demonstrated here, a single taxon can have a substantial influence on recovery processes.

*Acknowledgments.*—Dr. A. Benke provided some materials necessary for completion of this study. A Marcilio assisted with various aspects of field work. Constructive review of early drafts was provided by Dr. N. H. Anderson, J. Grubaugh, and two anonymous reviewers. This research was supported by grants BSR83-16082, BSR87-18005, BSR90-11661 and DEB892-07498 from the National Science Foundation.

#### LITERATURE CITED

- ANDERSON, N. H. AND J. R. SEDELL. 1979. Detritus processing by macroinvertebrates in stream ecosystems. *Annu. Rev. Entomol.*, **24**:351-377.
- CUFFNEY, T. F., J. B. WALLACE AND G. J. LUGTHART. 1990. Experimental evidence quantifying the role of benthic invertebrates in organic matter dynamics of headwater streams. *Freshwater Biol.*, **23**:281-299.
- CUMMINS, K. W. 1973. Trophic relations of aquatic insects. *Annu. Rev. Entomol.*, **18**:183-206.
- . 1974. Structure and function of stream ecosystems. *BioScience*, **24**:631-641.
- AND M. J. KLUG. 1979. Feeding ecology of stream invertebrates. *Annu. Rev. Ecol. Syst.*, **10**:147-172.
- , J. R. SEDELL, F. J. SWANSON, G. W. MINSHALL, S. G. FISHER, C. E. CUSHING, R. C. PETERSON AND R. L. VANNOTE. 1983. Organic matter budgets for stream ecosystems: problems with their evaluation, p. 299-353. *In*: J. R. Barnes and G. W. Minshall (eds.). *Stream ecology: application and testing of general ecological theory*. Plenum Press, New York.
- DAY, F. P., JR., D. L. PHILLIPS AND C. D. MONK. 1988. Forest communities and patterns, p. 141-149. *In*: W. T. Swank and D. A. Crossley Jr. (eds.). *Forest hydrology and ecology at Coweeta*. Springer Verlag, New York.
- FISHER, S. G., L. J. GRAY, N. B. GRIMM AND D. E. BUSCH. 1982. Temporal succession in a desert stream ecosystem following flash flooding. *Ecol. Monogr.*, **52**:93-110.
- AND G. E. LIKENS. 1973. Energy flow in Bear Brook, New Hampshire: an integrative approach to stream ecosystem metabolism. *Ecol. Monogr.*, **43**:421-439.
- GOLLADAY, S. W., J. R. WEBSTER AND E. F. BENFIELD. 1983. Factors affecting food utilization by a leaf shredding aquatic insect: leaf species and conditioning time. *Hol. Ecol.*, **6**:157-162.
- GRAFIUS, E. AND N. H. ANDERSON. 1979. Production dynamics, bioenergetics, and the role *Lepidostoma quercina* Ross (Trichoptera: Lepidostomatidae) in an Oregon woodland stream. *Ecology*, **60**:433-441.

- AND ———. 1980. Population dynamics and role of two species of *Lepidostoma* (Trichoptera: Lepidostomatidae) in an Oregon coniferous forest stream. *Ecology*, **61**:808–816.
- GRAY, L. J. AND S. G. FISHER. 1981. Postflood recolonization pathways of macroinvertebrates in a lowland Sonoran Desert stream. *Am. Midl. Nat.*, **106**:249–257.
- LUGTHART, G. J. AND J. B. WALLACE. 1992. Effects of disturbance on benthic functional structure and production in mountain streams. *J. N. Am. Benthol. Soc.*, **11**:138–164.
- MARTIN, J. T. AND B. E. JUNIPER. 1970. The cuticles of plants. Martin's Press, New York. 347 p.
- MOLLES, M. C., JR. 1985. Trichopteran communities of streams associated with aspen and conifer forests: long term structural change. *Ecology*, **63**:1–6.
- NAIMAN, R. J. AND J. R. SEDELL. 1979. Characterization of particulate organic matter transported by some Cascade mountain streams. *J. Fish. Res. Board Can.*, **36**:17–31.
- NIEMI, G. J., P. DEVORE, N. DETENBECK, D. TAYLOR, A. LIMA, J. PASTOR, J. D. YOUNT AND R. J. NAIMAN. 1990. Overview of case studies on recovery of aquatic systems from disturbance. *Environ. Manage.*, **14**:571–587.
- SHORT, R. A. AND P. E. MASLIN. 1977. Processing of stream litter by a detritivore: effect on nutrient availability to collectors. *Ecology*, **58**:935–938.
- SWANK, W. T. AND D. A. CROSSLEY, JR. (EDS.). 1988. Forest hydrology and ecology at Coweeta. Springer Verlag, New York. 469 p.
- WALLACE, J. B. 1990. Recovery of lotic macroinvertebrate communities from disturbance. *Environ. Manage.*, **14**:605–620.
- , T. F. CUFFNEY, J. R. WEBSTER, G. J. LUGTHART, K. CHUNG AND B. S. GOLDOWITZ. 1991. A five year study of export of fine organic particles from headwater streams: effects of season, extreme discharges, and invertebrate manipulation. *Limnol. Oceanogr.*, **36**:670–682.
- , D. H. ROSS AND J. L. MEYER. 1982a. Seston and dissolved organic carbon dynamics in a southern Appalachian stream. *Ecology*, **63**:824–838.
- , D. S. VOGEL AND T. F. CUFFNEY. 1986. Recovery of a headwater stream from an insecticide-induced disturbance. *J. N. Am. Benthol. Soc.*, **5**:115–126.
- , J. R. WEBSTER AND T. F. CUFFNEY. 1982b. Stream detritus dynamics: regulation by invertebrate consumers. *Oecologia*, **53**:197–200.
- WEBSTER, J. R. 1983. The role of benthic macroinvertebrates in detritus dynamics of streams: a computer simulation. *Ecol. Monogr.*, **53**:383–403.
- , M. E. GURTZ, J. J. HAINS, J. L. MEYER, W. T. SWANK, J. B. WAIDE AND J. B. WALLACE. 1983. Stability of stream ecosystems, p. 335–395. In: J. R. Barnes and G. W. Minshall (eds.). Stream ecology: application and testing of general ecological theory. Plenum Press, New York.
- AND J. B. WAIDE. 1982. Effects of forest clearcutting on leaf breakdown in a southern Appalachian stream. *Freshwater Biol.*, **12**:331–344.
- WHILES, M. R. AND J. B. WALLACE. 1992. First year benthic recovery of a headwater stream following a 3-year insecticide-induced disturbance. *Freshwater Biol.*, **28**:81–91.
- YOUNT, D. J. AND G. J. NIEMI. 1990. Recovery of lotic communities and ecosystems from disturbance—a narrative review of case studies. *Environ. Manage.*, **14**:547–569.
- ZAR, J. H. 1984. Biostatistical analysis. Prentice Hall, Inc., Englewood Cliffs, New Jersey. 718 p.