Relations Between Large Woody Debris, Physical Habitat, and Benthic Macroinvertebrates in Appalachian Mountain Streams.

by

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(Abstract)

Large woody debris (LWD) was added to North Fork Stony Creek and North Prong Barbours Creek as an experimental stream enhancement technique. The purpose of this study was to determine the stream channel responses to LWD additions at the scale of the stream reach and around individual logs, to determine benthic macroinvertebrate relations to physical habitat, and to determine the potential effects of LWD created stream channel alterations on benthic macroinvertebrates.

Pool frequency and total surface area increased substantially at the expense of riffles in the random and controlled placement sections one year after LWD additions in Stony Creek. Although logs oriented as ramps produced more channel scouring, only log dams created pools. Results in Barbours Creek were similar but less pronounced.

Substratum detritus was greater in pools than riffles for both streams, but there was no difference between years, sections, or as a result of LWD additions. Most functional feeding groups and invertebrate orders were

significantly more abundant in riffles than pools. The exception were collector gatherers which accounted for a large proportion of the benthic macroinvertebrate assemblage. Increased pool area at the expense of riffle area may decrease the overall relative abundances of functional feeding groups and result in a reduction of the stream's substratum detritus processing capabilities.

Decreased riffle area should result in an overall decrease in the abundance of potential food items for brook trout in Barbours Creek, but not in Stony Creek. However, average biomass was not significantly different between pools and riffles for either stream. Biomass may therefore compensate for a loss of prey items for brook trout because, although numbers may decrease, average individual weight increases with an increase in pool habitats.

Acknowledgements

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Introduction

The importance of large woody debris (LWD) is well documented in temperate ecosystems for both terrestrial and aquatic environments (Harmon et al. 1986), where it can serve as a direct interface between land and water. Perhaps the most important function of LWD in streams is its influence on channel morphology. Bisson et al. (1987) list four main actions of LWD in the stream channel: 1) maintains positioning of pools in the channel, 2) creates backwaters along the main channel, 3) causes lateral movement of the stream channel and creates secondary channel systems, and 4) increases depth variability. As a result of these functions, physical habitat complexity is increased, and the debris itself adds a new habitat dimension.

Pools associated with LWD may result from damming of the channel, scouring, backwater eddies, and plunge pools (Bisson et al. 1982). Large woody debris firmly anchored in the channel forms a sediment deposit immediately upstream of the piece and a plunge pool downstream. Numerous pieces acting in this way create a stepped longitudinal profile (Heede 1972) which serves to dissipate the stream's energy (Marston 1982).

Removal of LWD from streams may result in significant changes in stream channel structure. Bilby (1984) reported decreases in pool number and area after a stream cleaning project. Additionally, debris removal may result in reduced retention of sediments (Megahan 1982; Likens and Bilby 1982) and organic materials (Bilby and Likens 1980; Speaker et al. 1984).

Historical stream management practices have viewed LWD in streams as detrimental, and resulted in the removal of LWD from many stream channels (Sedell and Luchessa 1982). To complicate matters, most of the forests in the southeastern United States have been timbered within the last century, often to the very edge of the stream bank and leaving no riparian vegetation (personal observation from aerial photographs). LWD in these streams was often removed for timber salvage, or because of policy. Over time any debris present in stream channels slowly decomposed or was swept away during high flows. For example, Golladay et al. (1989) noted a gradual decline in LWD following logging. This, coupled with slow replacement due to forest succession, has left many Southeastern streams with channels deficient of LWD relative to pre-logging conditions. In the Great Smoky Mountains National Park, LWD volume was four times greater in unlogged streams compared to logged streams (Silsbee and Larson 1983). This pattern generally holds true for old growth verses logged streams throughout North America (Sedell et al. 1988; Murphy and Koski 1989; Robison and Beschta 1990; Richmond 1994).

Because of the importance of large woody debris in lotic ecosystems, a project was initiated to return LWD debris to two streams in the Jefferson

National Forest, Virginia. The intent of this project was to "jump start" the natural system by adding LWD with the hope that by the time the additions are washed out or decomposed, natural recruitment from the surrounding forests will be contributing a steady supply. Restoring LWD loadings to those more like old growth streams should eventually restore stream channel complexity to conditions analogous to those under which the resident biota evolved.

An important facet of this project is the biotic responses to the large woody debris additions. The benthic macroinvertebrate communities in these streams are ecologically important because of their role in processing detritus. Additionally, they serve as an important food base for the fish species inhabiting these streams. The purposes of this study were to determine 1) the stream channel responses to additions of LWD at both the stream reach scale and around individual logs, and 2) what effects channel changes due to LWD additions may have on benthic macroinvertebrates both as potential food items for trout and as a detritus processing community.

Study Site Descriptions

North Fork Stony Creek (hereafter referred to as Stony Creek) originates near the boundary between Monroe County, West Virginia and the northern tip of Giles County, Virginia (Figure 1). It flows in a southwest direction and is a third order tributary to the New River in Giles County, Virginia. The section studied in this experiment is located just above where Forest Road 10420 fords the stream, and extends upstream for approximately 1 km. In this section Stony Creek is first order, residing entirely on national forest lands within the Blacksburg District of the Jefferson National Forest and is in a high mountain valley at approximately 920 m (3000 feet) in elevation.

Stony Creek is a G-4c stream following the Rosgen (1985) classification system. Average width in Stony Creek is approximately 5 m, but is variable. Width to depth ratio is 7.8, and average gradient approximately 1%. The stream channel is composed primarily of large and small gravels, but with a fair amount of sand and some cobble substrates. However, boulders are rare.

Within the study section, Stony Creek flows through a mature forest of hemlock (*Tsuga canadensis*), white pine (*Pinus strobus*), yellow pine (*Pinus echinata*), and a few hardwoods such as black gum (*Nyssa sylvatica*). However, the entire watershed was extensively logged during the 1920's and aerial photographs show logging all the way to the channel, with no riparian

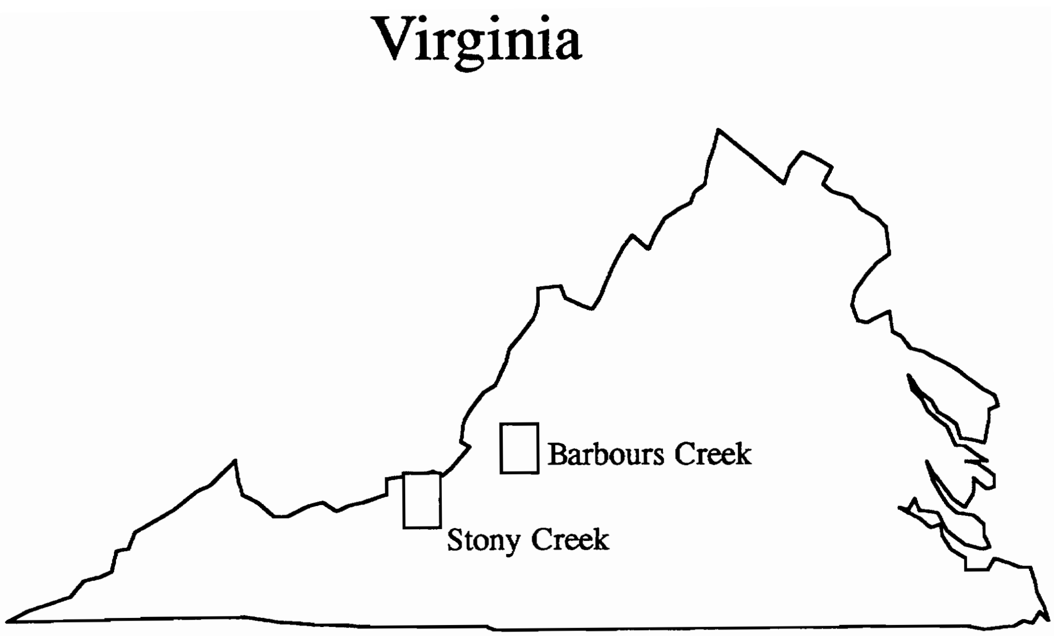

Figure 1. Map of Virginia showing the locations of Stony Creek and Barbours Creek.

vegetation left. Most likely because of this, there is a dense understory of rhododendron (*Rhododendron maximum*) and mountain laurel (*Kalmia latifolia*) covering the floodplain.

North Prong Barbours Creek (hereafter referred to as Barbours Creek) originates near the boundary of Alleghany County and the northeast corner of Craig County, Virginia. It flows southwesterly and is tributary to Craig Creek, which itself is tributary to the James River. The study section on Barbours Creek is approximately 2.5 km upstream of the confluence with South Prong Barbours Creek, and begins about 30 m above where Potts Mountain Trail crosses the stream. The study section extends upstream for approximately 1 km.

Like Stony Creek, Barbours Creek is a first order stream and flows through a mature, second growth hardwood forest dominated by oaks (*Quercus* spp.). The stream is best described as moderately steep A-3 type channel (Rosgen 1985) with an average gradient of 5% and an average width of about 5 m, although the stream is much narrower than its channel throughout most of the year. Width to depth ratio is 7.3 and the dominant substrates are large and small gravels and cobble with numerous boulders.

Chapter One:

Stream Channel Responses to Additions of Large Woody Debris.

I. Reach Level Effects

INTRODUCTION

Large woody debris is an important factor in the maintenance of stream channel complexity by creating and maintaining pools (Swanson et al. 1976; Keller and Swanson 1979; Swanson et al. 1984; Bisson et al. 1987; Maser et al. 1988; Fausch and Northcote 1992). Pools associated with LWD may result from damming of the channel, scouring, backwater eddies, and plunge pools (Bisson et al. 1982). Plunge pools are most common in small mountain streams while deflector and underflow pools increase in frequency with increasing stream order (Bilby and Ward 1989; Robison and Beschta 1990).

The majority of pools in many old growth streams are formed by LWD (Bilby 1984; Andrus et al. 1988; Carlson et al. 1990; Robison and Beschta 1990). Andrus et al. (1988) found that 70% of the pools in their streams were formed by LWD. Similarly, Carlson et al. (1990) reported that 50% of the debris within the bank full channel was contributing to pool formation. Removal of LWD may destabilize the channel by causing erosional downcutting (Bilby 1984) or widening (Maser et al. 1988), increased bedload transport, redistributing of gravel bars, and thalweg shifts (Smith et al. 1993a, 1993b). This instability may result in significant decreases in pool frequency and area (Bilby 1984; Elliott 1986; Ralph et al. 1994), depth (Angermeier and Karr 1984), volume (Fausch and Northcote 1992) and ultimately reduce channel heterogeneity and complexity (Swanson et al. 1984; Fausch and Northcote 1992) while riffle area increases.

The majority of studies encountered are surveys relating land use practices to instream LWD loadings or relating existing LWD to existing channel conditions. While these are valuable observations, they do not track debris-channel relations in real time. The LWD removal studies by Angermeier and Karr (1984), Bilby (1984), and Smith et al. (1993b) are among the few that track the channel adjustments in response to manipulations. The purpose of this study was to describe stream channel responses at the scale of the stream reach to additions of LWD in two Appalachian Mountain streams. The two streams studied had very low LWD loadings and represent typical low gradient (< 1%) and moderate gradient (4%) southern Appalachian Mountain trout streams in Virginia.

METHODS

The study reach of each stream was divided into three treatment sections each approximately 225 m long and separated by a 50 m buffer zone. The furthest downstream section was designated the random placement section and received LWD placed according to a randomization designed to simulate natural log drop. The middle 225 m section was designated the controlled placement section and received LWD placed according to our judgement of how best to manipulate stream habitats. The section farthest upstream served as a reference area in which no manipulations took place. This section was termed the reference section. On Stony Creek the controlled placement and reference sections were separated by only a 10 m buffer because of limited stream length. However, since the reference section received no LWD and was upstream, a 10 m buffer was deemed sufficient.

Habitat Mapping

An initial survey of each stream was conducted to identify and demarcate pool and riffle habitat units following descriptions by Bisson et al. (1982). Runs were recorded as riffles and glides as pools. As each unit was determined, the boundary between it and the preceding unit was marked with surveyor's ribbon. Within each habitat unit, all LWD was recorded in one of four size classes: 1) 10-50 cm in diameter and 1-5 m long; 2) 10-50 cm in diameter and >5 m long; 3) > 50 cm diameter and 1-5 m long; or 4) > 50 cm in diameter and > 5 m long. Each piece was then tagged with its own personal identification number. Additionally, the major pool forming element (eg. LWD, boulder) was recorded.

To document changes in stream habitats, a habitat map was constructed using techniques based on right angle geometry. At the lower boundary of the study reach, a permanent reference stake was secured in the riparian zone. From this stake, a straight line going upstream and roughly parallel to the stream for as far as possible was established. At the terminus of this line, another permanent reference stake was set in the riparian zone. Between these reference stakes, a meter tape was stretched tight and served as a transit line. The compass orientation of the transit line relative to magnetic north was recorded.

At zero distance on the transit line, a transect was extended perpendicular in the direction of the stream channel. Along the transect, the distance from the transit line to the left and right edges of the wetted channel and the left and right edges of the stream bank when full were recorded. Continuing upstream, additional perpendicular transect lines were run where the channel width changes noticeably and at transition areas between habitat units. This procedure was repeated until the entire study reach was completed.

Upon completion of the mapping, stream maps were digitally constructed using ARC/INFO, a GIS software package. The mapping data as collected were a variant of polar coordinates and required transformation to the Cartesian coordinate system (Appendix A) for use in ARC/INFO. From the digital maps, individual habitat units were easily identified, marked, and surface areas for each unit calculated.

Temporal changes in the length of pools and riffles were examined using a three-way ANOVA with interactions. The Shannon-Wiener heterogeneity function (H′) was calculated using the formula

$$H' = -\Sigma\, p_i \times \ln(p_i)$$

to compare the evenness of dispersion of habitat types. Maximum heterogeneity equals the natural log of the number of habitat types sampled (for this study H'_{max} is $\ln(2) = 0.693$), and p_i is the proportion occupied by each habitat type. The distance to the next non-contiguous, like habitat unit (DNLHU) was determined for each individual unit and an overall average calculated (ADNLHU).

Additions of Large Woody Debris

Large woody debris was added to each stream during summer 1993. Stony Creek received 100 pieces representing seven tree species: 50 pieces in the random placement zone and 50 pieces in the controlled placement zone while Barbours Creek received 50 pieces representing six tree species: 25 in the random placement and 25 in the controlled placement zones. Logs were taken from the surrounding forest, but at least 10 m away from the stream bank.

Log additions in the random placement section were determined according to a randomized design intended to simulate naturally recruited wood. The randomization process was hierarchically structured based on four levels: 1) distance from the downstream end of the zone in five meter intervals, 2) side of the stream, 3) orientation to the stream bank in 45° intervals excluding parallel to the stream bank (Figure I.1), and 4) orientation to the water surface (Figure I.1). Positions of individual logs were then determined from computer generated random numbers.

Log positions in the controlled placement section followed the same basic orientations as the random placement section, but were subjectively determined by our judgements of how best to enhance stream habitats. Prior to actual additions, log positions for both sections were marked with surveyor's ribbon on each side of the stream. The length of log needed at a particular site was determined by measuring the distance between ribbons. Selected trees were felled, stripped of limbs, and cut to length. An 11 ton John Deer front end loader with a 5/8 inch cable hydraulic winch was used to position logs in the stream. Logs were not keyed or otherwise pinned to the channel. Heavy equipment was operated in the riparian zone but did not enter the channel.

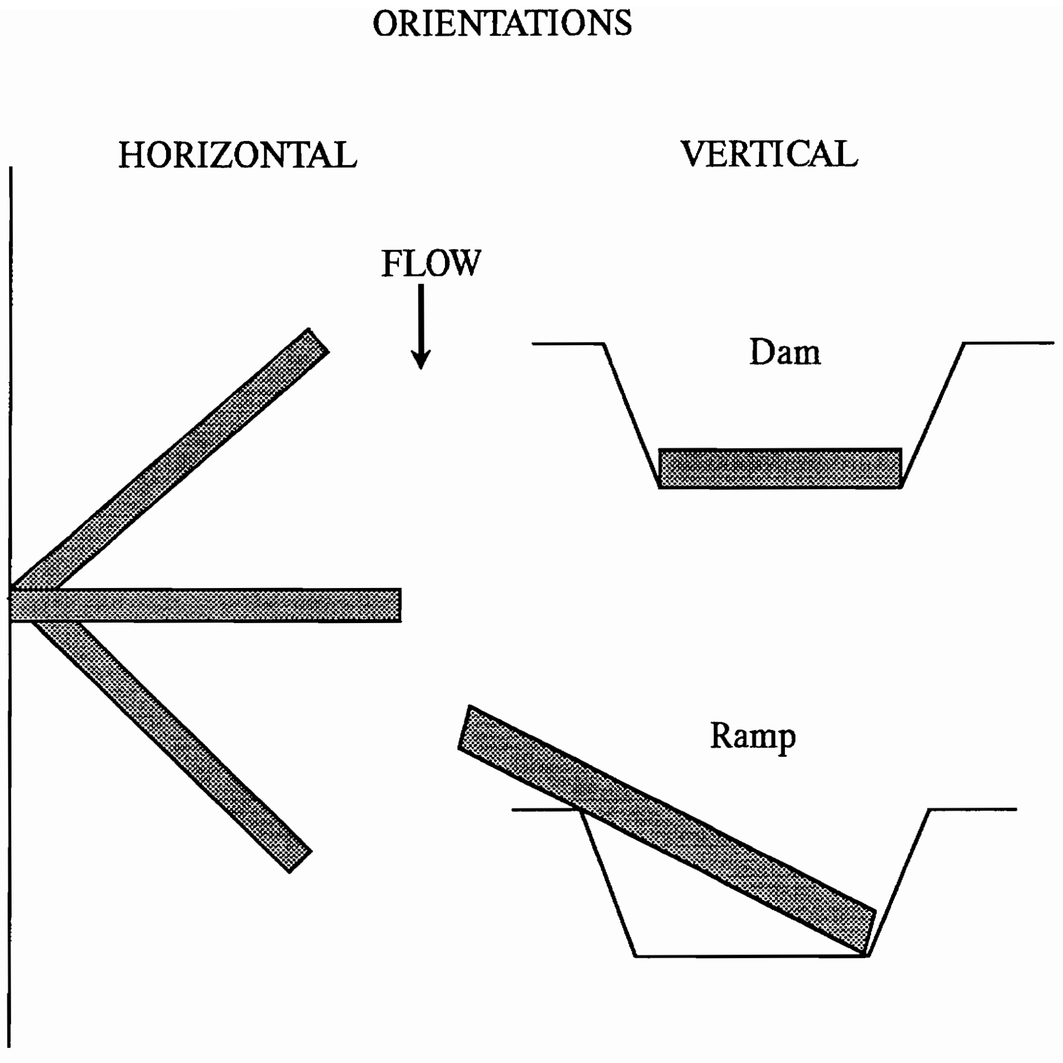

Figure I. 1. Horizontal and vertical orientations used in positioning the large woody debris.

Upon placement of LWD, each piece was tagged and length, diameter at each end, orientation, and species recorded. Additionally, each piece was placed on the computerized stream map using the techniques described earlier. Log volume was calculated by using the average diameter and assuming the log was a cylinder (Lienkaemper and Swanson 1987). The formula used was

$$\text{volume} = 3.1416 \times [(\text{butt diameter} + \text{top diameter}) / 4]^2 \times \text{length}.$$

RESULTS

North Fork Stony Creek

Prior to log additions, naturally recruited LWD in the study reach averaged four pieces per 100 m with the size distribution heavily skewed to the smallest class (Fig 2). Pieces were distributed similarly between pools and riffles (Fig 3) averaging 0.74 pieces per pool and 0.89 pieces per riffle, but followed no known statistical distribution.

Average log length was 6.4 m in the random placement section and 6.3 m in the controlled placement section (Table 1). Mean butt diameter was 37.5 cm and 39.8 cm while mean log volume was 0.57 m^3 and 0.62 m^3 for the random and controlled placement sections respectively. There were no significant differences between treatment sections for average log length, diameter, or volume. Total wood volume added was 28.7 m^3 and 31.2 m^3 in each respective section.

The random placement section consisted of five pools and five riffles prior to LWD additions (Fig 4). Of the five pools, two were formed by pre-existing LWD. One year after LWD additions, there were eight pools and seven riffles. The three additional pools were all created by our log additions. Log 17

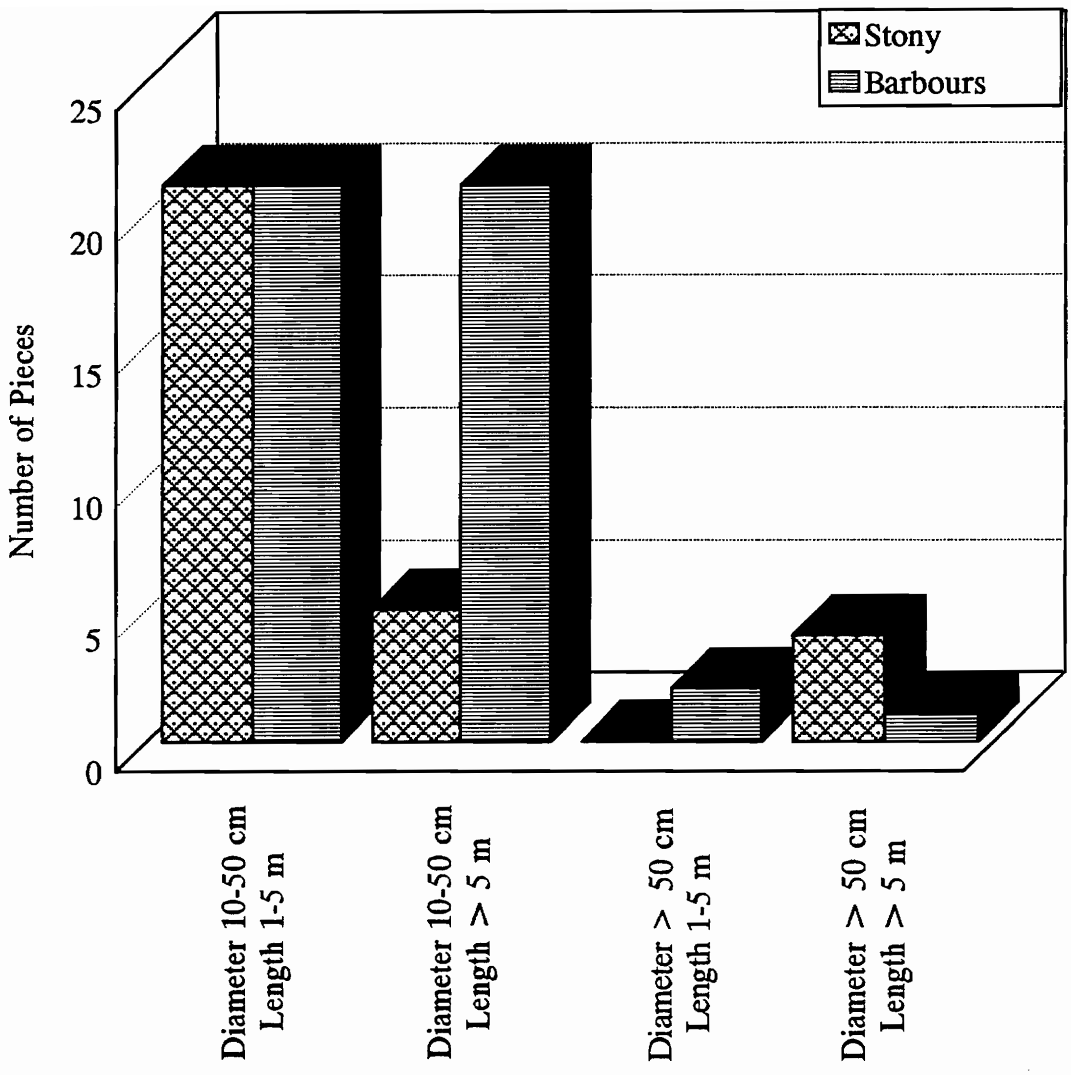

Figure I. 2. Distribution of naturally recruited large woody debris size classes in Stony Creek and Barbours Creek in spring 1993 before log additions.

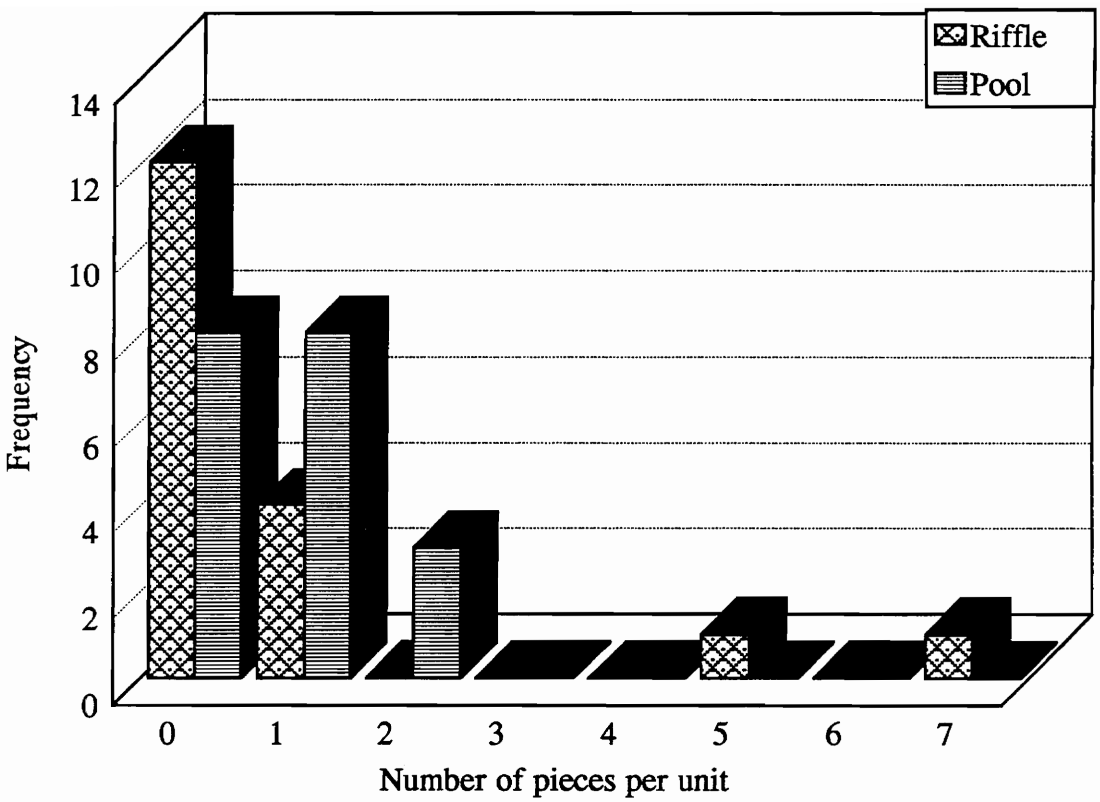

Figure I. 3. Frequency distribution of naturally recruited, pre-existing large woody debris in Stony Creek in spring 1993 before log additions.

Table I. 1. Large woody debris loadings and characteristics in the random and controlled placement sections of Stony Creek and Barbours Creek.

	Stony Creek		Barbours Creek	
	Random	Controlled	Random	Controlled
Number of logs	50	50	25	25
Mean length (m)	6.4	6.3	7.4	8
Mean butt diameter (cm)	37.5	39.8	49.7	49.2
Mean top diameter (cm)	28.8	30.1	38.5	34.5
Volume (m^3) max	1.3	1.3	2.28	2.49
min	0.18	0.29	0.54	0.39
mean	0.57	0.62	1.16	1.17
total	28.6	31.2	29.1	29.1

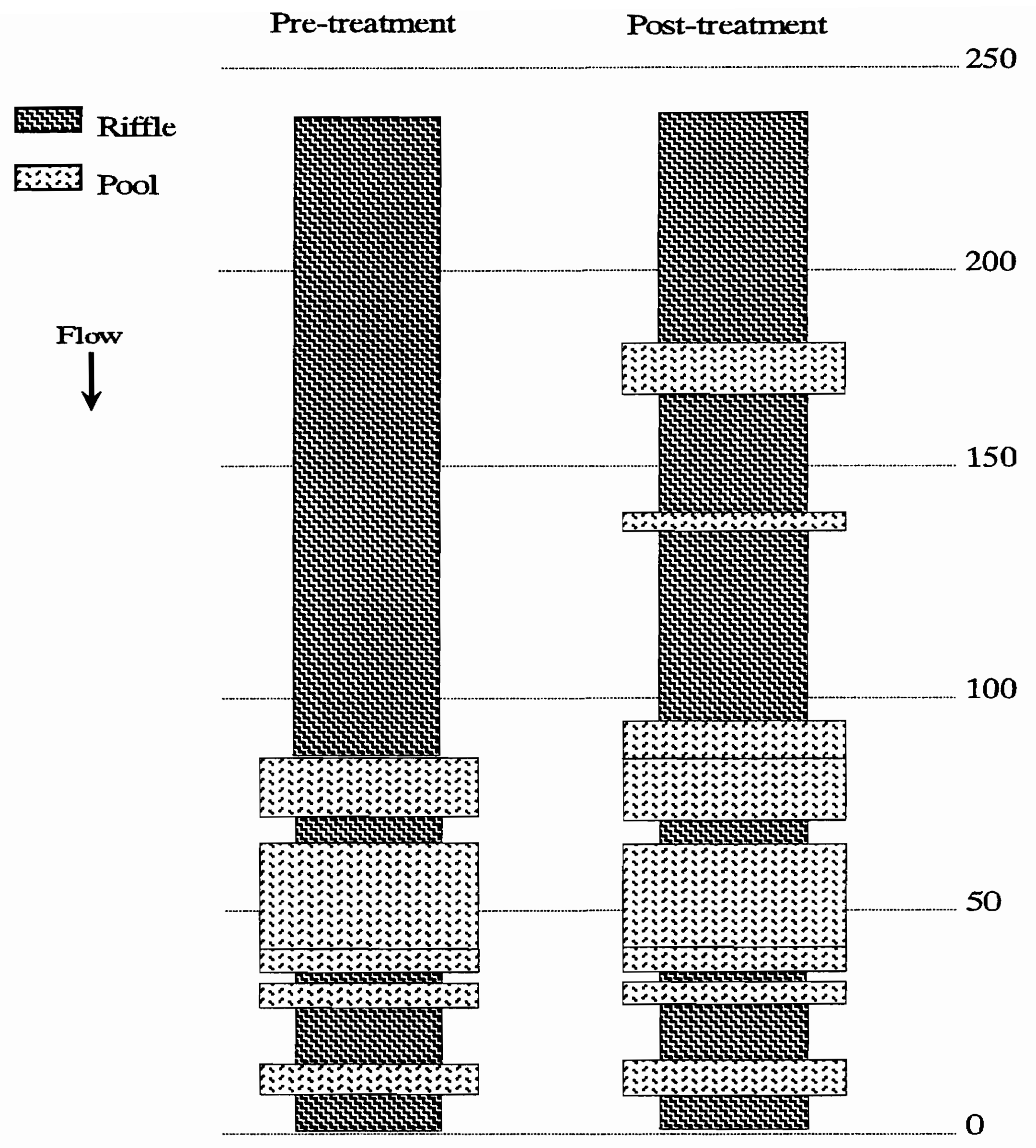

Figure I. 4. Pool-riffle sequencing for the random placement section of Stony Creek before large woody debris additions (pre-treatment) and one year after additions (post-treatment).

dammed a section of stream forming an upstream dam pool and downstream scour pool. Additionally, log 26 caused a partial damming effect to create a pool upstream.

The controlled placement section consisted of five pools and six riffles before log placements (Fig 5). Two of the five pools were formed by naturally recruited LWD. After log additions, the section contained 10 pools and eight riffles. Of the five new pools, four were created by our log placements and one was formed by scour around a midchannel accumulation of detritus. Logs 64, 65, and 67 were responsible for the formation of a scour pool. Log 91 formed an extensive dam which raised the water surface approximately 25 cm, creating the upstream dam pool and a plunge pool immediately downstream. Similarly, log 94 partially dammed the stream and formed an upstream dam pool.

The reference section was comprised of four pools and four riffles (Figure 6). Two of the four pools were formed by pre-existing LWD. Pool-riffle sequencing did not change in this section during the study period. Average riffle length decreased by 43% in the random placement section and by 60% in the controlled placement section after LWD additions while remaining unchanged in the reference section (Table 2). Neither of these differences were significant, although riffles were significantly longer than pools (ANOVA $P=0.004$). Average pool length decreased by 25% in the random

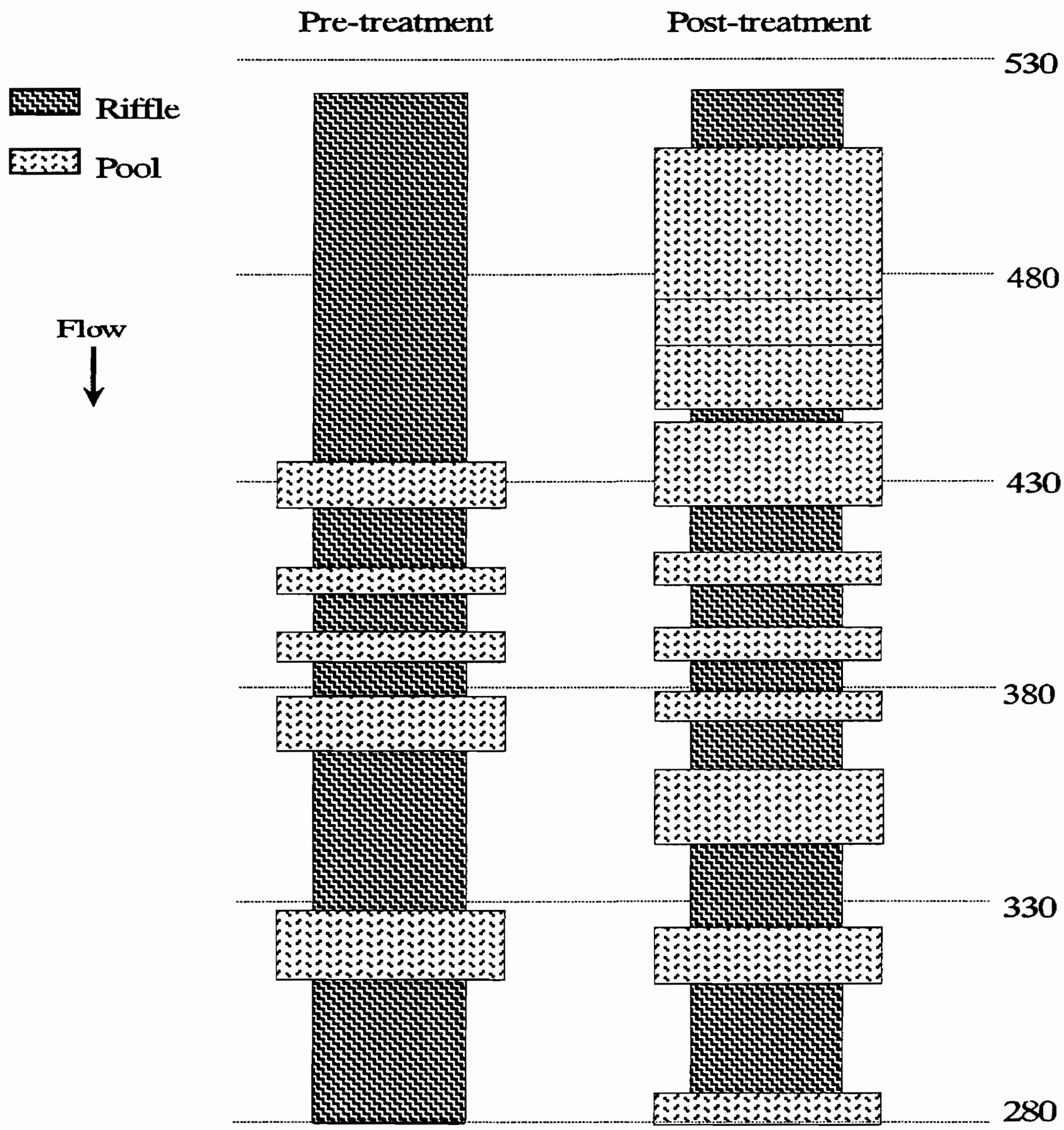

Figure I. 5. Pool-riffle sequencing for the controlled placement section of Stony Creek before large woody debris additions (pre-treatment) and one year after additions (post-treatment).

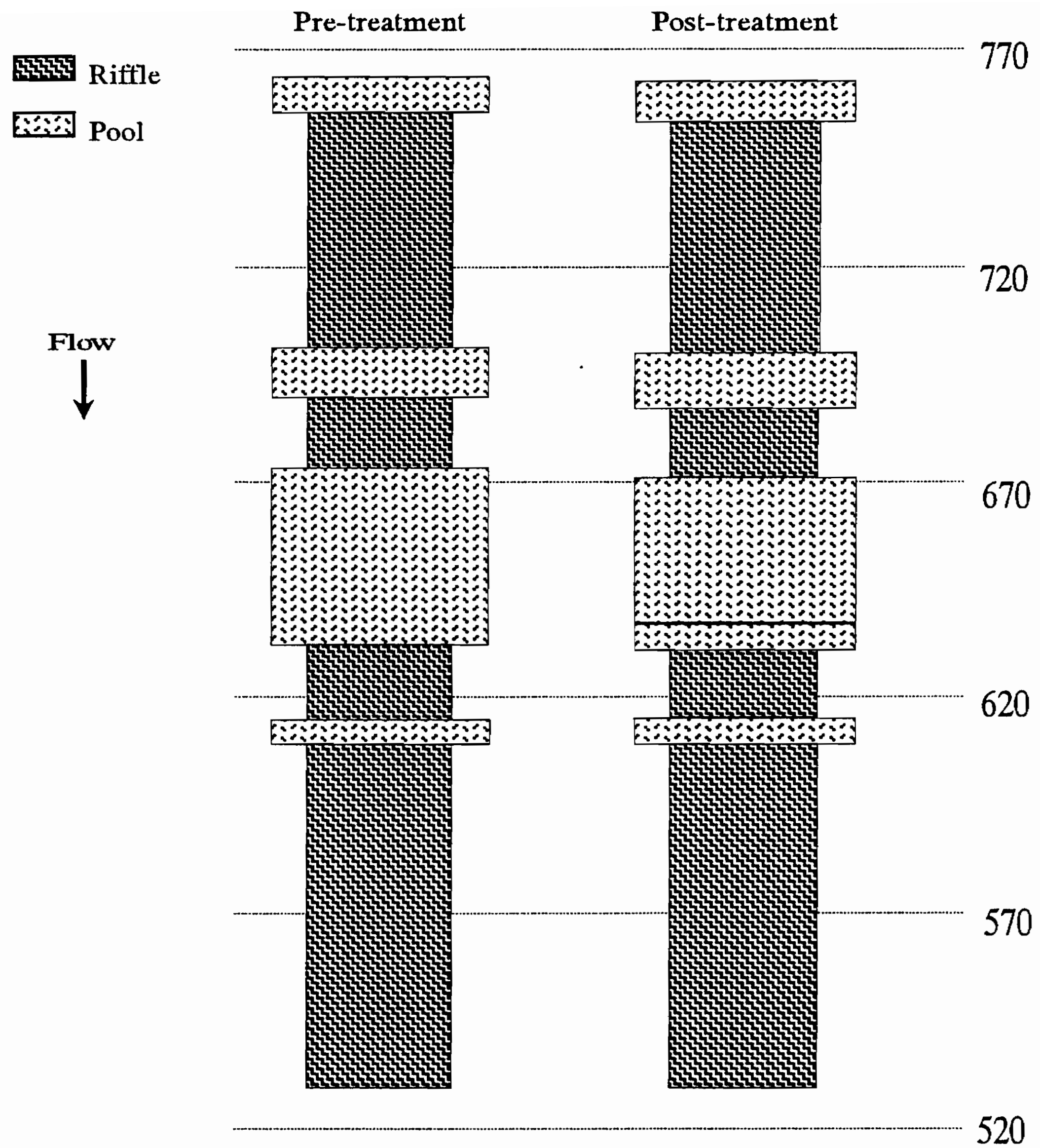

Figure I. 6. Pool riffle sequencing for the reference section of Stony Creek before large woody debris additions (pre-treatment) and one year after (post-treatment).

Table I. 2. Reach level channel characteristics for all three sections of Stony Creek before log additions (1993) and one year after additions (1994).

	Random Placement		Controlled Placement		Reference Section	
	1993	1994	1993	1994	1993	1994
Number of pools	5	8	5	10	4	5
Number of riffles	4	6	6	7	4	4
Mean pool length	11.8	10.4	10.6	14.1	16.6	13
[SD]	[8.5]	[7.5]	[4.0]	[9.3]	[16.5]	[13.5]
Mean riffle length	43.2	24.6	32.1	12.8	41.8	41.5
[SD]	[72.6]	[21.3]	[29.5]	[6.9]	[30.3]	[30.6]
Mean pool area	78	64	45	55	53	44
[SD]	[52.4]	[43.0]	[16.2]	[37.5]	[59.9]	[35.9]
Mean riffle area	219	124	128	55	15	151
[SD]	[357.2]	[100.5]	[119.7]	[21.8]	[136.3]	[137.4]

placement section, increased by 24% in the controlled placement section, and decreased by 10% in the reference section. Surface areas followed the same trends.

Total stream area occupied by pools increased dramatically in both the random and controlled placement sections after log additions while the reference section remained unchanged (Table 3). Percent stream area as pools increased from a pre-treatment value of 29.5% to 39.1% post-treatment in the random section and from 22.4% to 55.2% in the controlled placement section. This translated into a 32% and 146% relative increase in pool area in the random and controlled placement sections respectively. As expected, stream area as riffles decreased proportionately.

Shannon-Wiener heterogeneity (H') as measured for both unit lengths and areas increased from pre-treatment to post-treatment, approaching the theoretical maximum of 0.693 in both the random and controlled placement sections (Table 4). Conversely, heterogeneity in the reference section decreased slightly for both measures.

The average distance to the nearest similar and non-contiguous habitat unit (ADNLHU), decreased through time in all sections, but was most pronounced in the random and controlled placement sections (Table 4). Perhaps more importantly, the standard deviation for ADNLHU in the random and controlled

Table I. 3. Surface areas (m^2) occupied by each habitat type for each experimental section and the percent change before large woody debris additions (pre-treatment) and one year after additions (post-treatment) for Stony Creek.

	Random Placement		Controlled Placement		Reference Section	
	1993	1994	1993	1994	1993	1994
Pool	388	512	222	546	213	220
Percent Change		32		146		3
Riffle	926	796	768	443	623	615
Percent Change		-16		-42		-1

Table I. 4. Shannon-Weiner heterogeneity (H') values for unit length and unit surface area, average distance to the next similar but non-contiguous habitat unit (ADNLHU), and standard deviation for ADNLHU [SD] for each section on Stony Creek prior to large woody debris additions (1993) and one year after additions (1994). H' is measuring the proportions of lengths and surface areas occupied by pools and riffles in each section. At maximum heterogeneity, H'=0.693.

	Random Placement		Controlled Placement		Reference Section	
	1993	1994	1993	1994	1993	1994
H' Length	0.568	0.653	0.522	0.681	0.597	0.595
H' Surface area	0.617	0.676	0.533	0.688	0.571	0.58
ADNLHU	29	19.2	22.8	15.2	29.2	28.9
[SD]	[50.6]	[16.9]	[25]	[13.6]	[26.3]	[26.3]

placement sections was reduced in the post-treatment to approximately one half of the pre-treatment values.

North Prong Barbours Creek

Loading of naturally recruited LWD was 5.2 pieces per 100 m in Barbours Creek, averaging 1 piece per riffle and 0.45 pieces per pool (Fig 7). Logs of the smallest diameter class dominated, but the distribution was split evenly among log lengths (Fig 2).

Average log lengths were 7.4 and 8.0 m in the random and controlled placement sections respectively (Table 1). Mean butt diameter was 49.7 and 49.2 cm, while mean log volumes were 1.16 and 1.17 m^3 for each respective section. Total volume of wood added was 29.1 m^3 in each section.

Prior to LWD additions, the random placement section contained 14 pools and 11 riffles (Table 5). None of the pools could be attributed to formation by LWD. After LWD additions, there were 14 pools and 10 riffles, but the sequencing changed somewhat (Fig 8). Log 6 spanned the channel forming an overhead constriction that at elevated flows created a scour pool. Logs 21 and 22 had the same constriction effect resulting in the lengthening of a pool.

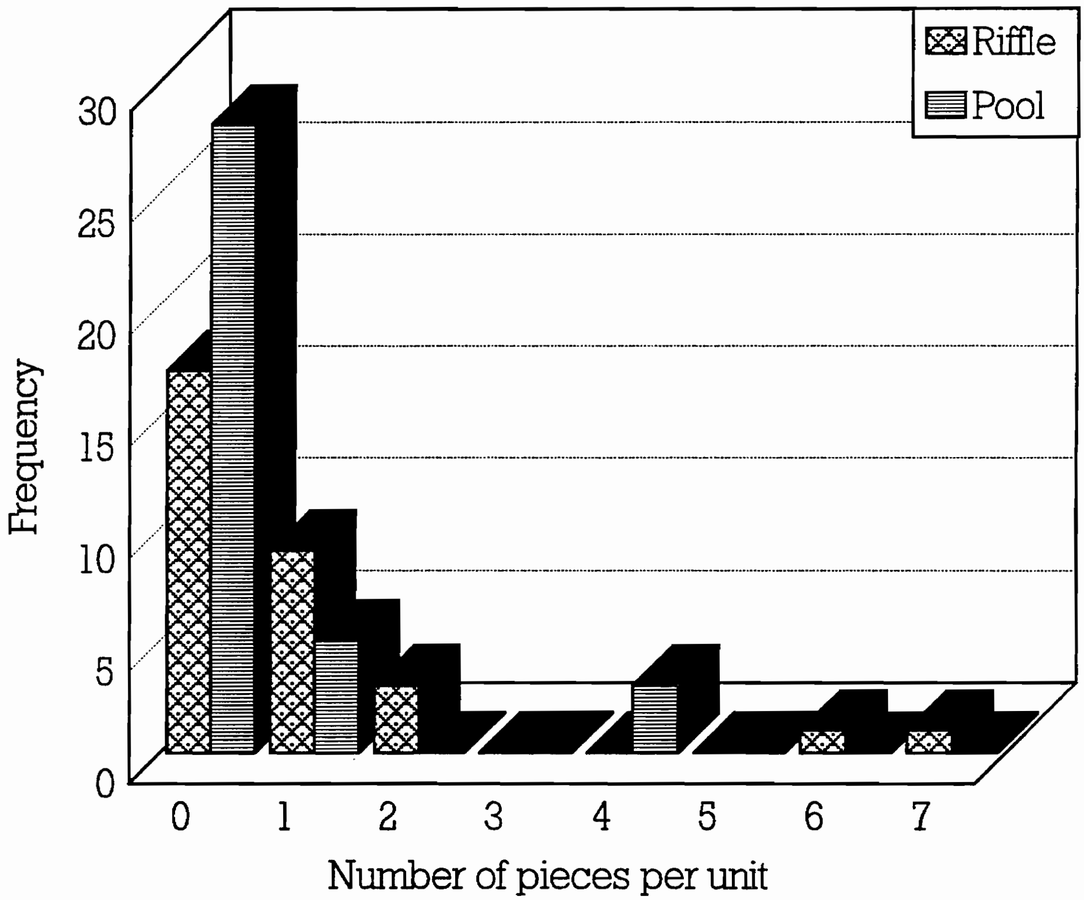

Figure I. 7. Frequency distribution of naturally recruited, pre-existing large woody debris in Barbours Creek in spring 1993 prior to log additions.

Table I. 5. Reach level channel characteristics for all three sections of Barbours Creek before log additions (1993) and one year after additions (1994).

	Random Placement		Controlled Placement		Reference Section	
	1993	1994	1993	1994	1993	1994
Number of pools	14	14	9	10	7	7
Number of riffles	11	10	8	9	5	5
Mean pool length	8.3	9.1	8.4	8.9	9.4	9.4
[SD]	[4.4]	[4.1]	[3.9]	[4.0]	[3.5]	[3.5]
Mean riffle length	12.1	12.2	23.1	19	34.8	34.8
[SD]	[6.5]	[5.0]	[23.6]	[23.1]	[28.0]	[28.0]
Mean pool area	38	41	36	38	33	33
[SD]	[18.9]	[17.5]	[13.9]	[13.7]	[13.7]	[13.7]
Mean riffle area	49	49	95	79	124	124
[SD]	[23.0]	[23.4]	[109.6]	[107.9]	[91.4]	[91.4]

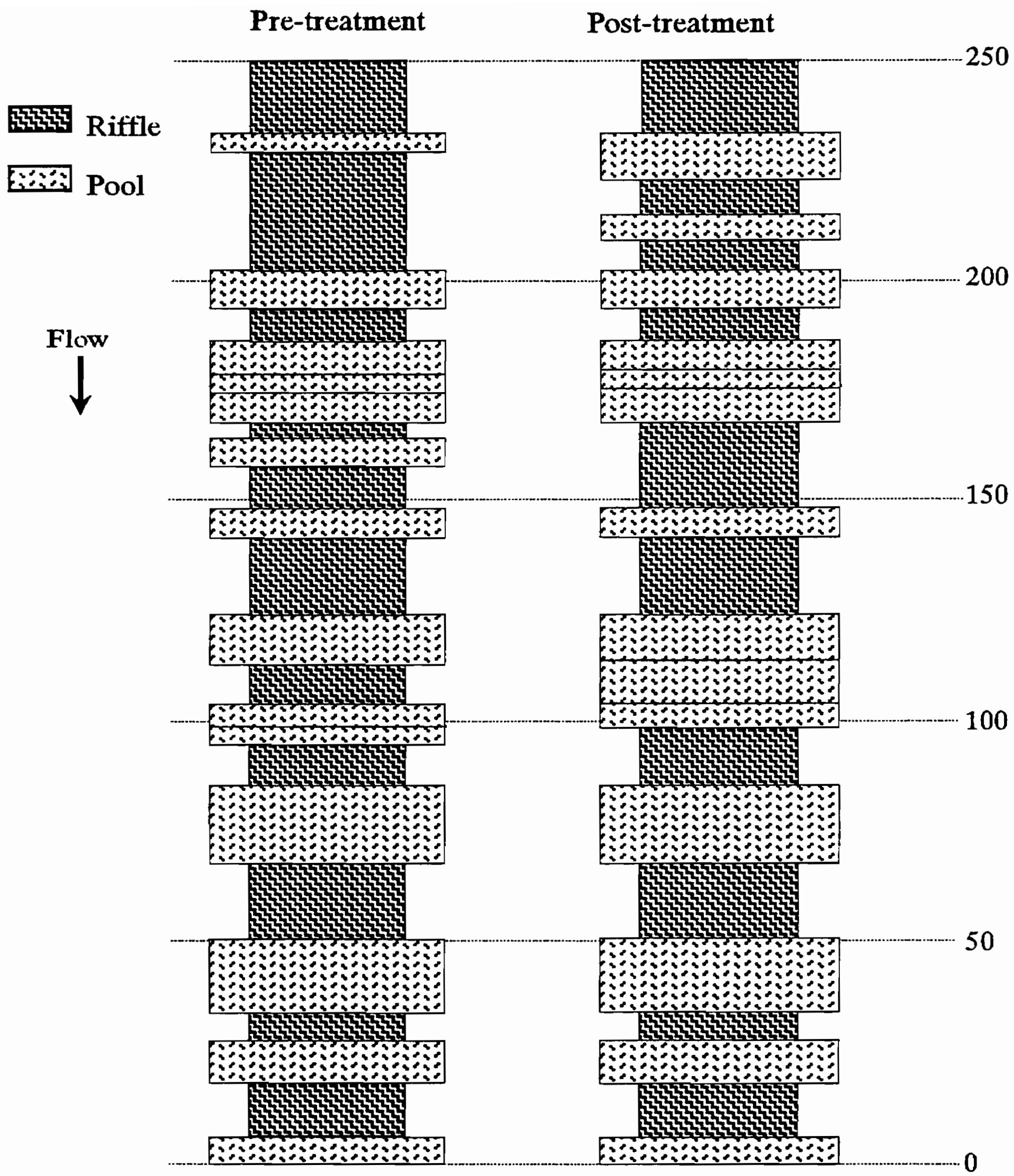

Figure I. 8. Pool-riffle sequencing for the random placement section of Barbours Creek before large woody debris additions (pre-treatment) and one year after additions (post-treatment).

Another new pool was possibly formed by pre-existing LWD. Additionally, one small pool approximately 6 m below the pool formed by Log 6 reverted to a riffle, and another pool reverted to riffle.

The controlled placement section consisted of 9 pools and 8 riffles before log additions (Fig 9) with none of the pools determined to be formed by pre-existing LWD. One year after LWD additions, the section contained 10 pools and 9 riffles. Log 44 in combination with detritus formed a complete dam to create a new pool 13.8 m long.

The reference section contained 7 pools and 7 riffles both before and after LWD additions (Fig 10). No change was detectable. However, naturally recruited, pre-existing LWD was responsible for the formation of 3 of the 7 pools. Two of the pools were a sequence of a downstream plunge pool and an upstream dam pool, while the third was a scour pool.

Average unit length was significantly greater in the reference section than either the controlled placement or random placement sections (ANOVA $P=0.0096$). Additionally, riffles were significantly longer than pools (ANOVA $P=0.0001$). Average riffle length did not change appreciably through time in the random and reference sections, but decreased by 4 m in the controlled placement section. Similarly, mean pool length was greatest in the reference section and approximately equal lengths in the random and controlled placement sections.

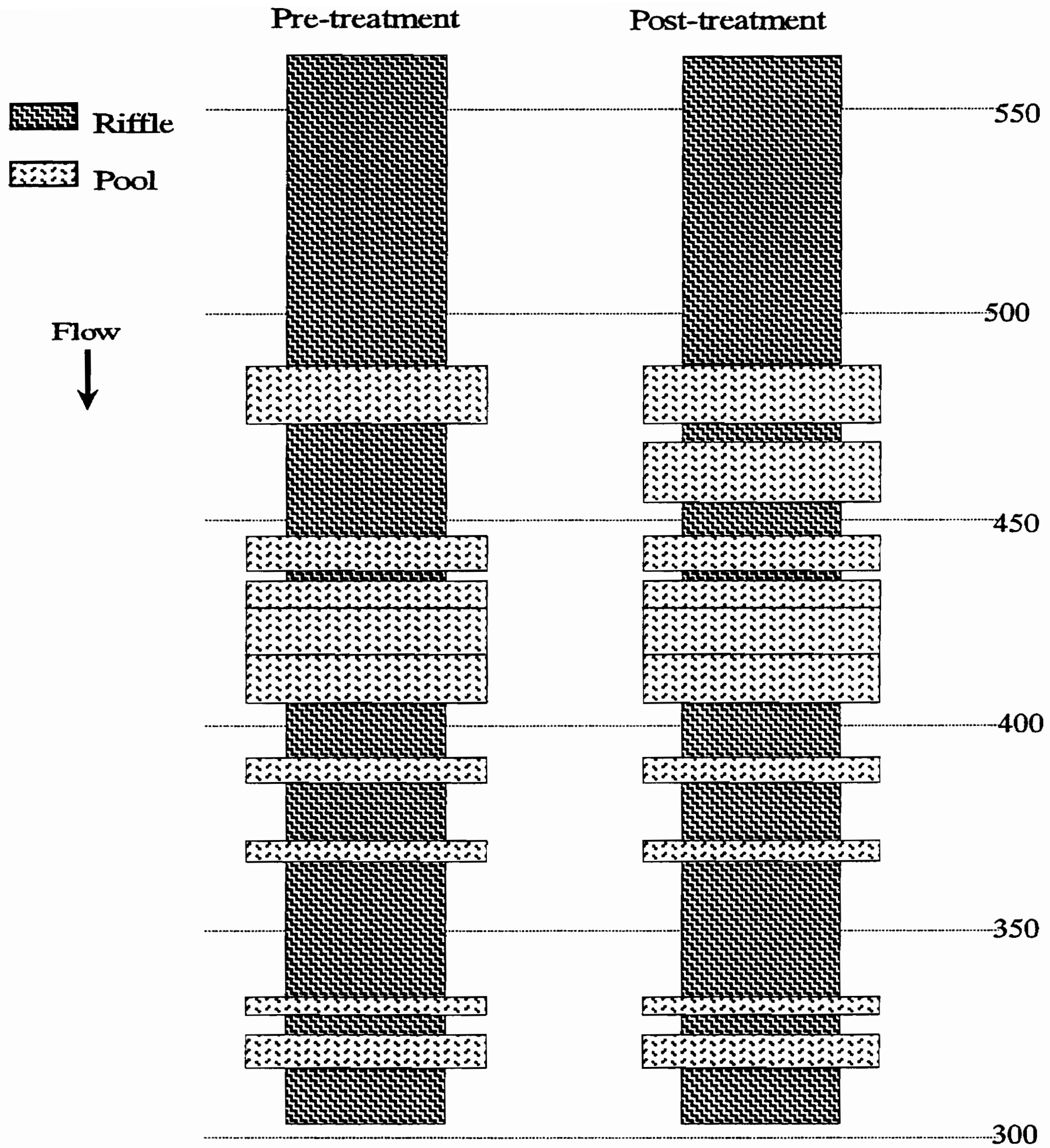

Figure I. 9. Pool-riffle sequencing for the controlled placement section of Barbours Creek before large woody debris additions (pre-treatment) and one year after additions (post-treatment).

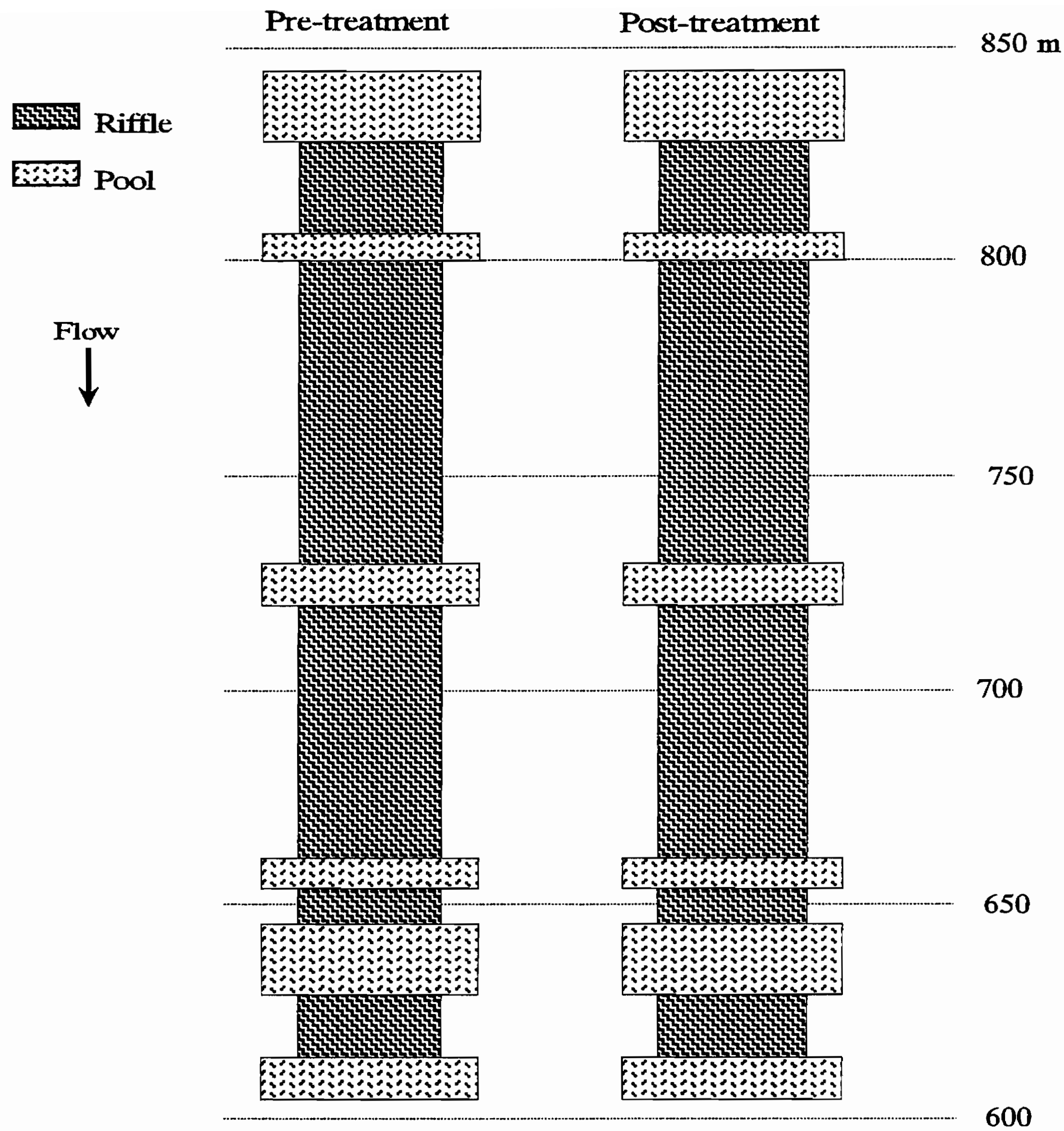

Figure I.10. Pool-riffle sequencing for the reference section of Barbours Creek before large woody debris additions (pre-treatment) and one year after additions (post-treatment).

(Table 5). Lengths did not change appreciably through time. The same general trends also held for mean area occupied by pools and riffles (Table 5).

Total stream area contained in pools increased by 7.9% and 14.8% during the study period for the random and controlled placement sections respectively. The reference section did not change (Table 6). Pool area in the random placement section shifted from 50% of total area pre-treatment to 53.9% post-treatment. Similarly, pool area in the controlled placement section increased from 30% of total to 34.5%, while pool area in the reference section remained at 25.1% of total.

Heterogeneity as measured by unit lengths increased to the theoretical maximum (0.693) in the random placement section. However, this was not reflected in H′ as measured by surface area, where heterogeneity decreased slightly (Table 7). Heterogeneity for the controlled placement section increased slightly through time for both length (0.643) and surface area (0.645). The reference section did not change from its initial value of 0.587.

ADNLHU remained essentially the same in the random placement section (11.9 m), but the standard deviation decreased by 1 m (Table 7). The connectance value for the controlled placement section decreased by 2 m to 15.3 m. Additionally, the standard deviation decreased by 1 m, reflecting a shortening of ADNLHU. The reference section was unchanged.

Table I. 6. Surface areas (m^2) occupied by each habitat type for each experimental section and the percent change before large woody debris additions (pre-treatment) and one year after additions (post-treatment) for Barbours Creek.

	Random Placement		Controlled Placement		Reference Section	
	1993	1994	1993	1994	1993	1994
Pool	533	575	327	375	233	233
Percent Change		8		15		0
Riffle	534	492	759	711	622	622
Percent Change		-8		-6		0

Table I. 7. Shannon-Weiner heterogeneity (H') values for unit length and unit surface area, average distance to the next similar but non-contiguous habitat unit (ADNLHU), and standard deviation for ADNLHU [SD] for each section for Barbours Creek prior to large woody debris additions (1993) and one year after additions (1994). H' is measuring the proportions of lengths and surface areas occupied by pools and riffles in each section. At maximum heterogeneity, H'=0.693.

	Random Placement		Controlled Placement		Reference Section	
	1993	1994	1993	1994	1993	1994
H' Length	0.691	0.693	0.602	0.643	0.587	0.587
H' Surface area	0.693	0.69	0.611	0.645	0.586	0.586
ADNLHU	11.4	11.9	17.3	15.3	21.8	21.8
[SD]	[5.7]	[4.7]	[18.8]	[17.7]	[21.9]	[21.9]

DISCUSSION

Abundance of LWD

Loadings of naturally recruited LWD in both North Fork Stony Creek and North Prong Barbours Creek were well below those of old growth and undisturbed streams, but in the range of values reported for streams flowing through disturbed watersheds. Richmond (1994) reported debris loadings of 2-26 pieces per 100 m in disturbed streams in north central Colorado while Sedell et al. (1988) reported debris loadings of 1-5 pieces per 100 m in streams flowing through recently timbered second growth forests in Oregon. In contrast, debris loadings in southeast Alaska streams draining old growth forests were 15-46 pieces per 100 m (Murphy and Koski 1989; Robison and Beschta 1990), 18-45 pieces in Oregon coast and Cascade Range streams (Sedell et al. 1988), and 18-64 pieces in north central Colorado streams (Richmond 1994). Wood volume in unlogged streams in the Great Smoky Mountains National Park was over four times greater than in logged streams (Silsbee and Larson 1983).

Although both streams currently flow through mature second-growth forests, it is clear from the extremely low debris loadings that both streams were disturbed at some time in the past. Until recently LWD was considered to be a navigation and flood hazard, and was and still is routinely removed from streams and rivers (Shields and Nunnally 1984). Additionally, during logging operations, instream LWD was often salvaged for lumber, and slash accidentally deposited in streams was removed along with resident debris (Bryant 1983).

While I have no evidence of these specific practices occurring in either of the two study streams, it is likely that something similar contributed to the current debris poor conditions. For example, aerial photographs of the Stony Creek watershed in the early 1930's showed logging all the way to the stream banks and evidence of fire damage near the study section. Additionally, I have observed charred remnants of tree stumps in the riparian area within 20 m of Stony Creek. Another possible explanation for low LWD loadings is that of cumulative effects. Both watersheds were extensively timbered during the first quarter of this century, and timbering extended to the stream banks. Consider the following scenario. While the riparian zone progresses through successional stages to second growth forest in the decades that follow logging, instream debris is slowly lost through decomposition or washout. However, this loss is not always gradual. In a five year period separating old growth and clear-cut stream sites, Bilby and Ward (1991) found a decrease in wood abundance, decrease in mean log size, and a change in LWD species composition. Using a model of LWD input and depletion for southeast Alaska, Murphy and Koski (1989) estimate that 90 years after clear-cut logging, instream LWD will be reduced by 70%. As each piece leaves the stream, one more roughness element that dissipated stream energy is lost and stream power increases locally. With the increase in power, each remaining piece is exposed to more force, thus increasing the probability of a piece washing out. In this scenario, it is possible to envision the gradual depletion of debris to current levels. It is also likely that the surrounding forests are still too young to contribute large amounts of debris. Murphy and Koski (1989) estimate that under their model conditions described earlier, recovery to prelogging levels of LWD will take 250 years.

Removal of LWD from streams, such as might occur with intensive logging or decay without recruitment, may result in significant changes in stream channel structure. Bilby (1984) reported decreases in pool number and area after a stream cleaning project. Additionally, debris removal may result in reduced retention of sediments (Megahan 1982; Likens and Bilby 1982), organic materials (Bilby and Likens 1980; Speaker et al. 1984), and increase channel instability (Bilby 1984).

The purpose of this research was two-fold: to restore large woody debris loadings in North Fork Stony Creek and North Prong Barbours Creek to those found in old growth and undisturbed mature forest streams, and to compare the effects of these additions on stream habitats in a low-moderate gradient stream (Stony Creek) and a high gradient stream (Barbours Creek). Debris loading in the treatment sections of Stony Creek (20 pieces per 100 m) is comparable to loadings found in undisturbed streams in the studies mentioned earlier, but loadings in Barbours Creek are still low relative to other locations. However, difficult terrain limited stream accessibility with heavy equipment on Barbours Creek and in the interest of all parties involved required us to limit the total number of log additions to 50. Nonetheless, the additions of LWD are an attempt to "jump start" the system by providing the streams with logs that would normally be present under pristine conditions.

Stream Channel Responses

Additions of LWD increased both the number of pools and total surface area comprised by pools dramatically in both treatment sections of Stony Creek. These effects were most pronounced in the controlled placement section where the number of pools doubled from 5 to 10, and pool area increased by a factor of almost 2.5 after log additions. We established two treatment sections rather than just one so that we could compare not only the effects of LWD additions against the unmanipulated reference section, but also compare human judgement and perceptions against a random design. In the first case, it is quite clear that the LWD additions rearranged channel morphology relative to the reference section. Although mean pool length, area, and their standard deviations decreased in the reference section, it was due to the splitting of one large pool by a barrier to make two pools in 1994.

With regard to comparisons between treatment sections, the results on Stony Creek show that professional judgement in the placement of logs was more effective in creating pool habitats than by just placing logs at random. While both sections initially contained 5 pools apiece, two more pools were created in the controlled placement section than were created in the random placement section. Additionally, the proportional area occupied by pools increased substantially more in the controlled placement section than the random section.

While the differences between sections are clear, the differences between methods of LWD placement are much more subtle. By the very nature of the project (simulating natural LWD recruitment) we were constrained within narrow bounds. A log can be positioned in an almost infinite number of orientations, but without physical anchoring, most positions are too unstable. Therefore, log orientations in the controlled placement section were much the same as in the random placement section. The major difference is that in the controlled placement section, we were allowed the flexibility of using multiple log combinations wherever we wanted. Unfortunately, I cannot quantify subjective (although trained) judgements as to where we decided to place logs. It was a combination of local stream conditions and intuition that decided placement, but the basic orientations were the same as those of the random placement section.

Three of the four LWD additions forming pools were dams. Two were oriented perpendicular to flow and one angled. Bilby and Ward (1989) reported that most of the debris forming pools in their study streams was oriented either perpendicular to flow or angled downstream. These findings were supported by Fausch and Northcote (1992). However, for logs spanning the entire channel, as is the case in this study, the distinction between angled upstream or downstream is irrelevant for dams because each end is seated against the stream bank. The fourth pool was formed by the interaction of three pieces: a perpendicular ramp, a perpendicular dam, and a piece that moved downstream to wedge as a dam in mid-channel under the other two pieces and parallel to flow. Still, the pool was most likely formed by scour around the dams. Although no ramps were primarily responsible for pool creation, numerous areas of scour were observed around many logs. It is probable that some of these scours will lengthen through time to form pools as scoured bedload is transported downstream.

In contrast to the sweeping changes in channel structure induced by LWD additions on North Fork Stony Creek, channel changes in Barbours Creek were more modest. Relative to the reference section, changes in channel form did occur in both treatment sections. Mean pool length, area, and total area occupied by pools increased in both treatment sections while the reference did not change. Additionally, LWD additions created new pools in both sections. However, in the random placement section, pool creations were countered by the reversion of two pre-existing pools to riffles, making the net change in pool frequency zero.

While channel changes occurred in both treatment sections of Barbours Creek, there is no evidence that our placement decisions (controlled placement) were any more effective than log placement by randomized designs. In fact, the random placement method created one more pool than our controlled placement. However, the relative increases in pool area were almost identical between sections.

One plausible explanation for the general lack of change in the treatment sections is that channel conditions limited our abilities to make detailed log placements within the channel. In both of the treatment sections, Barbours Creek is constrained by high, steep banks. Additionally, the high flow channel is much wider than the active channel, indicating the possibility of very high discharges. One of the main placement considerations was that of log stability because we did not pin or otherwise key any pieces in place, and pieces were stripped of limbs. With the stream channel appearing to handle very large discharges, logs were cut to longer lengths and larger diameters to add stability (Likens and Bilby 1982). The net effect of longer log lengths was that most logs spanned the wetted channel and were partially or completely out of the stream during base flow conditions (generally around 25 cm above the water surface), and performed work only under higher flow conditions. It is possible that discharges did not reach sufficient magnitude during the study period to manipulate channel conditions on a scale as large as Stony Creek. The dominant substrate classes are large gravel, cobble, and boulder which require more stream energy and higher discharges to move than the small and large gravel dominated channel in Stony Creek.

Differences in stream gradient may also help to explain the smaller scale channel alterations in Barbours Creek relative to Stony Creek. Gradient ranged between 3-6% in the treatment sections compared to 0.95% in Stony Creek. Ralph et al. (1994) found that while pool frequency decreased in low gradient streams subjected to intensive logging, high gradient (>8%) streams were largely unaffected. They attributed this to factors other than LWD serving as hydraulic controls in the high gradient streams. While Barbours Creek does not constitute high gradient, it is steep enough that boulder and bedrock controls may override the influences of LWD in the steeper sections. Further, a few of the pieces added in the controlled placement section of Barbours Creek were placed specifically for fish cover. While these pieces may potentially manipulate channel features, the use of logs in this way reduces the number of effective logs relative to the random placement section.

Although the smaller magnitude of stream channel response in Barbours Creek to LWD additions may be due to any or all of the above factors, it simply may be that Barbours Creek received half of the number of pieces that Stony Creek received. Doubling the debris loading to equal Stony Creek could produce similar or greater channel responses. In coastal Oregon streams, an average of three pieces of wood were associated with the formation of each pool (Carlson et al. 1990). Gradient for those streams ranged from 2 to 7.5%. Additionally, there is a positive correlation between pool area and volume of LWD (Bisson et al. 1987; Andrus et al. 1989).

One potential source of error is the way in which the digital stream maps and resulting surface areas were constructed. The initial mapping during spring 1993 defined the wetted channel, bank full channel, and individual habitat units. It was assumed that channel widening did not occur, and only individual habitat units were remapped during spring 1994. The wetted and bank full channels were used as templates held constant through time and upon which the new (1994) habitat units were overlaid. In this way, only changes in lengths of habitat units were detected. For example, a slight widening of the stream would not be noticed. There were two reasons for this. First, the unknown measurement error incurred in repeating width measurements would most likely be greater than any increase in resolution we might have gained by remapping all channel widths. Secondly, water level fluctuations occur on a daily basis, and mapping a transient environment would introduce even more errors than using our method. Using the stream channel widths as a template held constant through time should be sufficient for the time scales involved in this study. However, the entire stream including widths should be remapped at least every five years to document lateral channel migrations and widening or narrowing.

Stream habitat "improvement" structures are constructed in an attempt to correct deficiencies in the habitat (requirements) of the target species - usually trout. Usually the goal, whether realized or not, is to achieve stream channel conditions, such as increased habitat complexity, analogous with old growth streams. However, improvement structures by design are anchored for stability (Seehorn 1992) while LWD resides in a transitory state. If the improvement structure is effective, the stream channel is modified in some way. Unfortunately, anchoring of structures may prevent channel morphology from incrementally adjusting to alterations elsewhere in the reach (Beschta and Platts 1986), or result in damage or outright failure of the structure. Frissel and Nawa (1992) reported failure rates ranging from 7-100% in southwest Oregon streams and 0-20% for southwest Washington streams and damage rates ranging from 27-100% and 0-89% for streams in both respective states. Additionally, they found that structures least likely to fail were those that made minimal modifications to the stream channel.

Streams are dynamic systems constantly adjusting to changing conditions. This must be kept in mind when physically altering a segment. Our additions of LWD clearly altered channel morphology. However, by not anchoring the logs, they are free to move as the channel morphology changes. Our design and intentions were to let the debris have the ability to move and adjust to changing channel conditions until each piece found a stable configuration. Pieces may move long distances during high flow before stabilizing (Lienkaemper and Swanson 1980), but once stable, pieces may not move for several decades (Swanson et al. 1984).

Stream channel complexity and LWD

Increased stream channel complexity is one of the benefits of LWD (Pearsons et al. 1992; Fausch and Northcote 1992). However, the definition of complexity and its quantification is difficult and somewhat quixotic. When addressing complexity, one must determine what type of complexity (chemical, physical, biotic, etc.), for what purpose, and the scale at which it operates. Heterogeneity measures such as the Shannon-Wiener H' statistic give an accurate measure of the relative amounts of each variable. Unfortunately, they do not describe the dispersion and sequencing of the variables. Descriptive statistics such as the mean give a general idea as to the typical size of a habitat unit, but also do not describe dispersion or sequencing. For example, a small average unit size would probably indicate good dispersion and generally high complexity. However, if these units were arranged as a series of many small plunge pools in a stepped profile, the results would be misleading because the net effect of many small pools in series would be one large pool.

One way to incorporate both size and dispersion of units while allowing for aggregations due to sequencing is the use of the measure whose acronym I term ADNLHU. This measure gives an overall view of the average distance from one meta-unit to the next. (A meta-unit is here defined as the sum total of several spatially contiguous and similar units such as a series of plunge pools separated only by elevation.) Although each individual unit is geomorphically discrete, when dealing with biological entities operating on very large or very small spatial scales such as benthic macroinvertebrates it may be more appropriate to think in terms of meta-units.

For Stony Creek, ADNLHU decreased dramatically in the random placement and controlled placement sections after LWD additions. This was due primarily to a shortening of riffle lengths by creating pools. If only the average unit length were used, the distance between meta-units would have been underestimated by 2 m for both the random and controlled placement sections. Depending on the ecological unit of interest, this difference could be of major importance. Additionally, the standard deviations for ADNLHU decreased, indicating a reduction in the variability of distances between meta-units. This in theory implies a progression toward greater habitat complexity for the units as defined at the reach scale. At this scale, maximum complexity should be attained when ADNLHU equals the mean channel width and variance is zero.

ADNLHU works well when only two types of habitat are considered. However, when the number of types exceeds two, a measure of the relative proportion for each habitat type is needed. Combining these data with Shannon-Wiener Heterogeneity (H') or other heterogeneity functions gives insight into the relative proportions and dispersion of units while accounting for unit sequencing. As applied in Stony Creek, there appears a trend toward increased habitat complexity due not only to ADNLHU, but also increased heterogeneity due to the creation of pools. Conversely, Barbours Creek did not change.

Although the progression toward greater habitat complexity seems good, we do not know what levels of complexity are beneficial or detrimental. High complexity may be detrimental to organisms such as trout that operate at scales larger than a channel unit. In contrast, organisms or processes at either scalar extreme may benefit from high complexity. It appears then, that the scale of observation and the number and types of habitat dimensions measured must be relevant to the organism or process of interest.

Chapter Two:

Stream Channel Responses to Additions of Large Woody Debris.

II. Microscale Effects

INTRODUCTION

Large woody debris is an integral component of undisturbed low order streams. As related to channel morphology, LWD functions mainly in the formation and maintenance of pool habitats (Swanson et al. 1976; Keller and Swanson 1979). Robison and Beschta (1990) found that 65 to 75% of the pools in undisturbed Alaskan coastal streams were associated with LWD. Stream size and positioning of LWD influences its effects on the channel. In mountainous watersheds, plunge pools dominate first order streams while deflector and underflow pools increase in frequency as stream order increases (Bilby and Ward 1989; Robison and Beschta 1990).

In a flume study, Beschta (1983) found that when LWD was suspended above the stream bed, larger pieces or aggregations resulted in longer and deeper pools. Similarly, Bilby and Ward (1989) found a positive correlation between pool area and the volume of the debris piece forming natural pools. In another flume study, Cherry and Beschta (1989) found that wood oriented upstream (relative to the left side facing upstream) caused major flow disturbances and large scour depths which decreased bank stability. Conversely, logs oriented downstream or perpendicular to the flow increased bank stability relative to the upstream oriented pieces. They postulated that logs positioned perpendicular or downstream to flow require less anchoring than upstream oriented logs for maintaining position at varying flow regimes.

In a British Columbia coastal stream, Fausch and Northcote (1992) reported that most of the debris pieces forming natural pools were oriented perpendicular to flow or downstream. Similarly, Robison and Beschta (1990) reported a high proportion of logs perpendicular to flow in coastal southeast Alaska streams. Additionally, Bilby and Ward (1989) discovered that the frequency of pieces oriented upstream was significantly less than expected while pieces oriented perpendicular or downstream were more frequent than expected and the frequency of pieces parallel to flow equal to expectations.

While many studies report orientations of LWD and infer pool characteristics to wood orientation, there are few field studies examining the processes of channel bed adjustments to LWD and pool formation. The objective of this study is to describe trends in channel aggradation and degradation relative to orientations of LWD.

METHODS

LWD was added to the random placement and controlled placement sections of North Fork Stony Creek as detailed in Chapter 1. In each treatment section, 15 pieces were selected at random for the establishment of permanent channel cross sectional profiles. At each selected piece, the transit line used for the mapping of that particular section of stream was resurrected. Perpendicular to the transit line, three cross sectional transect lines were set around each log, one transect approximately 0.5 m upstream of the log, and two transects at distances 0.5 and 1.5 m downstream of the log. Thus there were three transects at 1 m intervals surrounding each selected log. Each cross sectional transect was anchored in space with rebar stakes set outside of the high water channel, and a metric measuring tape strung tight between stakes.

LWD pieces were categorized in five groups based on horizontal and vertical orientations relative to the stream channel and flow direction (Figure 1). Using the butt end as the point of reference, pieces oriented horizontally relative to flow direction were classified as facing upstream, downstream, or perpendicular. Pieces oriented vertically relative to the channel were classified as flat on the channel (dam) or angled (ramp).

Beginning at the rebar stake (0 m) anchoring the transect, channel bed

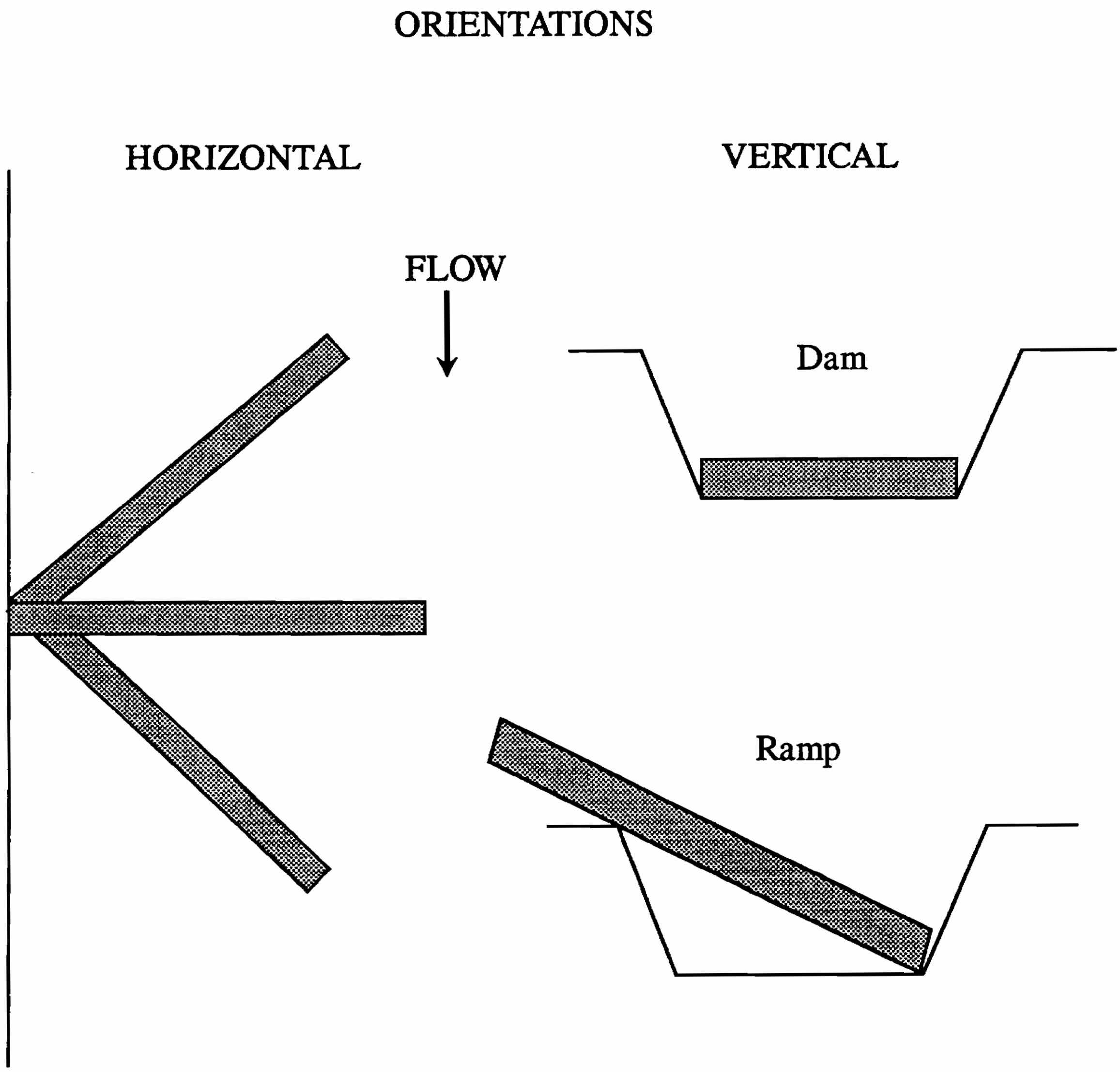

Figure II. 1. Horizontal and vertical orientations of logs relative to the stream channel.

elevation was surveyed using a level and stadia rod moving at 0.5 m intervals from left stake to right when facing upstream. This was done for all three cross sectional transects at each selected log shortly after LWD placement in summer 1993 and again in late spring 1994. Surveying data were calibrated against reference benchmarks set in the surrounding forests.

These calibrated data allowed direct comparison of the 1993 and 1994 surveys because each pair of readings hold space constant and change only through time. Changes in stream bed elevation through time were determined by subtracting each 1994 datum from its 1993 value. Average change in cross sectional depth was calculated in addition to before and after cross sectional profiles. In addition, the stream channel was divided into three sections: left, middle, and right to gain resolution on where average values of aggradation or degradation were occurring relative to each log and each orientation class. Unfortunately, some of the selected logs were displaced out of their survey transects. These transects were not considered in the analyses.

RESULTS

Sixty-three channel cross sections were surveyed both years around 21 logs categorized into five orientation classes: 1) dam perpendicular to flow; 2) dam angled to flow direction; 3) ramp facing upstream; 4) ramp facing downstream; and 5) ramp perpendicular to flow. Dams facing upstream and dams facing downstream were combined into the dam angled to flow category due to low sample size. A dam oriented downstream can be envisioned as the mirror image of a dam oriented upstream. Therefore, values for dam facing downstream were interpreted right to left along the stream channel.

Cross sectional transects are oriented moving in an upstream direction with cross section transect 3 (T3) located furthest downstream, transect 2 (T2) immediately below each log, and transect 1 (T1) located upstream of each log. An arbitrary minimum difference of 2 cm must have occurred between years for a survey point to be classified as changed.

Mean channel elevation change frequencies were split almost equally between degradation (38%) and remaining unchanged (43%) for those cross sections most distant and downstream of logs (T3), while 19% experienced filling (Table 1). All of the T3 cross sections with an average drop in elevation (degradation) were associated with ramps. Dams of both horizontal orientations remained largely unchanged.

Table II. 1. Mean elevational change (cm) in entire channel cross sections and the frequency of occurrence in each response category for differing log orientations. Negative values denote degradation while positive values indicate aggradation. A difference of at least 2 cm was required to be considered a change. Locations of cross sectional transects are: T1, upstream of logs; T2, immediately below logs; and T3, 1-2 m downstream of logs.

		Dams		Ramps		
		Angled	Perpen.	Upstream	Downstream	Perpen.
	N =	3	2	4	7	5
T1	mean	5.63	-1.2	-2.13	-2.31	0.84
	# filled	1	0	1	1	1
	# scoured	0	0	2	4	1
	# no change	2	2	1	2	3
T2	mean	2.17	-2.4	-4.7	-1.64	-2.62
	# filled	7	0	1	1	0
	# scoured	0	1	3	4	3
	# no change	2	1	0	2	2
T3	mean	1.97	-0.65	-2.03	-1.41	-2.48
	# filled	1	0	0	1	2
	# scoured	0	0	2	3	3
	# no change	2	2	2	3	0

In contrast, mean elevation change in T2 cross sections was most frequently scouring (52%) while 38% remained unchanged and aggradation occurred in only 9% of the cross sections. Similar to T1, almost all cross sections with mean net scouring occurred at ramps while dams remained mostly unchanged. Mean net aggradation was rare and occurred for only one ramp and one dam.

In T1 cross sections, net changes through scouring and filling occurred only for ramps with dams remaining unchanged (Table 1). However, mean elevation for the majority (53%) of these cross sections remained unchanged, followed by souring (33%) and filling (14%).

Microscale trends

To standardize comparisons, all logs are described as oriented relative to the left bank for an observer facing upstream. For example, an upstream ramp is a log whose bole is out of the water and resting on the left bank, in the water on the right side of the stream, and pointing upstream from the left bank.

Although variable, most scour for ramps generally occurred immediately downstream of the log and along the sides of the channel. Ramps oriented upstream resulted in mean net scouring along both sides of the channel in all cross sections (Table 2, Figure 2). Additionally, scouring occurred in the middle of the channel immediately downstream of the logs (T2), but mid-channel

Table II. 2. Mean elevational change (cm) in channel cross sectional cells for differing log orientations. Negative values denote degradation while positive values indicate aggradation. A difference of at least 2 cm was required to be considered a change. Locations of cross sectional transects are: T1, upstream of logs; T2, immediately below logs; and T3, 1-2 m downstream of logs.

	Transect	Elevation of left side	Elevation of mid-channel	Elevation of right side
Angled Dam N=3	T1	9.1	2.2	3.6
	T2	4.5	-2.4	3.7
	T3	8.2	-2.3	2.3
Perpendicular Dam N=2	T1	-4.9	-1	2.8
	T2	2.3	-5.9	-2.2
	T3	0.5	0.9	-4.7
Upstream Ramp N=4	T1	-2.1	0.4	-5.3
	T2	-6.4	-2.2	-6.9
	T3	-3.4	-0.9	-2.1
Downstream Ramp N=7	T1	-3.9	0.3	-2.9
	T2	-4.5	1.4	-2.6
	T3	-6.4	0.4	0.8
Perpendicular Ramp N=5	T1	2.9	0.3	-3.3
	T2	-4.2	-3.5	1.1
	T3	-5.2	-0.7	-0.7

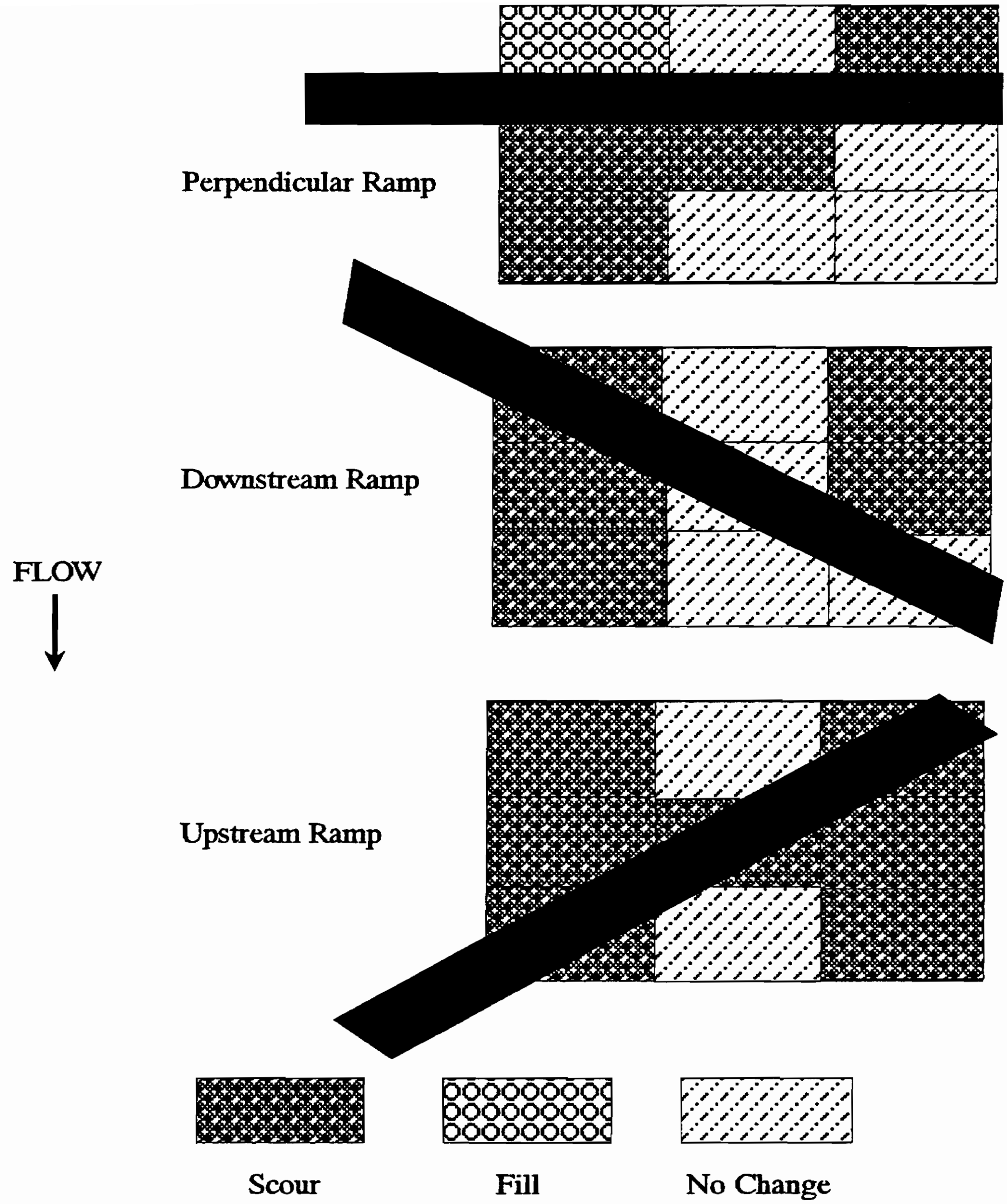

Figure II. 2. Mean channel elevation change for each cross sectional cell relative to orientations of ramps. A difference of 2 cm must have occurred to be counted as a change.

elevations remained unchanged for cross sections T1 and T3.

Ramps oriented downstream demonstrated scour patterns similar to upstream ramps (Table 2; Figure 2). Maximum scour depth occurred downstream of the log and on the left side of the channel (the side upon which the log was ramping). Scouring intensity decreased moving upstream on the left side. Mean channel elevation was unchanged in mid-channel for all three cross sectional transects, and on the right side downstream (T3) of the tip of the ramp. Remaining on the right side of the channel, scour occurred around the zone of contact between log and stream and upstream of the tip.

Scour patterns followed a right to left diagonal moving in a downstream direction for ramps oriented perpendicular to flow (Table 2; Figure 2). Scour was initiated on the right side above the log tips (T1), continued into T2 in mid-channel and the left side, and through the left side in T3. Aggradation occurred on the left side upstream of the ramps while the right side did not change downstream of the ramp tips.

Interestingly, perpendicular dams demonstrated a virtual mirror image of the trends described for perpendicular ramps with the scour diagonal running left to right (Table 2; Figure 3). Additionally, fill occurred immediately downstream of logs (T2) on the left side.

Angled dams experienced filling along both the left and right sides at all three cross section transects and scour in mid-channel in both cells below the

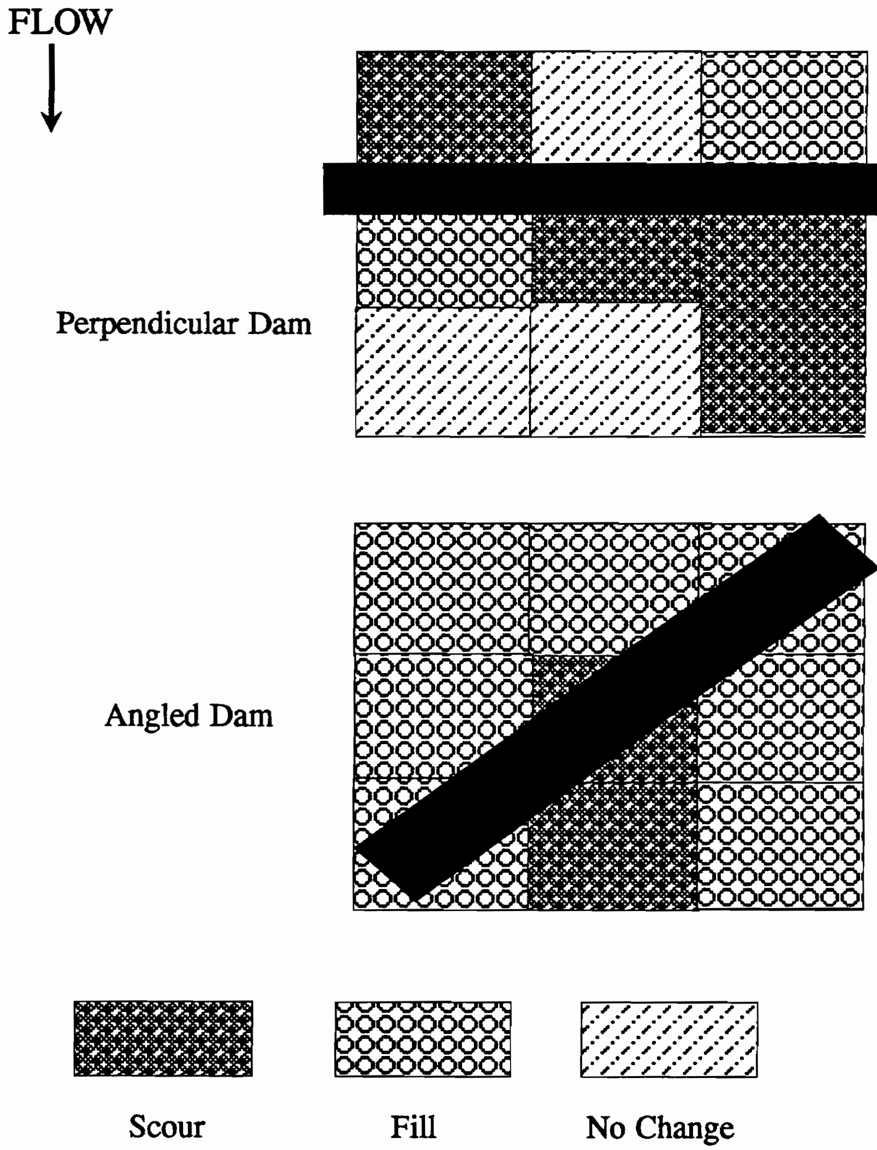

Figure II. 3. Mean channel elevation change for each cross sectional cell relative to orientations of dams. A difference of 2 cm must have occurred to be counted as a change.

logs (Table 2; Figure 3). Additionally the mid-channel cell above the dams experienced filling. Every channel cell for this orientation experienced an average elevational change with no cells remaining unchanged.

DISCUSSION

Cross Sectional Trends

The effects of instream LWD on local stream bed morphology depend in large part on the specific log orientations. When viewed from the average effect on an entire channel cross section, ramps of all orientations tend to increase channel scouring both above and below the logs. Scouring frequency was greatest immediately below the log, and diminished 1-2 m downstream. While the scoured bed load must go somewhere, the low frequency of aggradation 1-2 m below logs implies that bed materials are not merely redistributed locally, but move further downstream. When scour from the transects above logs are included, bed load must move at least 2 m downstream to go undetected by our surveying.

In contrast to the net scouring effect of ramps, the average stream bed elevation for cross sections around dams resisted change. However, the microscale measures showed that more individual cells changed elevation than remained unchanged. This discrepancy suggests that degradation in one section of the transect was countered by aggradation in another section. In addition to the possible lateral redistribution of bed load, deposition of bed load from upstream could balance bed load exported from the cross sections. Dams laying flat on the channel bed may alter hydraulic conditions such that flow is constricted through points such as the thalweg causing scour in some places, and deposition in other places due to physical obstruction of flow. Whatever the cause, the discrepancy in results highlights the importance of choosing the appropriate observational scale for the question asked. However, in this study, multiple questions were asked.

Microscale Trends

Cherry and Beschta (1989) found that in a flume study involving dowels held in controlled conditions, scour area was greatest for dowels held flat on the bed, and maximum scour depth for those dowels occurred near their middle. In the present study, mean maximum scour depth also occurred near the middle of the log (mid-channel) for both orientations of dams, but scour area was greatest for ramps. In fact, for angled dams the only scour was in mid-channel cells below logs, while cells on both sides aggraded above and below the logs. The difference is most likely due to the dams on Stony Creek spanning the entire channel width while the dowels used by Cherry and Beschta stopped in mid-channel. Their configuration allowed unrestricted flow through half of the flume while dams on Stony Creek influenced flow over all parts of the channel width. In the flume, scouring most likely began around the mid-channel tip of the dowel and undercut back along the dowel's length. Conversely, in Stony Creek, the thalweg was near mid-channel for all but one of the dams, so with the logs laying flat on the bed, the only clear flow path under the dams would be in mid-channel. This intuitively makes sense. Similarly, in closely examining Figures 2-6, most of the degradation-aggradation cell patterns found can be conceptualized when flow direction, constriction, and deflection are considered in relation to log orientations.

As mentioned earlier, ramps were more effective in causing scour than dams. Like dams, ramps spanned the entire channel width with the ramp's wetted tip touching the wetted portion of the stream bank. Such a configuration may constrict and redirect flow to result in the generalized scour patterns observed. The Cherry and Beschta (1989) flume study found that pieces oriented upstream decreased bank stability by redirecting flow. This same pattern was observed for upstream ramps in addition to deep scour around the log's tip. However, angled dams did not hold to this pattern, most likely because of the reasons discussed earlier for dams.

The dominant substrate classes in Stony Creek are large and small gravels. These substrates are more easily manipulated than the cobbles and boulders associated with higher gradient trout streams in the Appalachian Mountains, and should in theory produce larger changes. While the channel adjustment patterns observed in this study generally make sense, it must be remembered that sample sizes for each log orientation were small. Sample sizes ranged from two (perpendicular dams) to seven (downstream ramps) logs. Additionally, other factors such as thalweg position, type of channel unit (eg. pools), channel curvature, and other logs may mask or interact with each other to produce the observed results. Considering the variability in stream environments, these trends are just that - trends. A much larger sample size is needed for a definitive answer, and even then, local hydraulic conditions may override the predictable results of controlled flume studies.

Chapter Three:

Distribution and Abundance of Benthic Macroinvertebrates, Substratum Detritus Dynamics, and Effects of Large Woody Debris.

INTRODUCTION

LWD increases habitat diversity and adds structural complexity to stream systems. More complex habitats may yield greater stability to species assemblages during periods of disturbance than simple habitats (Schlosser 1987; Pearsons et al. 1992). Increased diversity of depths and current velocities created by LWD provides alternative strategies for organisms while increased depth may allow species to stack in the water column (Fraser 1969). Additionally, LWD itself provides an additional habitat dimension as a colonization substrate and food source for invertebrates (Anderson et al. 1978; O'Connor 1991).

Relative to other non-mineral substrates the residence time of LWD is much longer and it provides a stable energy base that often resists disturbances. More importantly, LWD increases retention of sediments (Megahan 1982) and allochthonous inputs (Bilby and Likens 1980, Speaker et al. 1984, Trotter 1990) making more food available for a longer period of time in a given area. While increasing the average residence time of a nutrient unit, LWD should shorten spiraling lengths and result in higher productivity (Elwood et al. 1983) and greater faunal biomass. In small streams, terrestrial organic matter retained by LWD occurs in frequent, small clumps (Bilby and Ward 1989). A high frequency of occurrence should reduce spatial variability in food sources allowing for the system to recover more quickly from disturbance. Thus, LWD and its secondary effects can greatly enhance the aquatic system's stability and resiliency.

With the exception of feeding, very little information is available on direct relations between benthic macroinvertebrates and LWD. Elliott (1986) reported a drastic decrease in benthic invertebrates immediately following debris removal in a southeastern Alaska stream. However, densities returned to pre-removal levels one year later. The initial decrease was attributed to the loss of substrates (LWD) that nearly all benthic taxa used. Similarly, Anderson et al. (1978) found that only a few of the aquatic invertebrates associated with LWD in Oregon streams were wood eaters; the majority used wood for attachment substrates, pupation, and emergence.

A large component of Invertebrate communities in forested, temperate streams are thought to be structured on the basis of detritus processing with several functional feeding groups (except scrapers and predators) evolved to capitalize on energy leakage of the larger particle detritivores (Cummins 1974, Vannote et al. 1980). Leaves appear to be the most important energy source driving the community (Minshall 1967). Anderson et al. (1978) found that invertebrate standing crops in leaf debris were two orders of magnitude higher than on wood. This relationship appears to hold for detritus trapped in the substratum as well (Egglishaw 1964, Rabeni and Minshall 1977, Flecker 1984).

More than 75% of the allochthonous input to streams comes from autumn litterfall (Cuffney and Wallace 1989). Because the majority of the energy input comes in a relatively short pulse, the retention capacity of LWD becomes very important for providing a food reservoir throughout the year. Experimental additions of whole-leaf detritus in a Pacific Northwest stream resulted in greater abundance of the collector-gatherer and collector-filterer functional feeding groups (Cummins 1973) than before additions (Richardson and Neill 1991). The presence of LWD increased retention of coarse particulate organic matter (CPOM; leaves). This allowed for more complete leaf processing at the site by shredders. Subsequently, more fine particulate organic matter (FPOM) was made available, which secondarily supported higher densities of collector-gatherers and collector-filterers.

From the little information available, it appears that the influence of LWD on invertebrates is more a function of habitat alteration and maintenance than it is a food source, with the habitat alterations affecting storage of food. Stream channel alterations by LWD additions can be inferred to significantly influence benthic invertebrates through changes in the numbers and sizes of pools and riffles. Abundances are reported to be greater in riffle (erosional) habitats than in pool (depositional) habitats for most taxa except Chironomidae (Rabeni and Minshall 1977; Huryn and Wallace 1987; Brown and Brussock 1991). Thus, changes in the proportions of each habitat type will trigger changes in the abundances of the benthic fauna at the scale of the stream reach and higher. Although as yet theoretical, these changes could create a cascading effect which may change the assemblage structure or system functions in a given stream reach.

The specific objectives of this study were:

1) to describe and characterize the benthic macroinvertebrate communities in North Fork Stony Creek and North Prong Barbours Creek.

2) to describe the substratum detritus dynamics in both streams.

3) to relate changes in stream channel characteristics via LWD additions (Chapter 1) to macroinvertebrate distribution and abundance.

METHODS

Benthic macroinvertebrates were quantitatively sampled using a portable invertebrate box sampler (PIBS) with 350 um mesh net. Total area sampled by one PIBS was 0.1 m^2. Sampling occurred on both streams during late May 1993 (pre-treatment) and late May 1994 (post-treatment). Both the reference section and controlled placement section were sampled on each stream. The random placement section was not sampled because of possible confounding effects of the controlled placement section located upstream, and it was assumed that responses should be similar for both treatment sections.

Five paired riffle samples and three paired pool samples were collected at randomly determined locations in each section. In an attempt to minimize variability, members of a pair were collected in similar local habitat conditions. However, to maximize variability among pairs (attempting to gain a representation of pools and riffles for the entire reach), samples were taken in differing habitat conditions within pools or riffles. Therefore, individual samples were collected in differing habitats within pools or riffles, but members of the pair comprising a sample were collected in similar local habitats. In total, 8 samples were collected for both the controlled placement and reference sections yielding a total of 16 samples per stream per sampling period.

During collection, the substrate was disturbed to a depth of 8-10 cm and was disturbed for as long as was felt necessary to collect all organisms. It was assumed that all leaf detritus was also collected during sampling. Because of low current velocities in pools, a hand operated diaphragm pump was used when sampling pools to generate the current necessary to carry invertebrates into the collection net. Additionally, the maximum depth sampled in pools was limited to the height of the PIBS sampler. Samples were preserved in 95% ethanol.

In the laboratory, organisms and leaf detritus were separated from any mineral substrates by elutriation in a column of moving water and collected in a 500 um sieve. Invertebrates were separated from the detritus by hand with the aid of a magnifying lamp. Additionally, the mineral substrates were examined for any remaining invertebrates. Organisms were counted and identified to genus where practical. Most Diptera except for members of Tipulidae were identified to family. However, identifications were taken as far as needed to determine trophic relationships (Merritt and Cummins 1984). All invertebrates (except Chironomidae) in a given sample were retained, dried, and collectively weighed for biomass calculations. Detritus collected in the sieve during elutriation was also retained, dried to constant weight, and weighed.

Statistical analyses

Analysis of variance (ANOVA) techniques were used to assess the patterns of variability within and among benthic macroinvertebrate, biomass, and detritus samples. In an attempt to control variability and increase statistical power, the mean of each pair constituting a sample (described earlier) was used for analysis rather than each individual member. (Treating samples in this way generally reduced the coefficient of variation for each group described below by approximately 10%). This was also done because members of a sample pair were not independent, which would violate one of the assumptions of ANOVA (Sokal and Rohlf 1981). The benthic macroinvertebrates were examined using three groupings: 1) functional feeding groups (community process); 2) taxonomic relationships (common orders); and 3) potential food organisms for brook trout (*Salvelinus fontinalis*). All data were normalized using a $\log_{10}(Y+1)$ transformation. All transformed data passed the Kolmogorov-Smirnov one sample goodness of fit test. Similarly, detritus and biomass were normalized using the same transformation.

Three-way ANOVAs were applied to assess the differences in density among habitat unit type, experimental section, year, and their interactions on the dependent variables detritus, biomass, and potential food organisms. Three-way multivariate analysis of variance (MANOVA) was used to assess overall differences among habitat unit type, experimental section, year, and interactions on all functional feeding groups simultaneously and on all taxonomic groupings simultaneously. In addition to overall effects, MANOVA performs separate ANOVAs using the specified class variables on all dependent variables. These were used to examine structure patterns within each group (eg. predators). A four-way ANOVA was used to determine numerical dominance in assemblage structure based on habitat unit type, section, and year. (This last test may appear to be unnecessary, but was critical in building a logical argument for assemblage structure and function.) Finally, a correlation matrix was constructed to examine the relationship of substratum detritus with the various functional feeding groups.

RESULTS

Due to the large number of interactions and groups examined, presentation of measures of variability were not feasible. Therefore, the mean and standard deviations of all taxa collected by habitat type and year are presented in Appendix B for Stony Creek and Appendix C for Barbours Creek.

Stony Creek

Over 27,000 invertebrates representing 47 families and genera and 11 orders were collected in Stony Creek for the spring 1993 and spring 1994 sampling periods (Table III.1). A three-way ANOVA on taxa richness showed no significant differences between pools or riffles ($P = 0.2945$), between years ($P = 0.1208$), or between sections ($P = 0.1386$). Mean number of taxa are given in Figure III.1. Diptera (true flies) were most abundant followed by stoneflies (Plecoptera), beetles (Coleoptera), mayflies (Ephemeroptera), caddisflies (Trichoptera), and worms (Oligochaeta) in that order (Figure III.2). Three-way MANOVA results show a very highly significant overall effect ($P < 0.0001$) of habitat unit type (pools and riffles) on density of the various groups (Table III.2). Indeed, three-way ANOVAs for each invertebrate group show significant effects ($P < 0.05$) of habitat unit type on all groups except Diptera (which was almost significant; Table III.3). Stoneflies, beetles, caddisflies, and

Table III. 1. A list of taxa collected in Stony Creek in the spring 1993 and spring 1994 samples. Asterisks denote taxa included as potential food items for brook trout.

Coleoptera:
- Elmidae:
  - Optioservus
  - Oulimnius
  - Promoresia
- Ptilodactylidae:
  - Anchytarsus

Collembola:
- Entomobryidae*
- Hypogastruridae*

Decapoda:
- Cambaridae:
  - Cambarus*

Diptera:
- Ceratopogonidae*
- Chironomidae*
- Empididae*
- Nymphomyidae*
- Simuliidae*
- Tabanidae*
- Tipulidae:
  - Dicranota*
  - Hexatoma*
  - Pedicia*
  - Pilaria*
  - Tipula*

Ephemeroptera:
- Baetidae*
- Ephemerellidae:
  - Ephemerella*
  - Eurylophella*
- Ephemeridae:
  - Ephemera*
- Heptageneidae:
  - Epeorus*
  - Stenonema*
- Leptophlebiidae:
  - Habrophlebia*
- Potamanthidae:
  - Potamanthus*

Lepidoptera:
- Noctuidae*

Megaloptera:
- Sialidae:
  - Sialis*

Odonata:
- Gomphidae:
  - Gomphus*

Oligochaeta*

Table III.1 Continued

Plecoptera:
- Chloroperlidae*
- Leuctridae:
  - Leuctra
- Nemouridae:
  - Amphinemura*
  - Shipsa*
- Peltoperlidae:
  - Tallaperla*
- Perlidae:
  - Eccoptura*
- Perlodidae:
  - Isoperla*

Trichoptera:
- Hydropsychidae:
  - Cheumatopsyche*
  - Diplectrona*
- Hydroptilidae:
  - Palaeagapetus*
- Lepidostomatidae:
  - Lepidostoma*
- Limnephilidae:
  - Neophylax*
  - Pycnopsyche*
- Odontoceridae:
  - Psilotreta*
- Philopotamidae:
  - Dolophilodes*
- Polycentropodidae:
  - Polycentropus*
- Rhyacophilidae:
  - Rhyacophila*

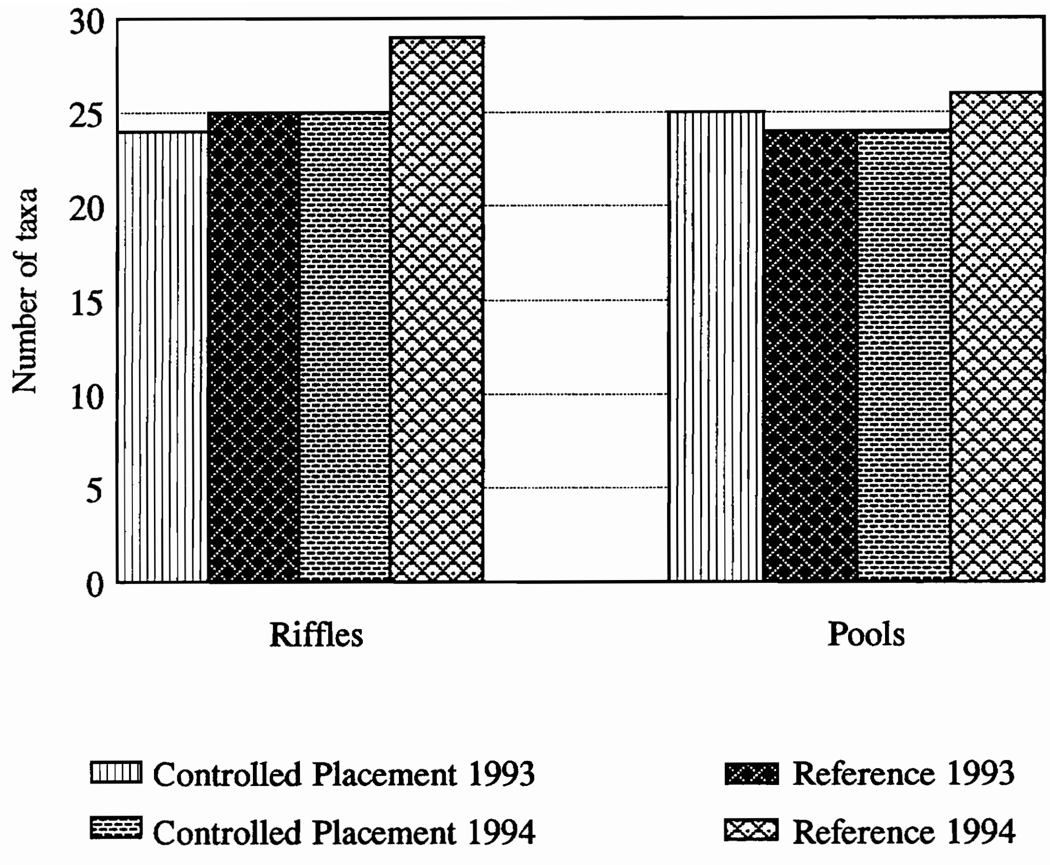

Figure III.1. Stony Creek mean number of taxa for each habitat type, section, and year.

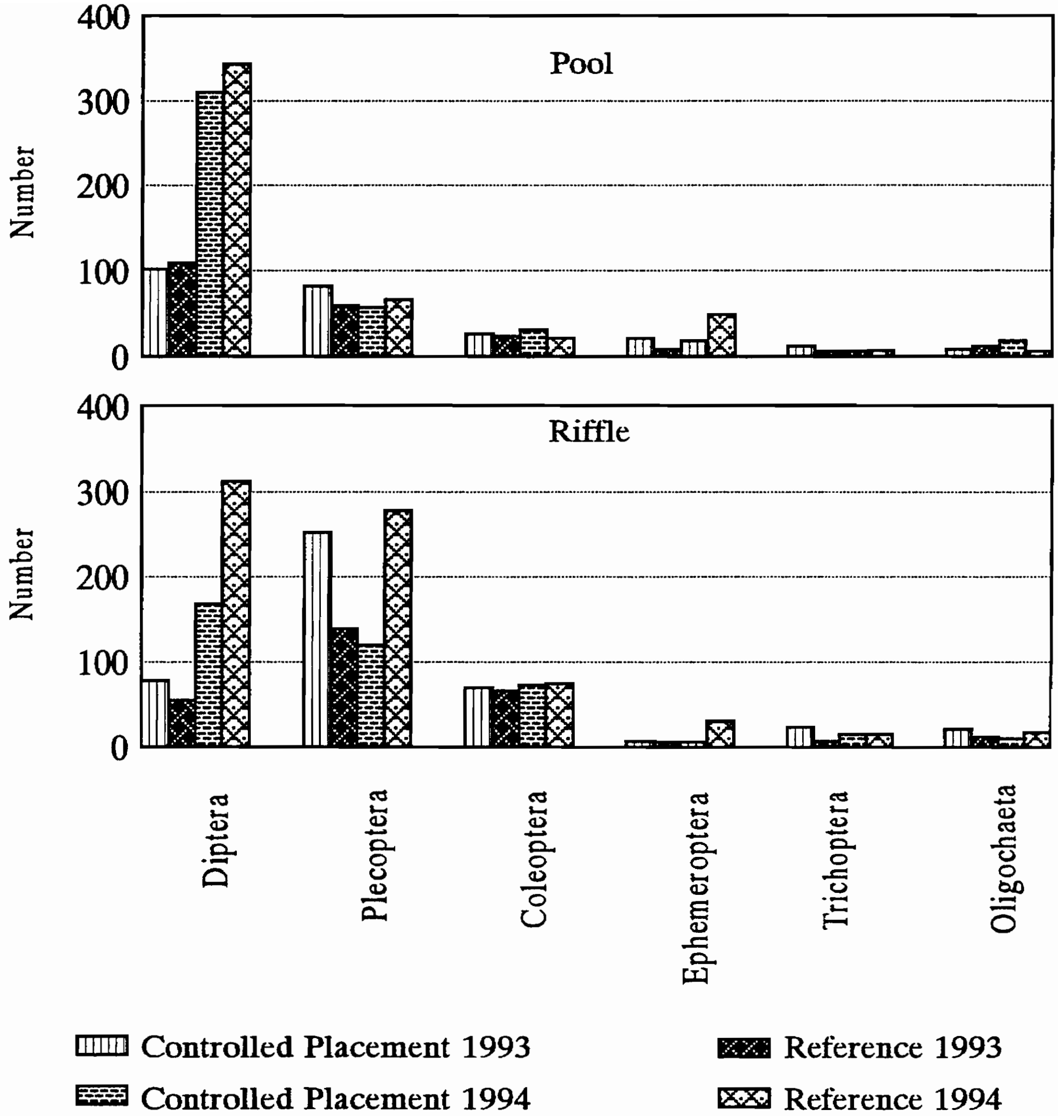

Figure III.2. Stony Creek mean numbers per 0.1 m^2 for common invertebrate groups for each habitat type, section, and year. See Table III.3 for significant effects.

Table III. 2. MANOVA results for the effects of habitat unit type, year, experimental section, and interactions on the common invertebrate group densities in Stony Creek.

Source	df	Wilk's lambda	F ratio	P
Unit	7, 18	0.1448	15.19	<.0001
Year	7, 18	0.3083	5.77	0.0013
Section	7, 18	0.4976	2.6	0.0487
Unit x Year	7, 18	0.8078	0.61	0.7392
Unt x Sect	7, 18	0.7861	0.7	0.6722
Yr x Sect	7, 18	0.4295	3.42	0.0168
Unt x Yr x Sect	7, 18	0.7923	0.67	0.6916

Table III. 3. Coefficients of variation (CV) and ANOVA table of probabilities for the effects of habitat unit type, year, experimental section, and interactions on the common invertebrate group densities in Stony Creek. Coleop = Coleoptera; Ephem = Ephemeroptera; Oligo = Oligochaeta; Pleco = Plecoptera; Trich = Trichoptera.

	Coleop	Diptera	Ephem	Oligo	Pleco	Trich
CV (%)	15.7	14.6	17.4	25.9	11.1	19.6
MODEL	0.0172	0.0111	<.0001	0.2122	0.0004	0.0054
Unit	<.0001	0.0541	0.0002	0.0337	<.0001	0.0027
Year	0.7963	0.0006	<.0001	0.6383	0.7868	0.3612
Section	0.9454	0.5922	0.0798	0.3164	0.9355	0.0148
Unit x Year	0.829	0.9976	0.8858	0.5597	0.4179	0.322
Unt x Sect	0.6854	0.7082	0.1001	0.4283	0.7657	0.6169
Yr x Sect	0.6649	0.3179	<.0001	0.6805	0.0102	0.0381
Un x Yr x Sct	0.7296	0.3391	0.7437	0.1184	0.4709	0.4914

worms were all significantly more abundant in riffles than pools (Table III.3; Figure III.2). Surprisingly, mayflies were significantly more abundant in pools than riffles. The effect of year on the entire assemblage was significant (MANOVA; $P < 0.05$; Table III.2), but barely so, with 1994 abundances generally greater than 1993, but significant only for Diptera and Ephemeroptera (Table III.3). Similar to the effect of year, experimental section had an overall significant effect in MANOVA (Table III.2), but was primarily driven by the greater abundances of Trichoptera in the controlled placement section relative to the reference section. Additionally, the interaction of year and experimental section was significant in MANOVA (Table III.2), and also in ANOVAs (Table III.3) for Ephemeroptera, Plecoptera, and Trichoptera. All three orders experienced a decrease in the controlled placement section through time, but increased in the reference section (Figure III.2).

Functional Feeding Groups

To assess what effect LWD additions might have on the system level processes involving benthic macroinvertebrates, several hypotheses were termed in the form of generic questions. Questions 2, 3, and 4 are applicable not only to functional feeding groups, but also to biomass, potential food for trout, and substratum detritus dynamics which are all covered later. These questions were: 1) Are there differences in abundances of the various functional feeding groups? 2) Does each group associate with pools or riffles? 3) Are there any differences in group abundances between experimental sections? 4) Are there any differences in group abundances between experimental sections after LWD additions (a year x section effect)?

Collector gatherers were the dominant functional feeding group in Stony Creek, followed by shredders, scrapers and collector filterers, and predators (Figure III.3). A MANOVA examining the effects of habitat unit type, year, experimental section, and their interactions on all functional feeding groups simultaneously showed a highly significant unit effect, and significant effects of section, unit x section, and year x section (Table III.4).

Subsequent analyses with three-way ANOVAs helped to explain the MANOVA results. All groups except predators demonstrated a significant association with either pools or riffles. Collector gatherers were significantly more abundant in pools while all remaining functional feeding groups were greater in riffles (Table III.5; Figure III.3). Predators, while not significant (ANOVA $P=0.07$), were more numerous in riffles.

The effect of section in MANOVA was significant (Table III.4), but was not significant in any ANOVA (Table III.5). This indicates a collective difference between sections, but no difference within any individual functional feeding group. There was also a significant year x section effect in MANOVA (Table III.4), driven by collector gatherers, predators, and scrapers (Table III.5).

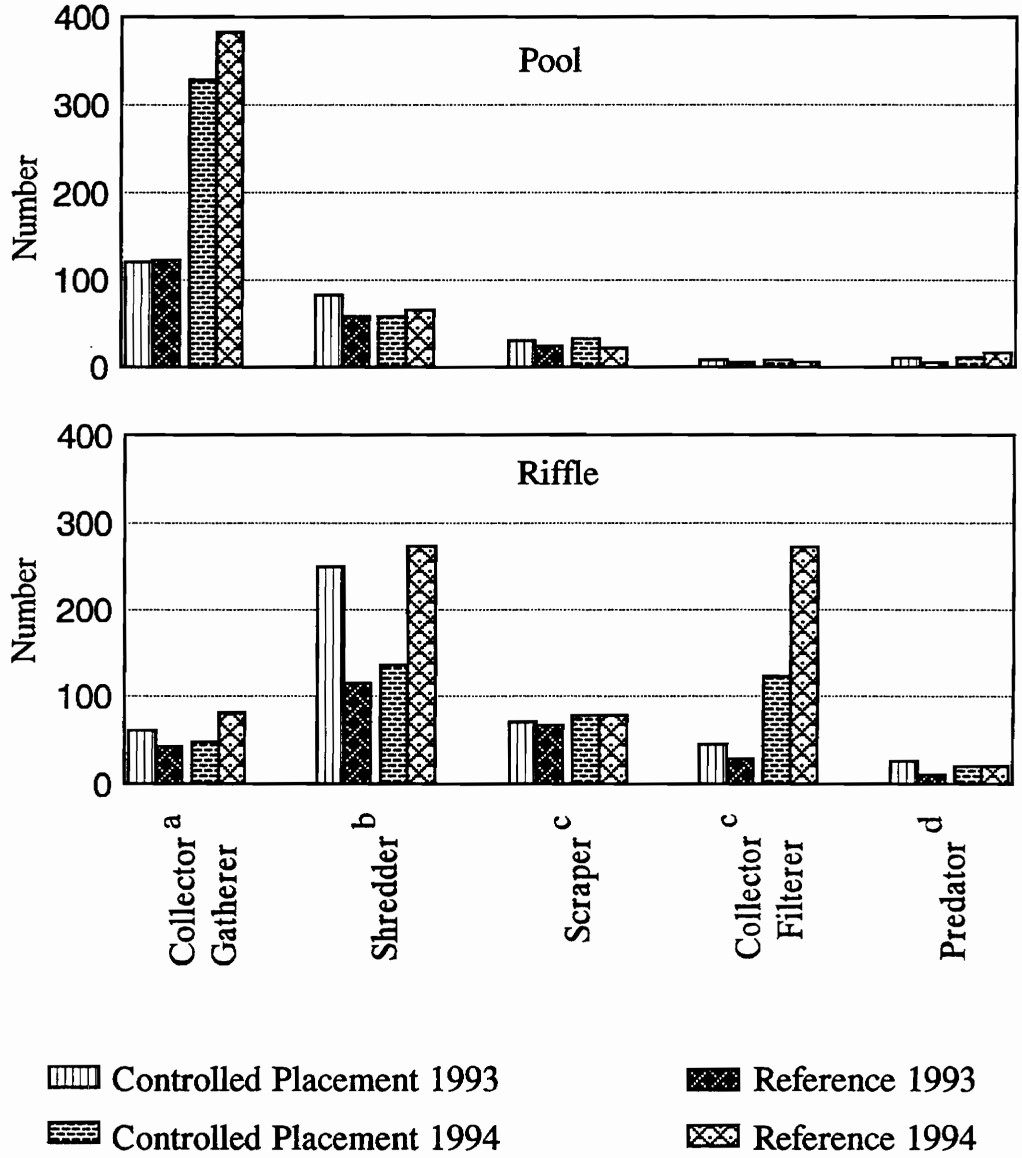

Figure III. 3. Stony Creek mean numbers per 0.1 m^2 for functional feeding groups by habitat type, year and experimental section. Superscripts accompanying groups indicate homogeneity of abundance based on the least significant difference (LSD) multiple comparison procedure. Refer to Table III. 5 for significant effects.

Table III. 4. MANOVA results for the effects of habitat unit type, year, and experimental section, and interactions on functional feeding group densities in Stony Creek.

Source	df	Wilk's lambda	F ratio	P
Unit	5, 20	0.2059	15.5	<.0001
Year	5, 20	0.6908	1.79	0.1606
Section	5, 20	0.5039	3.94	0.012
Unit x Year	5, 20	0.5985	2.68	0.0517
Unt x Sect	5, 20	0.5322	3.52	0.0192
Yr x Sect	5, 20	0.4947	4.09	0.0102
Unt x Yr x Sect	5, 20	0.6469	2.18	0.0969

Table III. 5. Coefficients of variation (CV) and ANOVA table of probabilities for the effects of habitat unit type, year, experimental section, and interactions on the functional feeding group densities in Stony Creek.

	Collector Filterer	Collector Gatherer	Predator	Scraper	Shredder
CV (%)	32.4	10.8	13.6	17.8	14.5
MODEL	0.0003	0.0003	0.0019	0.009	0.0116
Unit	0.0001	0.0001	0.0698	0.0039	0.0001
Year	0.072	0.8323	0.046	0.0659	0.8319
Section	0.9343	0.8466	0.2855	0.1222	0.8654
Unit x Year	0.0748	0.4403	0.0759	0.2165	0.6037
Unt x Sect	0.2845	0.8147	0.3566	0.3107	0.6904
Yr x Sect	0.3285	0.0081	0.0006	0.0276	0.5577
Un x Yr x Sct	0.2029	0.4189	0.2559	0.8146	0.9361

Examination of least squares means (in the SAS GLM procedure) showed an increase in all three groups through time in the reference section, while decreasing in the controlled placement section (Figure III.3). This could imply a treatment effect of LWD additions.

Potential food for trout, biomass, and detritus dynamics

Three-way ANOVA was applied to assess the spatio-temporal dynamics of potential food organisms for trout in Stony Creek. Results show a significant year effect (Table III.6) with densities greater in 1994 (345 items per 0.1 m^2) than 1993 (128 items per 0.1 m^2; Figure III.4). However, controlling for the year effect, no differences existed either between habitat unit types, experimental sections, or interactions.

Similar to potential food for trout, there was no treatment effect of LWD additions on total biomass (minus Chironomidae) per sample. Further, there was no significant effect at any level (Table III.6; Figure III.5). However, even with the data transformed and approximately normal, variability was high (coefficient of variation = 109%).

Substratum detritus retention was significantly greater in pools (9.2g per 0.01 m^2) than riffles (3.1g per 0.01 m^2; Figure III.6), but was not affected through time or by LWD additions (Table III.6). Collector gatherers showed a significant positive correlation with detritus loadings (r = 0.685, n = 64,

Table III. 6. Coefficients of variation (CV) and ANOVA table of probabilities for the effects of habitat unit type, year, experimental section, and interactions on the potential food items for trout, biomass, and detritus in Stony Creek.

	Food Items	Biomass	Detritus
CV (%)	12.6	108.7	30.9
MODEL	0.025	0.142	0.0142
Unit	0.3079	0.2738	0.0003
Year	0.0012	0.1069	0.9382
Section	0.9102	0.304	0.3477
Unit x Year	0.782	0.0948	0.188
Unt x Sect	0.9178	0.7163	0.7777
Yr x Sect	0.1151	0.6463	0.872
Un x Yr x Sct	0.272		

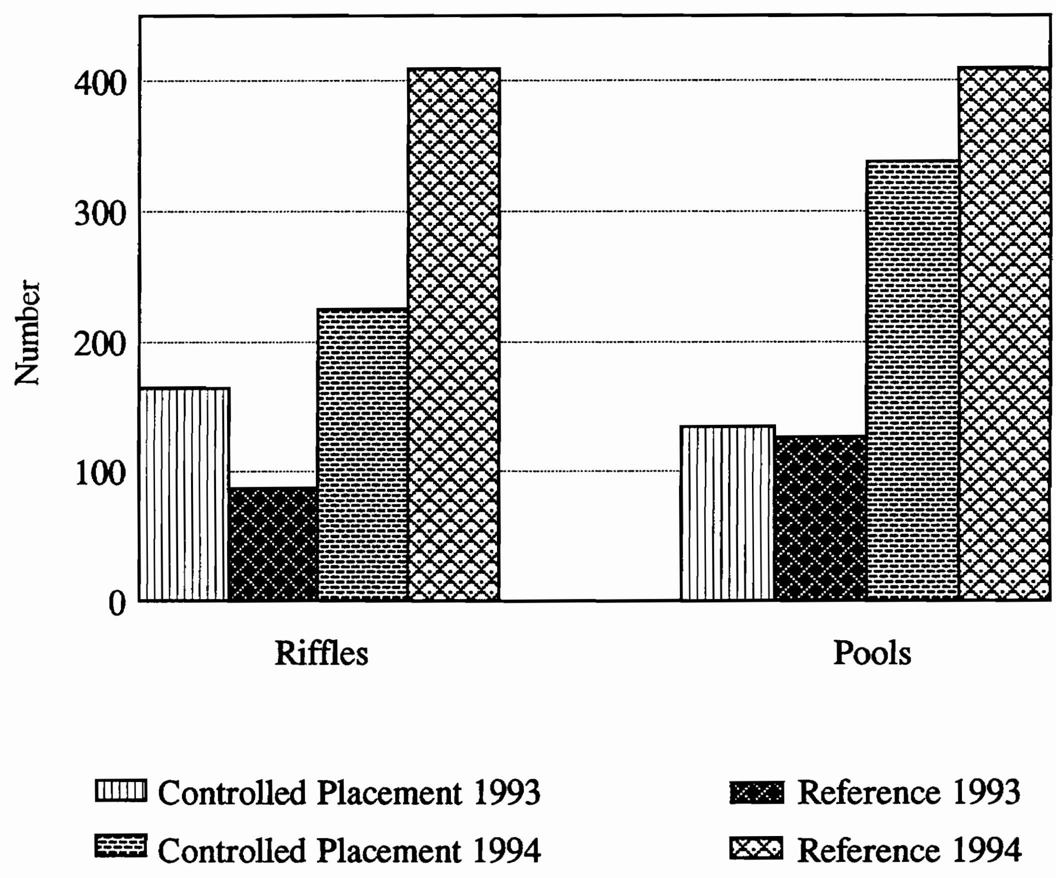

Figure III. 4. Stony Creek mean numbers per 0.1 m^2 for potential food items of trout for each habitat type, year, and experimental section. Refer to Table III. 6 for significant effects.

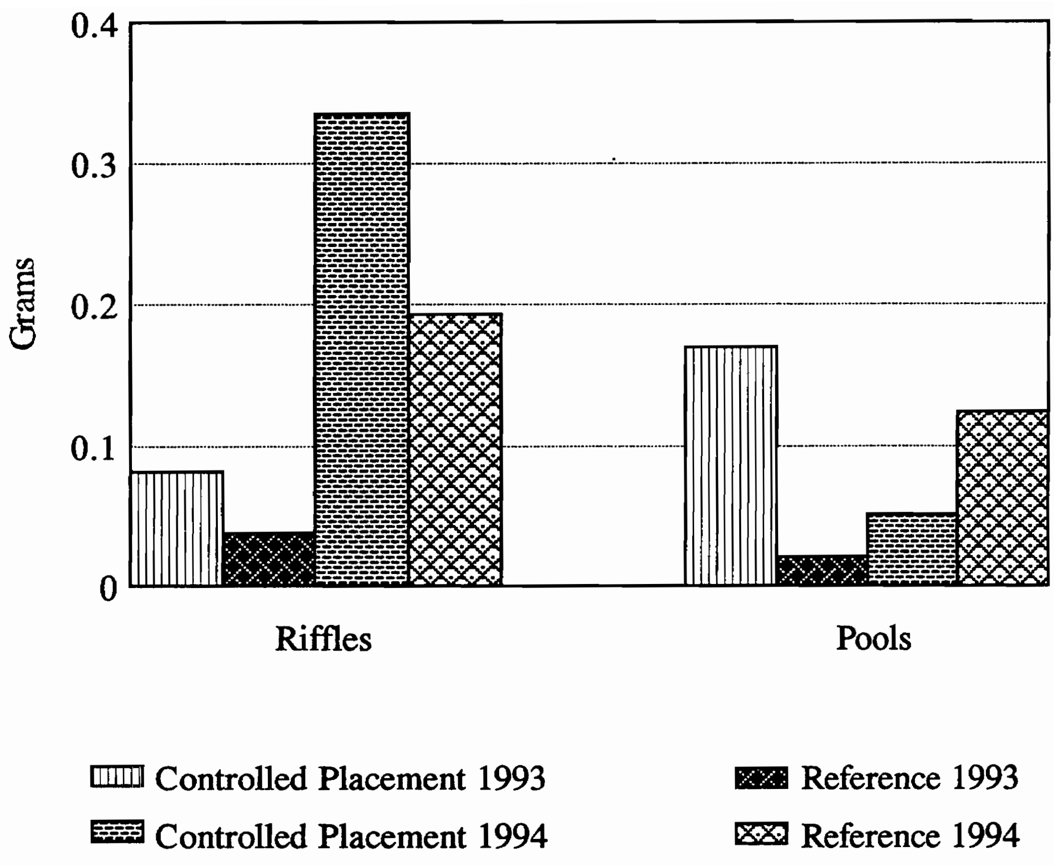

Figure III. 5. Stony Creek mean dry weight (g) per 0.1 m^2 for invertebrate biomass for each habitat type, year, and experimental section. Refer to Table III. 6 for significant effects.

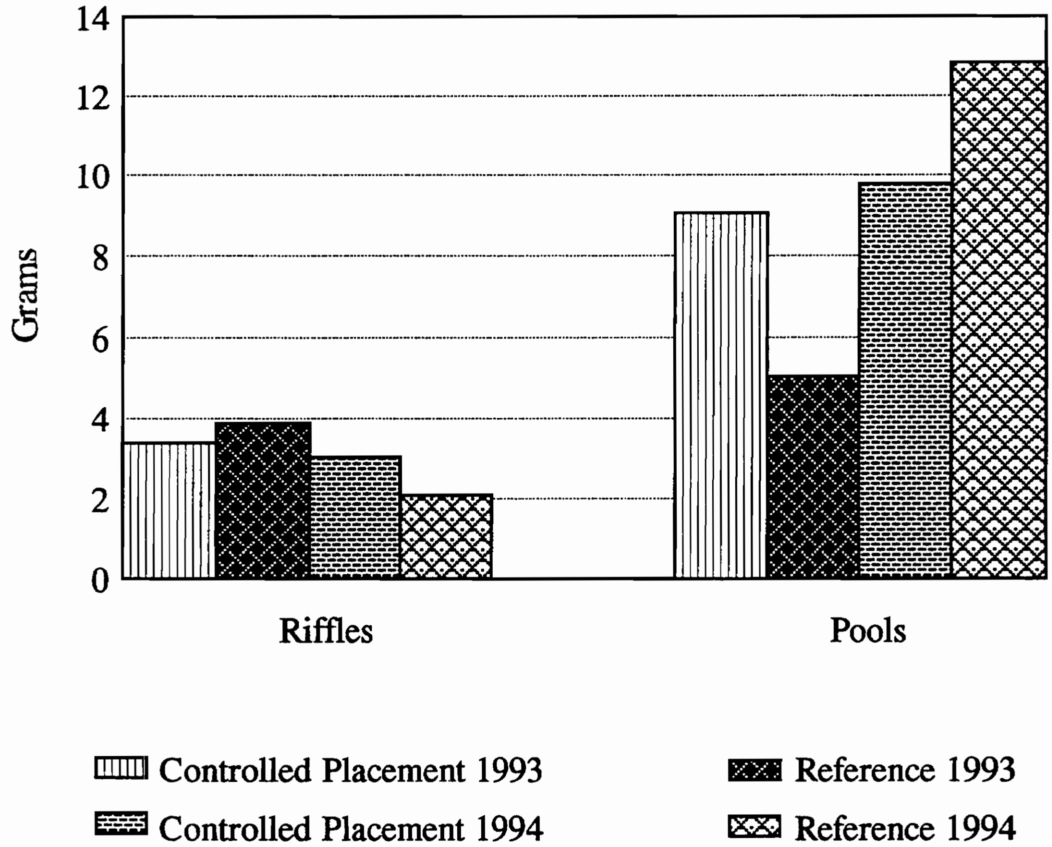

Figure III. 6. Stony Creek mean dry weight (g) per 0.1 m^2 for substratum detritus for each habitat type, year, and experimental section. Refer to Table III. 6 for significant effects.

$P < 0.0001$ and $r = 0.347$, $n = 64$, $P = 0.005$ respectively) while scrapers and shredders were negatively correlated ($r = -0.331$, $n = 64$, $P = 0.0076$; $r = -0.249$, $n = 64$, $P = 0.0474$ respectively). Biomass showed no relation to amount of detritus.

Barbours Creek

Over 29,000 invertebrates representing 62 taxa and 13 orders were identified from spring 1993 and spring 1994 samples of Barbours Creek (Table III. 7). A three-way ANOVA on taxa richness found that riffles contained significantly more taxa than pools ($P = 0.0001$; Figure III.7) while there were no significant differences between years ($P = 0.3050$), or between sections ($P = 0..0887$). In contrast to Stony Creek, mayflies were most abundant followed by beetles, stoneflies, true flies, caddisflies, and worms (Figure III. 8). MANOVA results showed significant overall effects of habitat unit type and year on abundances of the invertebrate orders (Table III. 8). Aquatic Coleoptera, Ephemeroptera, and Oligochaeta were significantly more abundant in 1994 compared to 1993 (Table III. 9; Figure III. 8). Controlling for the effects of year, Coleoptera, Ephemeroptera, Plecoptera, and Oligochaeta were significantly more abundant in riffles and Diptera in pools (Table III. 9; Figure III. 8).

Functional Feeding Groups

Just as with Stony Creek, the same questions involving invertebrate distribution and abundance, biomass, potential food for trout, and substratum detritus were asked in order to asses probable impacts due to LWD additions.

Table III. 7. A list of taxa collected Barbours Creek in the spring 1993 and spring 1994 samples. Asterisks denote taxa included as potential food items for brook trout.

Coleoptera:
- Elmidae:
  - Optioservus
  - Oulimnius
  - Promoresia
- Psephenidae:
  - Ectopria*
- Ptilodactylidae:
  - Anchytarsus
- Staphylinidae

Collembola:
- Entomobryidae*
- Hypogastruridae*

Decapoda:
- Cambaridae:
  - Cambarus*

Diptera:
- Athericidae*
- Blephariceridae:
  - Blepharicera*
- Ceratopogonidae*
- Chironomidae*
- Dixidae:
  - Dixa*
- Empididae*
- Nymphomyidae*
- Simuliidae*
- Tabanidae*
- Tipulidae:
  - Dicranota*
  - Erioptera
  - Hexatoma*
  - Pedicia*
  - Pilaria*
  - Tipula*

Ephemeroptera:
- Baetidae*
- Ephemerellidae:
  - Attenella*
  - Drunella*
  - Ephemerella*
  - Eurylophella*
  - Serratella*
- Ephemeridae:
  - Ephemera*
- Heptageneidae:
  - Cinygmula*
  - Epeorus*
  - Leurocuta*
  - Stenonema*
- Leptophlebiidae:
  - Paraleptophlebia*
- Siphlonuridae:
  - Ameletus*

Hemiptera:
- Gerridae:*
- Saldidae:*

Lepidoptera:
- Noctuidae*

Table III.7 Continued

Megaloptera:
- Corydalidae:
  - Nigronia*
- Sialidae:
  - Sialis*

Odonata:
- Gomphidae:
  - Gomphus*

Oligochaeta*

Plecoptera:
- Chloroperlidae*
- Leuctridae:
  - Leuctra
- Nemouridae:
  - Amphinemura*
  - Shipsa*
- Peltoperlidae:
  - Tallaperla*
- Perlidae:
  - Acroneuria*
  - Eccoptura*
- Perlodidae:
  - Isoperla*
  - Remenus*
- Pteronarcyidae:
  - Pteronarcys*

Trichoptera:
- Glossosomatidae:
  - Glossosoma*
- Hydropsychidae:
  - Cheumatopsyche*
  - Diplectrona*
  - Hydropsyche*
- Lepidostomatidae:
  - Lepidostoma*
- Limnephilidae:
  - Neophylax*
  - Pycnopsyche*
- Philopotamidae:
  - Dolophilodes*
- Polycentropodidae:
  - Polycentropus*
- Rhyacophilidae:
  - Rhyacophila*

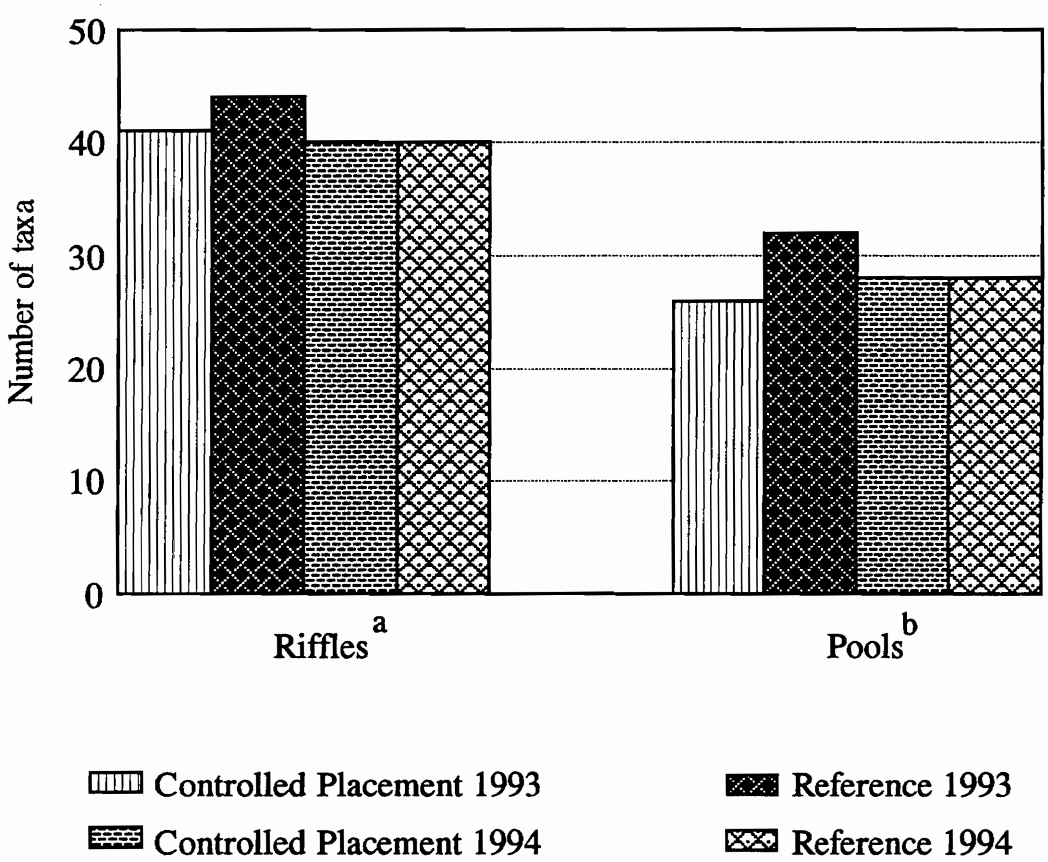

Figure III.7. Barbours Creek mean number of taxa for each habitat type, section, and year. Superscripts indicate significantly different groups.

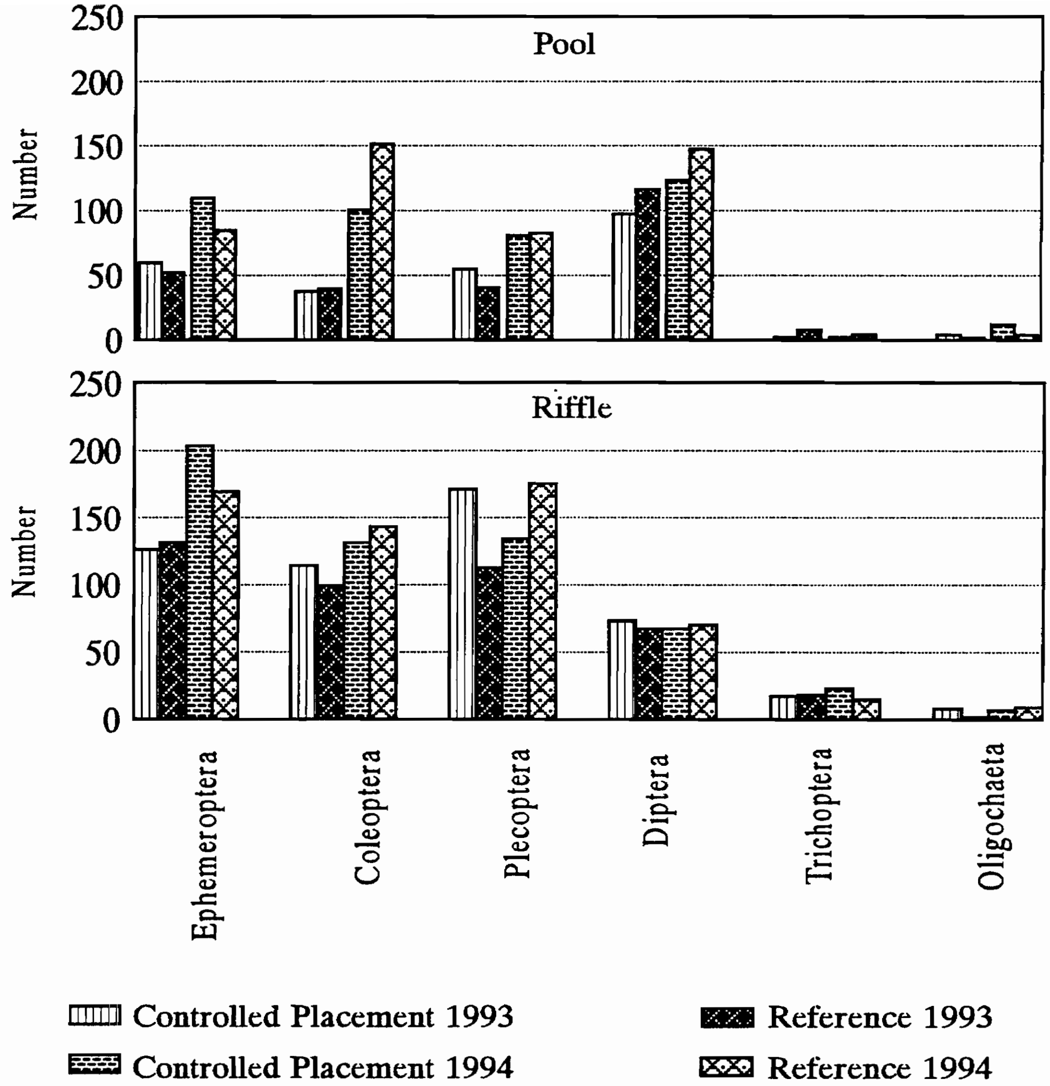

Figure III. 8. Barbours Creek mean numbers per 0.1 m^2 for common invertebrate groups for each habitat type, section, and year. See Table III.9 for significant effects.

Table III.8. MANOVA results for the effects of habitat unit type, year, experimental section, and interactions on the common invertebrate group densities in Barbours Creek.

Source	df	Wilk's lambda	F ratio	P
Unit	7, 18	0.1445	15.22	<.0001
Year	7, 18	0.4211	3.54	0.0145
Section	7, 18	0.5285	2.29	0.0738
Unit x Year	7, 18	0.7934	0.67	0.695
Unt x Sect	7, 18	0.5993	1.72	0.1672
Yr x Sect	7, 18	0.906	0.27	0.9592
Unt x Yr x Sect	7, 18	0.9351	0.18	0.9865

Table III. 9. Coefficients of variation (CV) and ANOVA table of probabilities for the effects of habitat unit type, year, experimental section, and interactions on the common invertebrate group densities in Barbours Creek.

	Coleop	Diptera	Ephem	Oligo	Pleco	Trich
CV (%)	10.8	8.9	8.9	48.9	15.7	24.2
MODEL	0.0088	0.0325	0.0011	0.2239	0.0056	<.0001
Unit	0.0015	0.0005	<.0001	0.3396	0.0008	<.0001
Year	0.003	0.4924	0.0089	0.0464	0.5102	0.3834
Section	0.689	0.3198	0.2503	0.1982	0.4166	0.0801
Unit x Year	0.0887	0.3248	0.8027	0.9667	0.92	0.64
Unt x Sect	0.9778	0.8017	0.5393	0.6291	0.5121	0.0461
Yr x Sect	0.7303	0.7445	0.4767	0.3227	0.73	0.3794
Un x Yr x Sct	0.7955	0.7648	0.709	0.3039	0.6648	0.7863

There were significant differences in abundance between functional feeding groups in Barbours Creek with collector gatherers and scrapers most abundant followed by shredders and predators, and collector filterers (Figure III. 9). MANOVA showed significant habitat unit type and year effects (Table III. 10). Individual three-way ANOVAs showed all groups were significantly more abundant in one habitat type relative to the other. Collector gatherers preferred pools while all other groups were more abundant in riffles (Table III. 11). Subsequent examinations using ANOVAs showed that significant increases in collector gatherers, predators, and scrapers were responsible for the year effect in MANOVA. Addressing questions 3 and 4, there were no significant effects of experimental section or experimental section x year (Table III. 11).

Potential food for trout, biomass, and detritus dynamics

The average number of benthic macroinvertebrates serving as potential food for brook trout was significantly greater in 1994 (298 items per 0.01 m^2) than 1993 (210 items per 0.01 m^2). Additionally, food item abundance in riffles was significantly greater than in pools (Table III. 12; Figure III. 10). However, no other effects were important.

Biomass was no different between pools and riffles or between years (Table III. 12; Figure III. 11). Unfortunately, biomass for 1993 pool samples was not

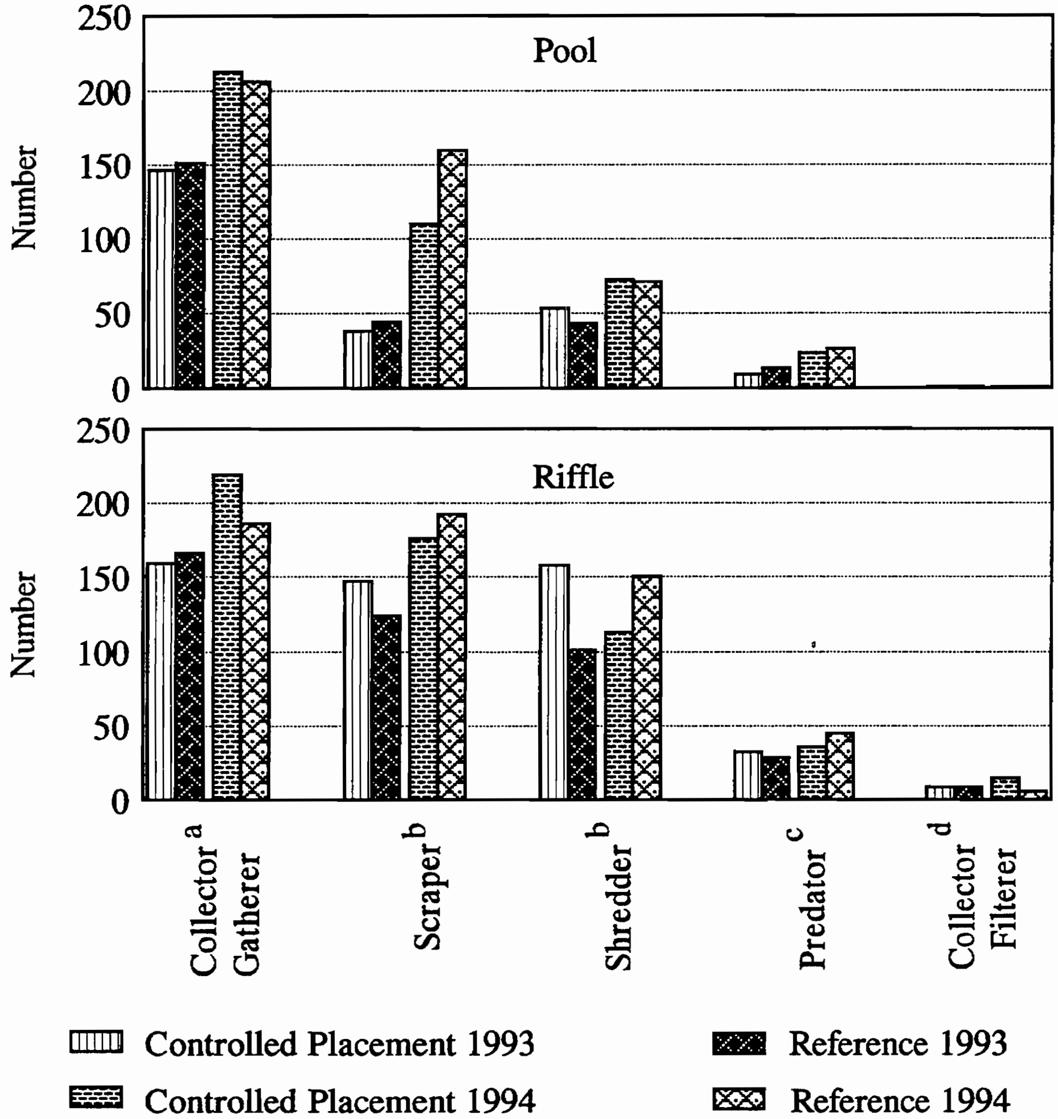

Figure III. 9. Barbours Creek mean numbers per 0.1 m^2 for functional feeding groups by habitat type, year and experimental section. Superscripts accompanying groups indicate homogeneity of abundance based on the least significant difference (LSD) multiple comparison procedure. Refer to Table III.11 for significant effects.

Table III.10. MANOVA results for the effects of habitat unit type, year, and experimental section, and interactions on functional feeding group densities in Barbours Creek.

Source	df	Wilk's lambda	F ratio	P
Unit	5, 20	0.165	20.24	<.0001
Year	5, 20	0.4253	5.41	0.0026
Section	5, 20	0.8433	0.74	0.6002
Unit x Year	5, 20	0.7189	1.56	0.2156
Unt x Sect	5, 20	0.8167	0.9	0.5014
Yr x Sect	5, 20	0.946	0.22	0.9457
Unt x Yr x Sect	5, 20	0.9278	0.31	0.9004

Table III.11. Coefficients of variation (CV) and ANOVA table of probabilities for the effects of habitat unit type, year, experimental section, and interactions on functional feeding group densities in Barbours Creek.

	Collector Filterer	Collector Gatherer	Predator	Scraper	Shredder
CV (%)	51.5	7.7	10.1	9.6	15.9
MODEL	0.0002	0.2095	0.0001	0.0004	0.1202
Unit	0.0001	0.0198	0.0001	0.0001	0.0021
Year	0.4388	0.0713	0.0003	0.0008	0.7668
Section	0.6577	0.7047	0.2787	0.7554	0.5513
Unit x Year	0.9591	0.276	0.0862	0.0774	0.9031
Unt x Sect	0.3654	0.9814	0.5593	0.7739	0.524
Yr x Sect	0.4333	0.8873	0.9913	0.7893	0.7372
Un x Yr x Sct	0.9667	0.6445	0.2249	0.6138	0.6334

Table III.12. Coefficients of variation (CV) and ANOVA table of probabilities for the effects of habitat unit type, year, experimental section, and interactions on the potential food items for trout, biomass, and detritus in Barbours Creek.

	Food Items	Biomass	Detritus
CV (%)	30.2	93.8	44.4
MODEL	0.0429	0.5627	0.0535
Unit	0.0155	0.3453	0.0019
Year	0.0055	0.2924	0.5769
Section	0.9796	0.92	0.299
Unit x Year	0.8047	N/A	0.4823
Unt x Sect	0.6476	0.2789	0.3969
Yr x Sect	0.7194	0.9867	0.8505
Un x Yr x Sct	0.8708		

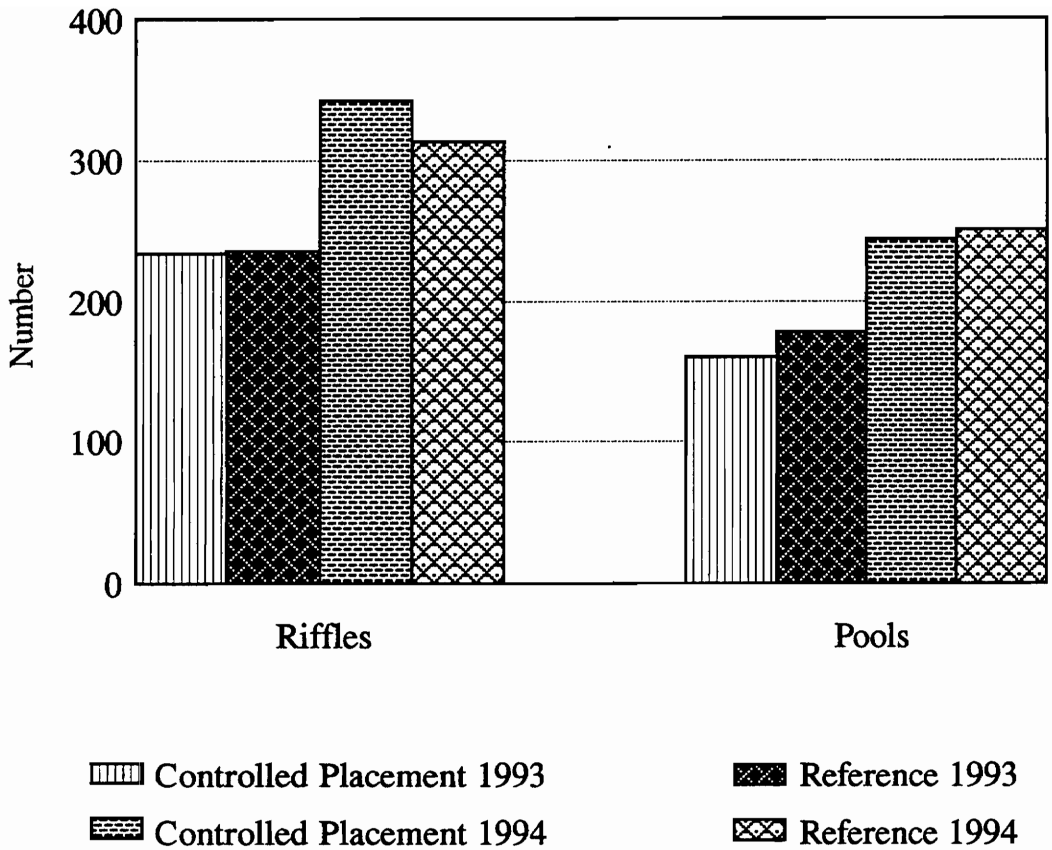

Figure III.10. Barbours Creek mean numbers per 0.1 m^2 for potential food items of trout for each habitat type, year, and experimental section. Refer to Table III. 12 for significant effects.

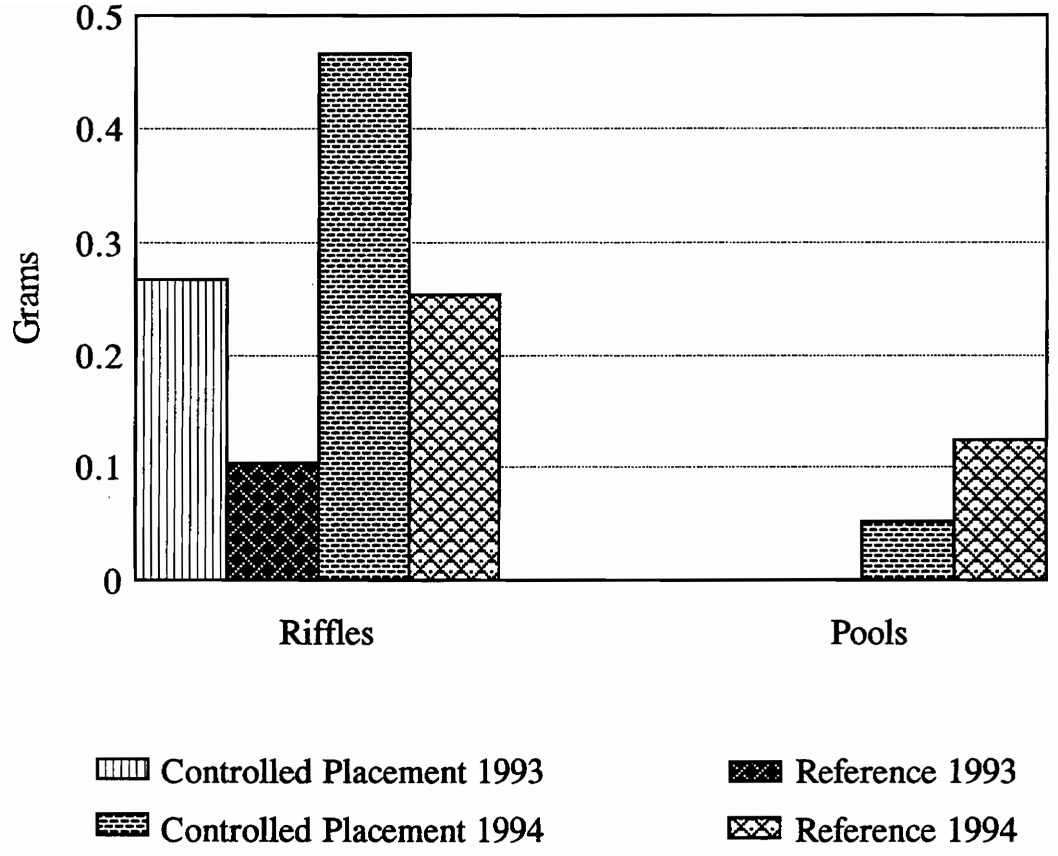

Figure III.11. Barbours Creek mean dry weight (g) per 0.1 m^2 for invertebrate biomass for each habitat type, year, and experimental section. Refer to Table III. 12 for significant effects.

available. However, similar to Stony Creek, biomass was highly variable (C.V. = 93.8%).

Detritus amount was significantly greater in pools compared to riffles, but did not differ between years, sections, or interactions (Table III. 12). In contrast to Stony Creek, no functional feeding groups demonstrated significant relationships to detritus loadings.

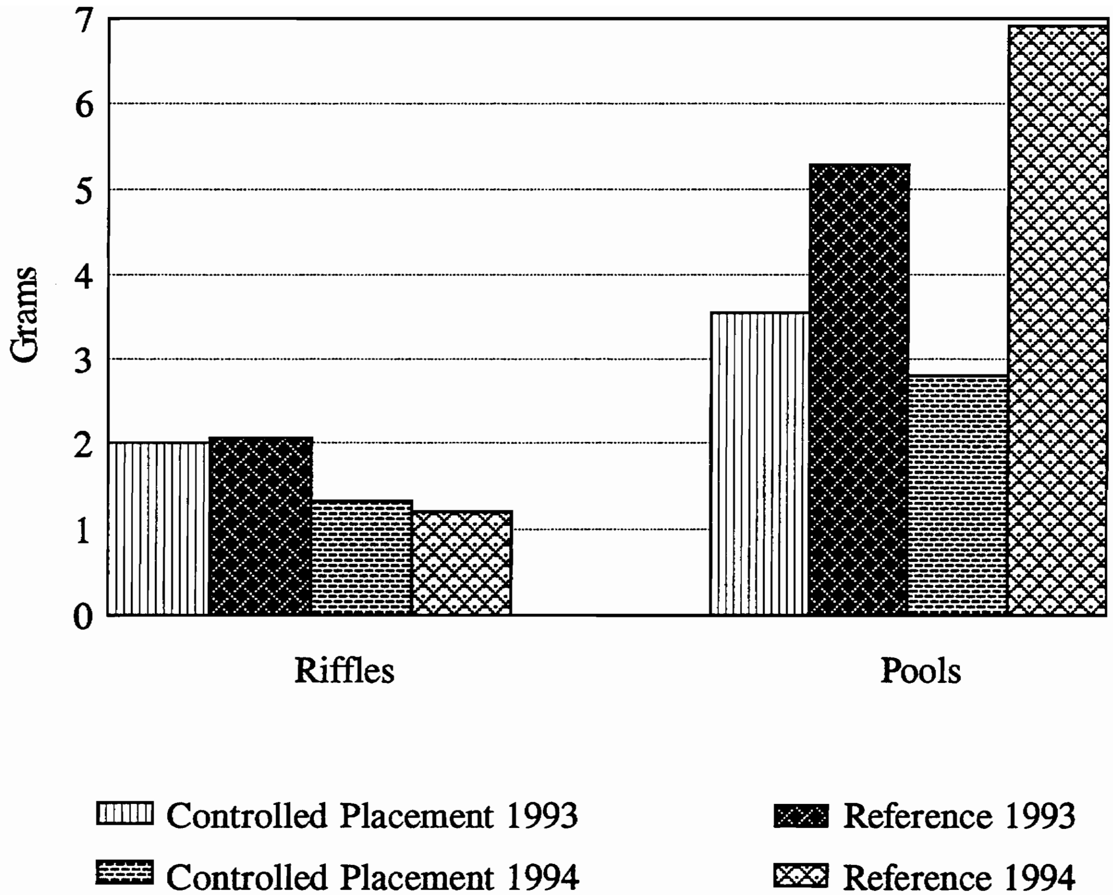

Figure III.12. Barbours Creek mean dry weight (g) per 0.1 m^2 for substratum detritus for each habitat type, year, and experimental section. Refer to Table III. 6 for significant effects.

DISCUSSION

Substratum Detritus Dynamics

Detritus retention in forested streams appears to be greatly enhanced by the presence of LWD. The effect is usually due to debris dams (Bilby and Likens 1980; Speaker et al. 1984; Trotter 1990). Retention may also result from decreased current velocities and deposition in pools created by LWD, although Speaker et al. (1984) reported that riffles trapped leaves more efficiently than pools regardless of substrate type. In the present study, substratum detritus amounts were significantly greater in pools than riffles for both streams. The dominant substrate classes in Stony Creek were small and large gravels while large gravel and cobbles predominated on Barbours Creek. Unfortunately, determining the influence of substrate type on substratum detritus retention was beyond the scope of this study. Personal observation during sampling identified a mixture of several substrate classes in each sampled area. It may have been possible to quantify each sample with pebble counts, but these were impractical for several reasons; most importantly, because too much time would have been spent in quantifying each sample. Additionally, detritus and invertebrates were sampled simultaneously, and the disturbance caused by pebble counts would disrupt the invertebrates unless the counts were conducted while the sampler was already in place. It is also unknown as to whether the uppermost layers of substrata influence retention or the layers underneath, or both. Lumping all layers verses stratification is well beyond the scope of this study.

It was hypothesized that substratum detritus retention would increase in the controlled placement section after LWD additions. Thus we would expect to see a section x year effect. Clearly this did not happen in Barbours Creek. While there was a weak (not significant) effect in Stony Creek, detritus amounts actually decreased slightly in riffles after LWD additions while amounts in pools increased in both experimental sections through time. I attribute this to random chance and have no explanation for the occurrence.

While no treatment effects involving detritus retention were detected, the LWD-induced shifts in total area occupied as pools and riffles may produce dramatic changes at the stream reach scale. Pools in Stony Creek contained roughly twice as much detritus by weight as riffles. Additionally, the total surface area of pools increased substantially in both sections receiving LWD (Chapter 1). The logical extension of this is a huge increase in the amount of detritus retained in the stream substrates because of the increase in pools. Barbours Creek should also follow in this line of reasoning, but to a smaller degree.

Functional Feeding Group Responses

Numerous studies have documented changes in benthic macroinvertebrate community structure in response to changing or contrasting environmental conditions (Woodall and Wallace 1972; Newbold et al. 1980; Haefner and Wallace 1981; Murphy and Hall 1981; Gurtz and Wallace 1984; Wallace et al. 1988). However, community differences in all of these studies are based on a changing background template of ecological succession in terrestrial vegetation influencing the stream. The influence is not only through allochthonous inputs, but also in the amount of sunlight allowed into the stream. In the current study, the background environmental template is held constant while local channel conditions were manipulated. Changes should therefore occur based on habitat associations of taxa and net changes in detritus retention and hydraulic properties of the stream. Changes in the quantity or quality of energy inputs should be incidental.

Increases in pool surface area at the expense of riffles will impact scrapers, and shredders most dramatically in Barbours Creek. Most groups should experience a net loss in total abundance while collector gatherers should increase. However, increases in the amount pool habitat were not very substantial. What impact, if any, this might have on system structure and function is unclear. Superficially, it appears that because densities are greater in riffles, the entire local system will lose some degree of substratum detritus processing capabilities. This might be so. However, a loss of riffle habitat does not mean a loss of all organisms present. It is a proportional loss because limnophilic organisms will replace the riffle dwellers, and all functional feeding groups were represented in pools, just to a lesser degree. The realized impact to processing if pool area increases substantially will probably be in the form of a longer time lag between leaf input and its incorporation into the system. It is doubtful that a small time lag will make much difference because with the increase in pool area comes an increase in detritus retention, and a larger detritus pool to draw from.

Similarly, in Stony Creek all functional feeding groups were closely associated with either pools or riffles. Overall, collector gatherers were the most abundant group and inhabited pools almost three to one over riffles. In contrast, shredders were the next most abundant group and numerically dominated riffle habitats almost four to one over pools. In the treatment sections where LWD additions substantially increased pool surface area at the expense of riffle area, we should expect to see a net decrease in abundances in all groups except collector gatherers. The most pronounced decrease will occur for shredders, but will also substantially reduce scraper and collector filterer numbers. In contrast, collector gatherers should experience a large net increase.

Theoretically, the decrease in shredders could have major impacts on the current assemblage structure because shredders are the initial component in processing allochthonous detritus inputs. Cuffney and Wallace (1989) reported a dramatic decline in FPOM export levels after shredder abundance was decimated by insecticide application. While the decrease was exacerbated by drought conditions, invertebrate removal accounted for 75% of the reduction. This decrease might potentially cause food limitations for the other functional groups and result in an overall decrease in abundance. Richardson and Neill (1991) reported increases in collector gatherer and collector filterer densities in response to increased inputs of whole leaf detritus in experimental stream channels. While they attributed the increases to higher survival, it could also be due to a lifting of food limitations. Even if this true, any effects might take several years to detect because the amount of FPOM stored in the stream channel is much greater than the amount exported (Cuffney and Wallace (1989). Thus even with a time lag introduced, a reduced shredder population and mechanical abrasion of leaf particles should be sufficient to maintain the FPOM reservoir. The River Continuum Concept (RCC) (Vannote et al. 1981) predicts shredder dominated assemblages that are driven by allochthonous detritus inputs in small, forested streams such as Stony and Barbours Creeks. While more an heuristic model, RCC is still a useful starting point in forested watersheds. By default, RCC implies a positive relationship between detritus and shredders, with the remainder of the processing community (functional feeding groups) adapted to exist on incomplete energy transformation by shredders. Stony Creek was dominated by shredders. Paradoxically, they were significantly more abundant in riffles than in pools, but pools contained significantly more detritus than riffles.

Relationships between detritus and benthic macroinvertebrates are not at all clear. Insect abundances and detritus amount were positively related for most insects in studies by Egglishaw (1964) and Minshall and Minshall (1977). Hildrew et al. (1991) reported a positive relationship for shredders while Townsend and Hildrew (1984) reported all other functional feeding groups declined in more highly retentive areas. Conversely, no relationship could be determined by Petersen and Cummins (1974). The results in this study are mixed. In Barbours Creek, all functional feeding groups were negatively correlated with detritus, but not significantly. In Stony Creek, collector gatherers demonstrated a significant positive relationship with detritus, predators showed no correlation, and shredders, scrapers, and collector filterers a significant negative relationship. It appears that more than detritus amount is driving these systems. Other possible factors could be due to differences between pools and riffles in dissolved oxygen levels, detritus nutritional quality, amount of fine sediments, biotic interactions, or catastrophic abiotic factors such as floods.

The majority of shredders were stoneflies, which as a group were significantly more abundant in riffles than pools. One explanation for the seeming contradiction is that riffles may contain higher quality detritus (food) than pools. A large portion of the detritus in pools appeared to be broken down into very small fragments relative to riffles, and during sampling, I observed a large amount of detritus/silt passing through the collection net. Possibly the size fraction in pools is too small to be of use to shredders, or the smallest size fraction somehow inhibits feeding or respiration by shredders and the other groups negatively related to detritus amount. Possibly detritus is just a surrogate for an unmeasured habitat characteristic. Another possibility is that the sampling protocol excluded shredder habitats. I did not sample leaf packs. However, sometimes large amounts of whole leaf detritus were collected in pool samples, and these would figure into the final analysis.

Responses of Taxa

Both Stony Creek and Barbours Creek contain very diverse benthic macroinvertebrate assemblages. With genus as the finest level of resolution, richness in Stony Creek was 47 distinct taxa, and 11 groups were identified only to family level. Although most taxa preferred either pools or riffles, there was no difference in the total number of taxa inhabiting pools and riffles.

Conversely, Barbours Creek contained 62 distinct taxa, and riffles contained significantly more taxa than pools. At the taxonomic level of Order, both streams were identical both in the number of orders and members. At such a coarse scale of resolution, and allowing for the underlying diversity and adaptation of aquatic insects in each Order, one might expect a convergence in community structure between streams. However, this was not at all the case. Stony Creek was clearly dominated by Diptera while Ephemeroptera and Coleoptera were most abundant in Barbours Creek. While all taxa were represented in both habitat types, like the functional feeding groups, all taxa except Oligochaeta were associated with either pools or riffles. Diptera preferred pools in both streams while all other taxa selected for riffles. In terms of biodiversity, it is doubtful that habitat manipulations will have any effects because pools and riffles will still exist, only the relative proportions will change.

In both streams in the current study, Coleoptera was numerically a very important component in the assemblage, with most of its members from the family Elmidae. What importance these might be to the overall community functions is not known although their feeding strategies are scrapers and collector gatherers (Merritt and Cummins 1984). While more abundant in riffles, a fair number were also collected in pools, and those that are collector gatherers may serve as an important transitional link in detritus processing between habitat types.

Production and Prey Items for Trout

While proportional shifts in the abundance of functional feeding groups or individual taxa will probably occur as a result of LWD additions, the system itself will most likely remain resilient because the components are still there. Nonetheless, depending on the magnitude, changes in abundance of taxa may have real effects on their vertebrate predators. Although invertebrate abundances differed markedly based on habitat types, biomass did not differ between pools and riffles, between sections, or between years in either stream. In some aspects this may be misleading because one large invertebrate such as a crayfish or predatory stonefly might weigh more than several dozen small individuals. However, biomass might be a better indicator because one large prey item may be more important energetically than many small items. Although biomass varied considerably in this study, the sample size was probably large enough to imply that the variability is natural to these streams, and probably more a reflection of site specific local conditions than a predictable reach level pattern.

The number of potential food items for brook trout varied considerably through time for both streams. However, only Barbours Creek showed any differences between pools and riffles. There, riffles contained significantly more potential food items than pools. In Stony there was a weak section x year effect, but did not appear to result from LWD additions. It was hypothesized that riffles would be the major food production sites for trout, and reduction of riffle habitats would ultimately reduce the food base for trout. The data indicate that this could happen in Barbours Creek if enough riffle area was lost. However, in Stony Creek, there is no evidence to support this hypothesis, and habitat manipulations were most extreme in Stony Creek. It must be cautioned that this study sampled the communities less than one year after LWD additions. It is quite possible that a time lag on the order of years may result in decreased production as was suggested earlier. While it may be difficult to partition the annual variability in invertebrate abundances to detect community shifts, sampling should be continued for the duration of this decade-long research project because one year of monitoring is simply not long enough.

The diet of brook trout usually parallels the taxa that predominate in the invertebrate drift (Allan 1981; Fechney 1988) although there is also evidence that a large portion of the diet may come from non-drifting bottom forms (Warren et al. 1964; Morgan and Robinette 1978). For this study, invertebrates included in the analyses as potential food for trout were compiled from several diet studies (Allan 1978; Morgan and Robinette 1978; LaRoche 1979; Fechney 1988; Hubert and Rhodes 1989). Even so, there was still some subjectivity involved in deciding what taxa should be included because trout diet was not examined in this study. For example, Hubert and Rhodes (1989) reported heavy use of Elmidae (Coleoptera) larvae and adults in a subalpine stream in Wyoming. Conversely, Elmidae were not even consumed in a detailed study by Allan (1978), and very rarely in the other studies. I did not include Elmidae in the analyses because only the one study (in a very different aquatic environment) reported their use. Elmidae constituted a large portion of many of my samples, and would greatly bias food availability results.

While the replacement of riffle habitats by pools may potentially decrease the availability of food items for trout, there is a tradeoff involved that may actually benefit trout. A large portion of a brook trout's summer diet comes from terrestrial insects (Hunt 1975; Light 1981; Ensign 1989; Meyer 1990). In the North Fork Tye River, Virginia, LaRoche (1979) reported that in over 60% of native brook trout diets in July-September were terrestrial invertebrates. Similarly, Morgan and Robinette (1978) found that terrestrial insects comprised 64%, 95%, 93%, and 56% of the total weight in diets of age classes 0, I, II, and III brook trout respectively in Great Smoky Mountain National Park streams. Age Class III brook trout also fed heavily on crayfish. A major characteristic of pools is slow current velocities relative to riffles. Lower current velocities in pools will slow the downstream export of terrestrial organisms and drift, and increases the probability that a food item, once detected, will be captured (Hughes and Dill 1990). Thus a reduction in riffles and food production may be offset by more efficient feeding by trout because of increased pool area. Additionally, the portions of LWD not covered by water may serve as a bridge, foraging site, or colonization substrate for terrestrial invertebrates, increasing the chance that prey items will fall and enter the drift.

Summer and early autumn are often periods of low flow conditions in the Appalachians because of decreased precipitation. LaRoche (1979) found that drought adversely affected brook trout populations in central Virginia streams. During these periods, current velocities and invertebrate drift are reduced, resulting in increased utilization of terrestrial invertebrates. The decrease in surface area due to flow reductions is generally less for pools than other habitats (Kraft 1972). Additionally, pools are usually deeper than riffles, increasing not only total volume available for use, but also resistance to dewatering relative to riffles. This increases both the importance of pools as potential refugia from drought, and the importance of LWD in the creation and maintenance of these refugia.

Summary and Conclusions

Large woody debris pieces oriented as ramps were most effective at scouring the stream channel, especially along the channel margins. Dams produced scour patterns in mid-channel downstream from logs, and angled dams caused filling along the channel margins. All pools created by LWD additions were formed by logs oriented as dams despite the fact that ramps produced more scour. However, scour around ramps was localized and is expected to increase through time to form additional pools.

Large woody debris additions increased both the number of pools and the total area occupied by pools at the expense of riffles in Stony Creek. The increased pool area was most pronounced in the controlled placement section lending support to the notion that human judgement in the placement of LWD can be more effective at manipulating stream habitats than by placing logs according to a randomized design. Large woody debris additions increased habitat heterogeneity at the scale of the stream reach and decreased the average length between habitat unit types.

In contrast to Stony Creek, Barbours Creek changed little after LWD additions although change did occur compared to the reference section. Additionally, there was little difference in channel changes between the random and controlled placement sections. It is hypothesized that the lack of stream channel response was due to lower loadings of LWD relative to Stony Creek, topographic constraints in placing logs, or higher gradient and larger channel substrates than Stony Creek.

Substratum detritus was greater in pools than riffles for both streams, but there was no difference between years, sections, or as a result of LWD additions. However, because of the dramatic increase in total pool surface area, it is hypothesized that detritus retention will increase substantially in Stony Creek.

Most functional feeding groups and invertebrate orders were significantly more abundant in riffles than pools. The exception were collector gatherers which accounted for a large proportion of the benthic macroinvertebrate assemblage. The relations between detritus amount and functional feeding groups were generally negative, but may be an artifact of higher abundance in riffles. Increased pool area at the expense of riffle area may decrease the overall relative abundances of functional feeding groups and result in a reduction of the stream's substratum detritus processing capabilities. However, it is hypothesized that mechanical and microbial processes should compensate for a loss of shredders, but a time lag in energy flow through the system may develop.

Decreased riffle area should result in an overall decrease in the abundance of potential food items for brook trout in Barbours Creek, but not in Stony Creek. However, average biomass was not significantly different between pools and riffles for either stream. Since benthic macroinvertebrate abundance was higher in riffles, it follows that the average weight of individual invertebrates is greater in pools. This may compensate for a loss of prey items of trout by increasing the average size of a prey item. Additionally, increased pool area may compensate for decreased prey abundance by providing areas of lower current velocities which increases the retention times of terrestrial and aquatic invertebrate drift in the water column. This may be important during times of low flow or drought conditions where pools also serve as refugia for brook trout. Thus the creation and maintenance of pools by additions of LWD may be an effective tool to enhance trout habitats in Virginia streams, especially low gradient streams.

Management Considerations

Barbours Creek is an A-3 channel (Rosgen 1985) of moderately high gradient with cobble and gravel substrates. This channel type is generally not recommended for receiving fish habitat improvement structures because habitat is usually not limiting (Rosgen and Fittante 1986). Considering this in light of the modest channel manipulation results of the present study, LWD might best serve as fish cover rather than for manipulating channel features in this type of stream channel. Additionally, with the amount of manpower and heavy equipment involved, the cost will most likely outweigh the benefits. Therefore, management of the riparian zone for LWD recruitment may be a more economical alternative.

In contrast to Barbours Creek, the desired channel manipulation results were being realized within one year in Stony Creek, whose channel was nearly one long riffle before LWD additions. Stony Creek is a G-4c channel type. This type is of low gradient (< 2%) with relatively small and mobile substrates (gravels). While not listed in Rosgen and Fittante (1986), approximating the channel type with a B-4 channel results in recommendations against most types of fish habitat improvement structures generally because of channel instability and the resulting ineffectiveness. In these channels, LWD additions may be feasible because the pieces can move and adjust to local channel conditions, unlike permanent habitat improvement structures. All stream habitat enhancement projects are merely bandages - not cures. Stream habitat restoration ultimately depends on protection of the riparian zone. However, in streams like Stony Creek, LWD additions may temporarily maintain stream channel complexity until there is sufficient debris recruitment from the riparian zone.

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Appendix A.

In order to calculate the Cartesian coordinates for the stream and its banks, the coordinates of each reference stake must first be calculated. The reference stake fartherest downstream (at the bottom of the study section) was designated the origin and assigned the coordinates 0,0. Please refer to Figure Appendix A.1 for a graphical layout.

Given:

Ψ = compass angle of the transit line relative to magnetic north
a = distance along the transit line
b = length of the perpendicular transect line
d = distance from one reference stake to the next

The compass angle of the transit line (Ψ) relative to magnetic north was transformed from the compass to polar system using:

If $\Psi < 90$ then $\lambda = 90 - \Psi$ OR If $\Psi > 90$ then $\lambda = 450 - \Psi$

Knowing the straight line distance to the next reference stake (d) and the angle of that line (λ) in degrees, the reference stake Cartesian coordinates (X_R, Y_R) were calculated using

$$X_R = d \cos\lambda$$
$$Y_R = d \sin\lambda$$

Although the straight line distance between stake 1 and 2 is known, the distance between 1 and 3 is not. The coordinates of stakes 3, 4, 5...n were calculated iteratively by temporarily assigning the stake immediately downstream as the new origin. Thus the coordinates of each reference stake are relative to the stake immediately downstream. Starting at the downstream most stake (0,0) and moving upstream, the coordinates of a particular stake were determined by adding its relative coordinates to the sum of the relative coordinates of every stake downstream.

Calculation of stream edge and bank coordinates are similar to those of the reference stakes with a few extra steps. A point Q with coordinates X_Q, Y_Q located perpendicular to the transit line at distance a, and removed from the transit line with distance b can be calculated using the following:

$$\Phi = \arctan b/a$$

and

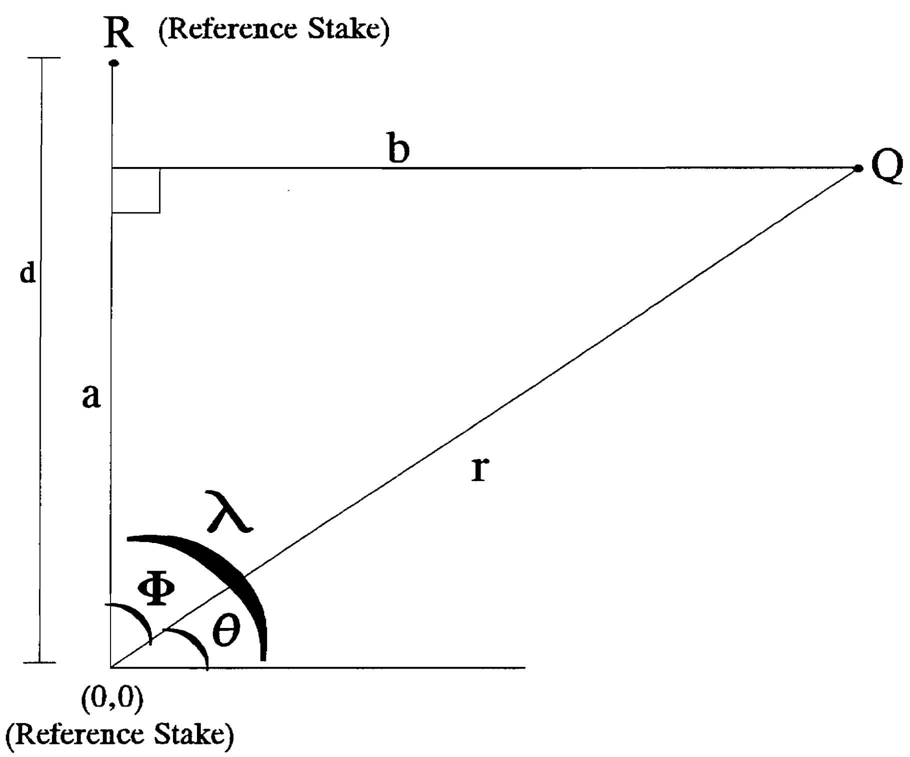
$$\theta = \lambda - \Phi$$

Since the transit and transect lines are perpendicular,

$$r = \sqrt{a^2+b^2}$$

where r = the straight line distance (hypotenuse) from the reference stake to point Q. Then,

$$X_Q = r \cos \theta$$
$$Y_Q = r \sin \theta$$



Appendix B Stony Creek means and standard deviations (std) for all taxa collected in riffles and pools for the 1993 and 1994 samples.

	Riffle				Pool			
	1993		1994		1993		1994	
	Mean	std	Mean	std	Mean	std	Mean	std
COLEOPTERA								
Anchytarsus	0.0	0.0	0.1	0.2	0.1	0.3	0.0	0.0
Optioservus	0.8	1.5	0.3	0.7	0.3	0.8	0.0	0.0
Oulimnius	33.5	21.9	23.9	18.4	21.3	11.7	22.6	18.4
Oulimnius adult	25.8	18.2	37.7	25.0	0.8	1.6	1.4	2.1
Promoresia	4.9	6.8	7.2	11.4	1.4	2.3	1.4	2.0
Promoresia adult	3.3	4.5	4.2	7.0	0.6	1.7	0.4	0.9
COLLEMBOLA								
Entomobryidae	0.1	0.2	0.1	0.3	0.0	0.0	0.5	0.7
Hypogastruridae	0.1	0.2	0.0	0.0	0.0	0.0	0.0	0.0
DECAPODA								
Cambarus	0.7	0.8	0.9	0.9	1.4	1.2	3.3	3.6
DIPTERA								
Ceratopogonidae	0.4	1.6	0.3	0.5	0.4	0.7	5.3	6.0
Chironomidae	29.6	16.6	35.6	29.2	98.4	43.1	306.7	155.2
Dicranota	0.9	1.2	1.2	1.5	0.1	0.3	0.2	0.4
Dipteran pupae	0.6	0.8	4.9	6.2	1.8	1.3	6.9	5.2
Empididae	0.2	0.4	0.1	0.4	0.3	0.5	0.6	0.7
Hexatoma	0.3	0.6	0.6	0.8	0.3	0.5	0.8	1.2
Nymphomyidae	0.0	0.0	0.1	0.3	0.0	0.0	0.0	0.0
Pedicia	0.2	0.5	0.8	1.8	0.4	0.9	0.1	0.3
Pilaria	0.6	0.7	1.0	1.5	1.3	0.9	2.2	2.8
Simuliidae	34.2	34.5	195.5	194.6	1.4	1.0	3.1	2.8
Tabanidae	0.0	0.0	0.0	0.0	0.3	0.7	0.0	0.0
Tipula	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.6
EPHEMEROPTERA								
Baetidae	4.5	4.2	13.0	15.9	0.3	0.5	0.4	0.8
Epeorus	0.0	0.0	2.1	2.2	0.0	0.0	0.0	0.0
Ephemera	0.0	0.0	0.0	0.0	0.2	0.4	0.0	0.0
Ephemerella	0.1	0.2	0.0	0.0	0.1	0.3	0.0	0.0
Eurylophella	0.2	0.5	0.2	0.4	1.8	1.3	18.1	20.4
Habrophlebia	0.9	1.8	0.9	2.1	10.1	8.4	13.2	9.5
Potamanthus	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.7
Stenonema	0.6	0.9	2.2	2.8	2.3	2.6	1.3	1.5
LEPIDOPTERA								
Noctuidae	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.3

Appendix B Continued

	Riffle				Pool			
	1993		1994		1993		1994	
	Mean	std	Mean	std	Mean	std	Mean	std
MEGALOPTERA								
Sialis	0.0	0.0	0.0	0.0	0.5	1.2	1.9	3.2
ODONATA								
Gomphus	0.2	0.4	0.2	0.7	0.0	0.0	0.1	0.3
OLIGOCHAETA	16.2	10.1	13.6	12.3	9.9	10.4	11.9	16.1
PLECOPTERA								
Amphinemura	31.3	29.3	29.7	20.8	1.0	1.0	1.5	3.2
Chloroperlidae	3.5	3.0	2.7	2.7	0.6	1.1	1.3	1.6
Eccoptura	0.7	0.9	1.0	1.5	0.6	0.9	0.2	0.4
Isoperla	0.1	0.3	0.4	0.6	0.1	0.3	0.0	0.0
Leuctra	147.7	89.6	156.9	125.2	68.3	36.2	57.6	38.1
Shipsa	0.3	0.7	7.5	9.0	0.2	0.6	0.9	1.8
Tallaperla	1.9	2.9	10.1	10.5	0.2	0.4	0.1	0.3
TRICHOPTERA								
Cheumatopsyche	0.0	0.0	0.9	1.2	0.0	0.0	0.0	0.0
Diplectrona	1.5	1.9	0.9	0.9	0.2	0.6	0.3	0.9
Dolophilodes	0.1	0.2	0.1	0.3	0.0	0.0	0.0	0.0
Lepidostoma	0.8	1.2	0.2	0.4	4.5	4.1	3.6	3.7
Neophylax	0.3	0.6	0.5	1.0	0.3	0.7	0.0	0.0
Palaeagapetus	0.3	0.7	0.2	0.5	0.0	0.0	0.1	0.3
Polycentropus	1.6	1.3	2.1	2.8	1.9	2.0	0.4	0.9
Psilotreta	0.0	0.0	0.0	0.0	0.3	0.7	0.3	0.6
Pycnopsyche	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.3
Rhyacophila	10.1	11.8	9.7	6.5	0.7	0.7	0.8	1.1

Appendix C Barbours Creek means and standard deviations (std) for all taxa collected in riffles and pools for the 1993 and 1994 samples.

	Riffle				Pool			
	1993		1994		1993		1994	
	Mean	std	Mean	std	Mean	std	Mean	std
COLEOPTERA								
Anchytarsus	0.1	0.2	0.1	0.2	0.0	0.0	0.0	0.0
Ectopria	1.0	1.1	2.3	2.2	1.5	1.2	2.5	2.3
Optioservus	1.9	3.1	13.2	14.9	10.7	11.5	5.2	5.7
Optioservus adult	0.1	0.2	0.2	0.4	0.0	0.0	0.0	0.0
Oulimnius	88.0	57.7	96.7	54.0	25.3	14.8	112.1	99.2
Oulimnius adult	15.1	9.3	24.5	19.4	0.5	1.0	5.3	8.3
Staphlynidae	0.1	0.2	0.2	0.4	0.1	0.3	0.0	0.0
COLLEMBOLA								
Entomobryidae	1.2	1.9	0.3	0.7	0.1	0.3	0.0	0.0
Hypogastruridae	0.1	0.4	0.0	0.0	0.0	0.0	0.0	0.0
DECAPODA								
Cambarus	1.6	1.5	1.0	0.6	1.6	1.6	0.9	1.2
DIPTERA								
Athericidae	0.0	0.0	0.0	0.0	0.1	0.3	0.1	0.3
Blepharicera	1.4	2.6	1.4	2.8	0.0	0.0	0.0	0.0
Ceratopogonidae	1.0	1.7	1.4	1.5	1.3	1.1	2.8	2.7
Chironomidae	48.8	31.2	50.9	32.9	91.1	33.9	111.9	32.6
Dicranota	0.2	0.4	0.5	1.1	0.2	0.4	0.1	0.3
Dipteran pupae	4.2	2.7	5.4	4.5	6.9	3.7	10.6	6.1
Dixa	0.1	0.2	0.1	0.2	0.0	0.0	0.0	0.0
Empididae	9.2	6.9	6.1	6.0	2.7	3.1	6.3	4.4
Erioptera	0.1	0.2	0.1	0.3	0.1	0.3	0.2	0.4
Hexatoma	2.0	1.4	1.0	0.9	0.8	1.0	0.9	0.9
Pedicia	1.1	1.7	0.8	0.9	0.3	0.6	0.0	0.0
Pilaria	0.2	0.4	0.8	1.0	0.9	0.9	1.8	1.7
Simuliidae	0.2	0.4	0.5	1.2	0.2	0.6	0.0	0.0
Tabanidae	0.0	0.0	0.0	0.0	0.6	1.2	0.6	1.4
Tipula	0.3	0.7	0.0	0.0	0.2	0.4	0.0	0.0
EPHEMEROPTERA								
Ameletus	0.0	0.0	0.2	0.6	0.0	0.0	0.0	0.0
Attenella	0.0	0.0	0.0	0.0	0.2	0.4	0.0	0.0
Baetidae	59.0	57.8	92.4	63.1	32.7	24.1	36.4	28.4
Cinygmula	17.9	14.0	28.3	14.5	0.8	2.0	7.4	9.1
Drunella	2.4	2.0	9.7	6.9	0.2	0.6	0.3	0.5
Epeorus	5.9	5.2	6.9	10.5	0.0	0.0	0.0	0.0
Ephemera	2.0	3.3	3.8	3.3	4.0	3.0	13.5	13.6
Ephemerella	28.7	18.7	31.1	18.4	6.7	6.5	32.0	27.1
Eurylophella	0.0	0.0	0.0	0.0	0.0	0.0	1.3	2.0
Leurocuta	0.3	0.6	0.0	0.0	0.0	0.0	0.0	0.0
Paraleptophlebia	9.9	10.6	8.9	9.1	9.5	7.6	4.9	6.8
Serratella	0.1	0.4	0.0	0.0	0.0	0.0	0.0	0.0
Stenonema	2.5	2.9	4.6	3.9	1.2	1.8	0.7	1.1

Appendix C Continued

	Riffle				Pool			
	1993		1994		1993		1994	
	Mean	std	Mean	std	Mean	std	Mean	std
HEMIPTERA								
Gerridae	0.1	0.2	0.1	0.3	0.0	0.0	0.0	0.0
Saldidae	0.1	0.2	0.0	0.0	0.0	0.0	0.0	0.0
LEPIDOPTERA								
Noctuidae	0.0	0.0	0.1	0.3	0.1	0.3	0.0	0.0
MEGALOPTERA								
Nigronia	0.3	0.6	1.0	1.2	0.4	0.7	0.0	0.0
Sialis	0.0	0.0	0.1	0.2	0.1	0.3	0.9	1.7
ODONATA								
Gomphus	0.8	1.0	0.5	1.1	0.5	0.7	0.3	0.7
Oligochaeta	4.9	6.7	7.2	6.4	2.7	2.8	7.9	11.8
PLECOPTERA								
Acroneuria	2.1	2.6	4.4	4.0	0.0	0.0	0.9	1.3
Amphinemura	0.8	1.5	1.0	2.9	0.1	0.3	0.2	0.4
Chloroperlidae	5.0	3.7	17.1	11.2	0.8	1.1	7.5	8.3
Eccoptura	0.1	0.2	0.0	0.0	0.0	0.0	0.0	0.0
Isoperla	1.1	1.4	1.9	2.1	0.0	0.0	0.6	1.0
Leuctra	119.9	117.7	99.7	69.5	44.6	17.4	67.9	60.9
Pteronarcys	0.4	0.7	0.5	0.7	0.0	0.0	0.1	0.3
Remenus	0.9	0.9	0.4	0.7	0.2	0.6	0.6	1.4
Shipsa	0.4	0.9	1.6	2.0	0.8	1.0	0.5	1.0
Tallaperla	5.6	6.8	28.3	28.5	0.4	0.9	2.4	1.9
TRICHOPTERA								
Cheumatopsyche	3.7	3.9	2.8	2.4	0.4	0.7	0.3	0.6
Diplectrona	1.5	1.7	0.4	0.6	0.0	0.0	0.0	0.0
Dolophilodes	2.4	2.7	7.1	15.7	0.0	0.0	0.0	0.0
Glossosoma	3.5	3.5	3.1	3.9	0.8	1.1	0.9	1.8
Hydropsyche	1.0	1.5	0.3	0.7	0.0	0.0	0.0	0.0
Lepidostoma	0.3	0.7	0.2	0.5	1.6	4.3	0.3	0.5
Neophylax	0.6	1.0	0.3	0.5	0.1	0.3	0.2	0.6
Polycentropus	2.1	2.0	2.6	3.2	0.5	0.8	0.8	1.1
Pycnopsyche	0.0	0.0	0.0	0.0	0.2	0.6	0.2	0.4
Rhyacophila	1.4	1.9	2.1	2.5	1.1	1.0	0.3	0.5

Vita

Robert Howard Hilderbrand was born May 9, 1969 in Prince Georges County, Maryland. He graduated from Beall High School, Frostburg, Maryland in 1987 and received his B.S. degree in Wildlife and Fisheries Management at Frostburg State University, Frostburg, Maryland, in 1992. In August 1992, he became a candidate for the Master of Science degree in Fisheries and Wildlife Sciences (Fisheries Science) at Virginia Polytechnic Institute and State University, Blacksburg, Virginia.

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