

**Final Report:****Integrated Headwater Stream Riparian Management Study**

A four-year study funded by the National Council for Air and Stream Improvement (NCASI) to provide information to help guide riparian management for headwater streams

**Final Report:****Recovery of Amphibian and Invertebrate Communities in Recently Logged Coastal Range Headwater Streams**

A one-year study funded by the Washington State Department of Natural Resources to support monitoring in 2001. This project is a continuation of the "Integrated Headwater Stream Riparian Management Study" which was funded by NCASI and which supported monitoring in 1998 (pre-harvest) and 1999 (post-harvest). In the summer of 2000, Rayonier supported sampling of macroinvertebrates.

**Submitted To:**

**NCASI,  
Landscape and Wildlife Advisory Group (LWAG), and  
TFW Cumulative Effects Steering Committee (CMER)**

**April 30, 2003**

by: C.R. Jackson, D.P. Batzer, S.S. Cross, S.M. Haggerty, and C.A. Sturm

C. Rhett Jackson, Principal Investigator, Assistant Professor of Hydrology  
Daniel B. Warnell School of Forest Resources  
University of Georgia  
Athens, GA 30602-2152  
(706) 542-1772 Phone; (706) 542-8356 Fax  
[rjackson@forestry.uga.edu](mailto:rjackson@forestry.uga.edu)

**Co-Investigators:**

Dr. Darold Batzer, Department of Entomology, University of Georgia  
Dr. Bruce Wallace, Department of Entomology, University of Georgia  
Dr. Brian Chapman, Dean, Arts and Sciences, Sam Houston State Univ.

**Graduate Students and Field Crew:**

\*Christopher Sturm, MS Geography  
\*Stephanie Haggerty, MS Entomology  
\*Sarah Cross, MS Wildlife  
Shannon Smalley, BS Wildlife

Stephanie Hyder, MS Wildlife  
Lane Rivenbark, MS Water Resources  
Chris Decker, MS Water Resources  
Sheldon Owens, MS Wildlife

\*Lead students for investigations into geomorphology, macroinvertebrates, and amphibians, respectively.

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*Note: much of the material included in this report has already appeared in journal articles or has been accepted for journal publication. These journals hold copyrights on some of the material herein.*

Citations for Journal Articles and Manuscripts Developed from this Project:

- Jackson, C.R., C.A. Sturm, and J. Ward. 2001. Timber harvest impacts on small headwater channels in the Coast Ranges of Washington. *Journal of the American Water Resources Association* 37(6): 1533-1550.
- Jackson, C.R. and C.A. Sturm. 2002. Woody debris and channel morphology in first- and second-order forested channels in Washington's Coast Ranges. *Water Resources Research* 38(9), 1177, doi:10.1029/2001WR001138.
- Haggerty, S.M., D.P. Batzer, and C.R. Jackson. 2002. Macroinvertebrate assemblages of headwater streams in Washington's Coastal Range. *Hydrobiologia* 479:143-154.
- Haggerty, S.M., D.P. Batzer, and C.R. Jackson. In Review. Macroinvertebrate assemblage responses to logging in coastal headwater streams of Washington, USA. Submitted to *Canadian Journal of Fisheries and Aquatic Sciences*.
- Cross, S. and C.R. Jackson. In team review. Stream-dwelling amphibian communities of headwater streams in Washington's Coastal Range: Habitat relationships.
- Cross, S. and C.R. Jackson. In team review. Stream-dwelling amphibian communities of headwater streams in Washington's Coastal Range: Responses to Logging.

Master's Theses Developed from this Project:

- Sturm, C. A. 2000. Controls of woody debris on stream morphology in Washington state. M. S. Thesis. University of Georgia, Athens, Georgia.
- Haggerty, S. M. 2000. Macroinvertebrates in headwater streams from Washington's coastal mountains: assemblage structure and response to logging. M. S. Thesis. University of Georgia, Athens, Georgia.
- Cross, S.S. 2001. Amphibian assemblages in headwater streams of Washington's coastal mountains: habitat associations and response to logging. Master's Thesis. University of Georgia, Athens, GA.

## EXECUTIVE SUMMARY

In response to the need for information to guide management of non-fish-bearing headwater channels, the abiotic and biotic response of these streams to adjacent timber harvest was investigated by an inter-disciplinary team. Geomorphology, macroinvertebrate communities, and amphibian communities were surveyed in 15 streams within and near four logging sites in the Coast Ranges of Washington State before and after timber harvest. At each site, one stream served as a reference. Of the remaining 11 streams, four were buffered or partially buffered, and seven were clearcut to the channel. Pre-harvest surveys were conducted in summer 1998, six months to a year prior to logging. Complete post-harvest surveys were conducted in summer 1999, immediately following logging, and in summer 2001, two years following logging. Macroinvertebrate surveys were also conducted in 2000. In addition to surveys on these fifteen streams, channel surveys were conducted on larger channels to investigate how amphibian and macroinvertebrate communities changed along a gradient of stream size. Additionally, data from another 27 headwater streams was pooled with the original 15 streams and analyzed to characterize geomorphologies of such streams.

Because of the lack of scientific information on small streams, management decisions for these streams have been based on information gathered in larger fish-bearing streams. Analysis of a relatively small set ( $N = 42$ ) of non-fish-bearing Coast Range streams suggests differences in small stream and large stream behavior that should affect management decisions. In terms of creating habitat complexity, woody debris  $> 40$  cm diameter is relatively unimportant in habitat formation in small steep streams. This is because woody debris in this size is a small component of the wood stock, and it does not create habitat more efficiently than smaller wood or inorganic debris. Wood  $> 40$  cm diameter accounts for less than 9% of steps in such streams. Woody debris of all sizes accounted for 45% of steps, the remainder of the steps were formed by rocks and cobbles, organic debris (no piece larger than 10 cm diameter), and mixed wood and rock matrices. Frequencies and size distributions of in-channel woody debris were not different between managed stands and virgin (unmanaged) timber stands. Pools, as commonly defined, were very rare in small steep streams. Subsurface habitat, where so much alluvial material is stored on the valley floor that the channel flows in a tunnel through the alluvial material, comprises more than six percent of the channel length in small perennial streams. Our sampling was biased against subsurface habitat because streams with large amounts of this habitat were intentionally excluded from the study. Observations suggest that subsurface channel sections are often formed by very large woody debris ( $> 60$  cm diameter).

Channel widths in headwater streams are larger than would be expected from extrapolations of channel width vs. basin area relationships developed from larger streams. Only one of the 42 streams surveyed in this study had an active channel width less than 0.9 meters. Basin area alone is not a great predictor of small stream channel width. Rather, channel width is positively related to step frequency, functional woody debris frequency, and basin area and was negatively related to gradient and the percentage of channel drop in steps.

At the reach scale, particle size distributions were driven by stream power. For low stream power, the percentage of fines was quite variable, whereas for higher stream power, the percentage of fines was uniformly low. This has obvious implications for the relative sensitivity of channels to fine sediment input.

Logging without buffers had immediate and dramatic impacts on channel morphology. The clearcut channels were covered or buried by 0.5 to 2 meters of logging slash, and the average depth of coverage or burial was one meter. This logging debris increased the hydraulic roughness of the channels, trapping fine sediments within the channels. Particle size distributions shifted to much finer particles as a result. The average percentage of fine sediments in the clearcut streams increased from 12% to 44% immediately following logging. Median particle size in the post-harvest clearcut streams was significantly lower than in the post-harvest reference streams. While the median particle size in the clearcut streams decreased from 24 mm to 10 mm after harvest, a paired t-test yielded a p-value of 0.078, so the reduction in median particle size is not statistically conclusive due to the high variability in the pre- and post- harvest sample sets. However, this low p-value, coupled with our observations of channel conditions, strongly suggests a reduction in median diameter particle size (D50) in the clearcut streams due to trapping of fine sediments by the organic debris. Meanwhile, particle size distributions were almost unchanged in most of the buffered and all of the reference streams. Two years following harvest, percentages of fine sediments in the clearcut streams were dropping, but were still quite elevated. Fine sediments in the buffered streams have increased over time since logging, but the changes were not statistically significant.

Streamside buffers served as effective fences against the intrusion of logging debris, and channel morphologies in the buffered streams did not change in the first summer following logging. The slash burying the clearcut channels also acted as a thermal buffer, and temperatures in most of the clearcut streams did not change. Temperatures in the buffered streams increased slightly. Two years following harvest, the slash in the clearcut channels had degraded somewhat, but channel conditions had not changed appreciably. The buffers, meanwhile, experienced substantial blowdown, ranging from 33 to 64 percent of buffer trees. As a result of the increased light on the buffered channels, significant growth of herbaceous and shrubby vegetation was observed along the buffered channels two years after harvest.

Macroinvertebrate taxa richness and densities appear to be naturally low in headwater streams of Coastal Washington. Crayfish dominate (92.7%) the invertebrate biomass, with shredding insects dominating the non-crayfish component of the biomass. The importance of shredders declines from 1st- to downstream 3rd-order streams. An abundance of wood and a lack of algae and non-wood based detritus in the 1st-order streams led us to suspect that food webs are wood based, and our manipulations of wood availability supported this hypothesis. Most woody debris is a low quality macroinvertebrate food, which may explain the low richness, density, and non-crayfish

macroinvertebrate biomass in streams. An abundance of predatory salamanders and crayfish may also reduce standing stocks of many macroinvertebrates.

Macroinvertebrate assemblages were studied 1 yr prior to harvest (1998), immediately after harvest (1999), 1 yr following harvest (2000), and 2 yr following harvest (2001) in 4 study watersheds. Overall macroinvertebrate densities and collector and shredder densities exhibited treatment effects in 1999, with clear-cut streams supporting higher densities. Initial responses to harvest were likely due to increased inputs of woody debris and sediment in slash-covered streams. Accretion of organic and inorganic matter was higher in buffered and clear-cut streams after harvest than in reference streams. Subsequently in 2000 and 2001, no consistent macroinvertebrate responses to harvest were detected across watersheds. By 2001, however, watershed specific responses appeared to be emerging, with EPT (mayfly, stonefly, and caddis fly) taxa increasing in certain streams set in harvested landscapes and amphipod crustaceans increasing in others. Many of these macroinvertebrates feed on algae, and increases may result from increased primary productivity in streams as slash cover declines and streams become exposed to more sun light. In one watershed, harvested streams became dominated by sediment dwelling worms. We found no compelling evidence that any macroinvertebrate groups declined after harvest perhaps because streams are naturally oligotrophic.

We evaluated presence and/or abundance of stream-dwelling amphibians, specifically giant salamanders (*Dicamptodon* spp.), torrent salamanders (*Rhyacotriton* sp.), and Pacific tailed frogs (*Ascaphus truei*), in headwater streams of the Pacific Coast Ranges of Washington by four methods: 1) comparison of abundance and community composition across lithotopo units, 2) forward-stepwise regression of densities against reach-scale channel characteristics, 3) an evaluation of presence/absence within specific in-stream habitats, and 4) evaluation of abundance along longitudinal channel gradients. Amphibian richness and diversity varies greatly across sites, driven apparently by differences in lithology and topography. *Dicamptodon* are ubiquitous across lithology and topography units, insensitive to the specific habitat variables we considered, and present in streams of various sizes. Torrent salamanders are associated with low order, higher gradient streams with low flow and low drainage area. Torrent salamanders also preferred organic and mixed jams as well as riffles over other habitat units within streams. Surprisingly, torrent salamander densities were positively correlated with the percentage of pools in streams even though we observed little usage of pools by these animals. This anomalous result suggests that pool frequency is a good predictor of the type of stream habitat complexity favored by torrents. Tailed frog presence was strongly associated with geology and topography. Although the result was not statistically significant, tailed frogs appear to be more abundant in higher order streams, and their abundance appears to be weakly related to stream power and unit stream power.

The high variability in densities between sites (driven by lithotopo differences) and between streams within sites resulted in very low statistical power for the comparisons of amphibian densities prior to and following logging. Furthermore, tailed frogs were present in too few of the study site streams to support any sort of statistical

analysis. Therefore, the effects of logging on stream-dwelling amphibians were assessed through graphical methods only. In the summer following harvest, *Dicamptodon* decreased in five of six clear-cut streams, so much so they could not be detected in two of the streams. Meanwhile, their numbers showed no consistent trends in the reference and buffered streams. The results strongly suggest that clearcutting to the stream channels has short-term negative effects on local *Dicamptodon* populations. Two years following harvest, *Dicamptodon* numbers in the clearcut streams were still depressed relative to pre-harvest conditions and relative to numbers in the reference and buffered streams, but *Dicamptodon* numbers did rebound slightly. The ubiquitous presence of *Dicamptodon* in forest streams with previous timber harvesting history suggests they reoccupy sites after disturbance. While tailed frogs were found in only three clearcut streams, one reference stream, and one buffered streams, their response is also strongly suggestive. Immediately following harvest, tailed frogs disappeared in all three clearcut streams, while numbers showed no trends in the reference and buffered streams. Tailed frog numbers were highly variable in the reference stream and the buffered stream over the three summers of monitoring. Torrent salamanders, on the other hand, did not appear to be affected much by timber harvest. Their densities showed no clear trends in any treatment category. Numbers in one of the clearcut streams decreased substantially, but pre-harvest numbers were affected by a large torrent salamander hatch as evidenced by a large number of young-of-year animals. Torrent salamander numbers increased in the other clearcut streams at sites 12 and 13. At site 17, torrents had always been present in only low numbers, and torrents disappeared in the clearcut stream in the second year after timber harvest. The physical changes to the streams and riparian areas following treatment prevented terrestrial surveys in the post-season, therefore impacts of harvest on these species were not determined.

## 1. INTRODUCTION AND OBJECTIVES

This project is a multi-disciplinary effort to provide descriptive data on the geomorphology, macroinvertebrate communities, and amphibian communities in small, non-fish-bearing headwater streams and to evaluate the response of these streams to adjacent timber harvest with and without linear buffers. The overall objective of the project was to develop information to guide future management of non-fish-bearing headwater streams (historically classified as Type 4 and 5 streams; now Type N streams).

The broad goals of this effort were to 1) characterize physical controls of invertebrate and amphibian populations in non fish-bearing headwater streams; 2) assess how adjacent timber harvest alters these physical controls; and 3) assess how invertebrates and amphibians respond to these alterations. The project was designed to collect one summer of baseline data on a set of streams draining mature commercial timber and then to repeat the surveys following timber harvest on these streams. Surveys were conducted in 11 streams draining four separate commercial logging units and in a reference stream for each unit, for a total of 15 streams. Additionally, surveys were conducted in 27 other headwater streams (11 draining virgin timber, 16 draining managed forests) from the Olympic foothills and lowlands to provide more data on small stream morphology. A total of 42 streams scattered in the Coast Ranges of Washington State were used to provide geomorphic descriptions of small streams.

In this headwater macroinvertebrate study we 1) describe patterns of macroinvertebrate richness, abundance, and biomass in headwater streams of Washington's coastal range, 2) determine the effects of logging on macroinvertebrate assemblages of perennial 1<sup>st</sup> - order streams, and 3) assess the efficacy of riparian buffer strips at minimizing change. Because benthic macroinvertebrate communities of Washington's coastal range have not been previously described, reference conditions have not been established for these systems. Baseline information will be useful in assessing impacts of anthropogenic disturbances. Stream macroinvertebrates are sensitive to changes in sediment, organic material, temperature, and light and thus can be useful bioindicators to assess effects of timber harvest (Newbold et al. 1980, Hachmoller et al. 1991, Brown et al. 1997). We assessed harvest effects by comparing streams with buffer-strips, streams without buffer-strips, and reference streams using pre- and post-logging data.

The initial study was funded for only two years, and the post-harvest surveys were conducted only one to three months following timber harvest, so the post-harvest surveys reflected immediate effects of harvest on stream conditions. Therefore, reproductive or chronic stress effects on amphibians could not be detected from the data. Rayonier Northwest Forest Resources provided funding to conduct macroinvertebrate sampling in the 15 study streams in the summer of 2000. Washington DNR provided funding to extend the project to monitor the same set of streams two years after harvest (in the summer of 2001) to determine how conditions had evolved since harvest. The main objective of this project continuation was to describe and quantify longer-term responses to timber harvest of channel conditions and amphibian and macroinvertebrate communities in non-fish-bearing headwater streams.

## **2. CONCEPTUAL FRAMEWORK FOR ANALYZING AND MANAGING HEADWATER STREAMS**

The mechanical roles of woody debris in streams can be broadly categorized as hydraulic alteration, which will affect both flow and sediment routing, and habitat formation, which results from scour and deposition of sediment and organic material caused by hydraulic alteration. Since wood transport is a function of piece size relative to channel width and to the amount of flow in a channel, wood transport is very different between small and large streams. Small streams have little ability to move wood (except during debris flows), so relatively small woody debris can form jams. Conversely, very large wood tends to move the channel so that it flows around the wood or it buries the valley in accumulated sediment and creates subsurface stream habitat. Because of high colluvial input and low fluvial power in small streams, the role of large woody debris in creating habitat complexity and shaping channel structure in small streams should be much less than in larger streams which can flush small wood and which have relatively low frequencies of inorganic structures.

Export of gravel and larger particles from small steep streams is driven by rare catastrophic events (debris flows) and infrequent fluvial events. After debris flows, streams are usually scoured to bedrock, and the stream goes through a process of recovering a colluvial/alluvial valley floor. This process may take 60-100 years before

the stream appears "recovered", and woody debris larger than 40 cm diameter may be necessary to help store sediments in the valley (May and Gresswell, 2001). Recurrence intervals of scour events in first-order debris flow prone streams are around 600 years (Benda and Dunne, 1997) so long between-disturbance periods exist where channel and valley structure is relatively stable (May and Gresswell, 2001). During these periods of stability, export of wood and coarse sediments from small headwater streams should be minimal.

The biotic conceptualization of small streams has been well explored. The River Continuum Concept (Vannote et al. 1980) postulates that the functioning of headwater streams is strongly influenced by riparian vegetation. Dense riparian vegetation influences the trophic structure (i.e., functional feeding group composition) of small headwater streams (1<sup>st</sup>- to 3<sup>rd</sup>- order) by contributing allochthonous detritus and reducing autotrophic production by shading (Vannote et al. 1980, Hachmoller et al. 1991).

Ideas about what is "quality" habitat in small streams cannot borrow from knowledge of fish-bearing streams, but should be driven by habitat needs and preferences of amphibians and macroinvertebrates which comprise top trophic levels in headwater streams. Structural habitat "quality" in PNW non-fish bearing streams is best evaluated against the known habitat preferences of stream-dwelling amphibians such as the Pacific tailed frog (*Ascaphus truei*), Pacific and Cope's giant salamanders (*Dicamptodon tenebrosus* and *D. copei*), and torrent salamanders (*Rhyacotriton spp.*). These creatures are thought to be most prolific in streams with abundant interstitial spaces (steps and clean coarse sediment) and cool water temperatures. Step habitat contributes to low water temperatures due to hyporheic exchange that occurs in steps (Skaugset, 2001). Thus, management approaches thought to be best for the fauna of small non-fish-bearing streams should maximize steps, minimize fine sediments, and maintain cool water temperatures.

Since typical periods of small stream channel stability are long enough to grow several rotations of commercial timber, timber managers need information on basic habitat relationships in these streams to infer how management might affect habitat and to guide road, harvest, and buffer policies. Timber management activities affect the structural habitat quality in four principal ways:

- 1) routing road runoff and road-derived sediments to streams (e.g. Reid and Dunne, 1984; Megahan et al., 1983; Swift, 1984),
- 2) altering wood loading through harvest practices, (e.g. Froehlich, 1973; Jackson et al., 2001)
- 3) altering long term wood loading by changing riparian stands (e.g. Ralph et al., 1994), and
- 4) increasing the probability of landslides from hillslopes and of debris flows in channels (e.g. Swanson and Dyrness, 1975; Ziemer and Swanston, 1977; Ziemer, 1981).

Road runoff delivered to these streams increases fine sediment loads and thus the percentage of fine sediments. Harvest practices and riparian buffer policies can alter the timing, type, and amounts of woody debris recruitment to streams (e.g. Jackson et al.

2001). Reducing root strength and evapotranspiration on hillsides after harvest increases the incidence of landslides that deliver sediment and wood to channels (e.g. Swanson and Dyrness, 1975; Ziemer and Swanston, 1977; Ziemer, 1981). The data and analysis presented in this paper are intended to provide a better understanding of habitat structure in small streams. The analysis will explore the relative role of wood in creating desired habitat in non-fish bearing streams, not on the role wood plays in long-term valley aggradation after disturbance. Habitat and woody debris characteristics are compared between managed and unmanaged streams to yield inferences on management effects on small streams.

### 3. LITERATURE REVIEW

In steep, headwater catchments, streams are strongly influenced by shading and organic inputs from the riparian area (Vannote et al. 1980). Thus, headwater streams are particularly sensitive to changes in land-use practices on adjacent land (Vuori and Joensuu 1996). Commercial timber harvesting has multiple effects on streams including introduction of sediment and woody debris, which alters geomorphology, and the loss of riparian shading, which alters irradiance and temperature regime (Murphy and Hall 1981). In the Coastal Mountains of the Pacific Northwest, U.S.A., commercial forests of 2<sup>nd</sup> or 3<sup>rd</sup> - growth timber cover much of the headwater landscape (Brown and Krygier 1971), yet only a limited amount of information is available about impacts of logging on headwater streams of this region.

Many states have implemented regulations requiring buffer-strips to be left in stream riparian areas to minimize the impacts of logging. Riparian zones can provide important protections for water quality and are vital components of the landscape (Vannote et al. 1980, Gregory et al. 1991). However, logging practices typical of Washington's coastal range in the 1990's did not require buffer-strips around small, fishless 1<sup>st</sup> - and 2<sup>nd</sup> - order streams (Washington Dept. of Natural Res. Forest Practices Div., 1993).

#### 3.1 Wood/geomorphology

A large and growing body of literature exists on the role of wood in streams, but nearly all this literature is based on fish-bearing streams with channel widths > 4 m. Consequently, land managers have assumed the role of wood in small streams is equivalent to its role in larger streams. A summary of the literature on large woody debris (LWD) in larger PNW streams is provided to illustrate the relationships assumed to hold in small streams and to set a baseline for evaluating the function of woody debris in small streams. The literature review focuses on data from fish-bearing PNW headwater streams most similar to the small streams studied in this project.

Studies of forested channels between 4 and 30 m in width have shown that large woody debris (usually defined as wood > 10 cm diameter and 1 m length):

- 1) increases frequency and volume of pools;
- 2) traps organic material and slowly releases nutrients to the stream;
- 3) provides substrate and food for aquatic invertebrates;

- 4) traps sediments; and
- 5) increases hydraulic roughness and habitat complexity (*sensu* Independent Multidisciplinary Science Team, 1999; Bilby and Bisson, 1998).

Furthermore, land managers assume that larger wood provides better function than smaller wood because of positive relationships between residual pool depth and woody debris size shown for alluvial pool-riffle streams (Bilby and Ward 1989, Keller and Swanson 1979). Through fluvial and catastrophic transport, smaller streams serve as sources of wood to larger channels.

Wood is an important determinant of habitat structure. An increase in LWD can be associated with an increase in pool formation (Montgomery et al, 1995). Andrus et al (1988) discovered that nearly three-fourths of all pools present in a small Oregon watershed were associated with LWD or organic debris dams. Such pools are formed during high discharge flood events capable of scouring holes or reorienting LWD (Whittaker and Jaeggi, 1982). Both Bilby and Ward (1989) and Keller and Swanson (1979) found pool volumes were positively correlated to the size of the pool-forming LWD element. Pool frequency and gravel size distributions are a function of LWD abundance, channel slope, and channel size, and pool frequency is more sensitive to LWD abundance in moderate slope channels than in low slope channels (Beechie and Sibley 1997; Keller and Tally, 1979; Montgomery et al. 1995).

LWD plays important roles in shaping aquatic communities and routing sediment (Swanson and Lienkaemper 1978). Scarlett and Cederholm (1996) found that cutthroat trout populations in the state of Washington were greatly diminished after debris flows scoured sample streams. Wood steps that create pools are eliminated during these mass wasting events. Removal of woody debris has been related to a decrease in fish and invertebrate density and diversity (Piegay and Gurnell, 1997) because resultant channelization decreased viable habitats (Keller and Swanson, 1979). The channelization process leads to rapid removal of stored sediment and exposes bedrock (Montgomery et al, 1996; Keller and Swanson, 1979). In steep headwater channels, LWD retains colluvial material in the valley floor that would otherwise be occupied by bedrock (Montgomery et al, 1996; Heede, 1972). Large woody debris and colluvial material are episodically flushed from these channels by debris flows (Benda and Dunne, 1987). Marston (1982) indicated that sediment stored behind LWD in third- to fifth-order streams is 123% of the total annual sediment yield. The removal of LWD from headwater streams can temporarily increase basin sediment yield by an order of magnitude which affects downstream river geomorphology by increasing deposition in sink sites (Piegay and Gurnell, 1997; Smith et al, 1993).

Large woody debris creates areas of low energy on smaller streams that slow the transport of sediment and organic material (Bilby and Ward, 1989; Marston, 1982; Heede, 1972; Montgomery et al 1996). Large woody debris aligned perpendicular to the channel create steps where, in smaller streams, waterfalls form, and these sites can account for thirty to eighty percent of the overall channel drop (Keller and Swanson, 1979). The area occupied by steps is low compared to overall reach length; however, much of the stream's energy is dissipated where steps are located (Keller and Swanson,

1979; Abrahams et al, 1995). Stream energy is also reduced as LWD increases channel roughness (Smith et al, 1993), and the depletion of energy reduces bed and bank erosion potential (Froehlich, 1973).

Steps and scour pools in steep streams dissipate fluvial kinetic energy and thereby reduce the transport capacity. Abrahams et al. (1995) postulated that "step pool streams evolve toward an arrangement of steps that maximizes resistance to flow" reasoning that such an arrangement of steps would constitute a stable equilibrium morphology. Using laboratory flumes, they discovered that flow resistance was maximized when the ratio of the average step steepness ( $H/L$ ) to the average channel slope,  $s$ , lies between one and two. They surveyed 18 Adirondack mountain streams with step pool morphology and found the ratio of  $H/L/s$  to range from 1.18 to 1.85, indicating that these channels indeed featured a morphology that maximized flow resistance. Since Abraham et al.'s field surveys only included streams with step-pool morphology, it is unclear how steep step-pool streams differ from other steep streams in terms of fluvial resistance and whether sample bias influenced the field results.

Some researchers have indicated that densities of LWD and organic debris dams decrease as the order of a stream increases (Bilby and Ward, 1989; Bilby and Ward, 1991; Swanson and Lienkaemper, 1978). Larger streams have higher discharges capable of transporting LWD, whereas smaller streams may not reach flows capable of transporting this material (Piegay and Gurnell, 1997). The distribution of LWD and organic debris dams in smaller streams is often independent of stream hydraulics (Heede, 1972), and thus results in a randomly distributed pattern (Froehlich, 1973; Swanson and Lienkaemper, 1978). Some of the larger first- and second-order streams may be capable of transporting the smallest of LWD pieces; however, it is likely that these pieces will be retained in organic debris dams until a debris flow evacuates the channel and sweeps all woody debris downstream.

### 3.2 Macroinvertebrates

While headwater streams are important to the function of riverine ecosystems (Vannote et al. 1980, Grubaugh et al. 1997, Hershey & Lamberti 1998), little published information exists on macroinvertebrates in headwater streams from the Coastal Mountain range of Washington, U.S.A. Headwater streams of coastal Washington typically have steep gradients and flow through incised bedrock gullies covered by a dense riparian canopy, suggesting a tight terrestrial-aquatic linkage.

Allochthonous inputs to many headwater streams on the Pacific coast of the United States and Canada are dominated by conifer needles and woody debris (Anderson et al. 1978). Woody debris alters stream geomorphology, creates retention areas, and increases habitat heterogeneity (Anderson et al. 1978, Benke & Wallace 1990, Grubaugh et al. 1997). Some chironomid midges (Anderson 1989), pleurocerid snails (Anderson et al. 1978), and larval elmids beetles (Hershey & Lamberti 1998, Steedman & Anderson 1985), which are abundant in streams of the Pacific coast, feed directly on wood. However, wood itself provides little nutritional value. It is the wood - associated epixylic biofilms

dominated by bacteria and microinvertebrate forms that are most beneficial to stream macroinvertebrates (Hax & Golladay 1993).

### 3.3 Amphibians

Headwater streams and near-stream riparian areas in the coastal Pacific Northwest of Oregon and Washington provide fundamental habitat for several amphibian species:

- Torrent salamanders (4 geographically-separated species, *Rhyacotriton* spp.),
  - Southern torrent salamander, *Rhyacotriton variegatus*
  - Columbia torrent salamander, *Rhyacotriton kezeri*
  - Olympic torrent salamander, *Rhyacotriton olympicus*
  - Cascade torrent salamander, *Rhyacotriton cascadae*
- Pacific giant salamanders, (*Dicamptodon tenebrosus*),
- Cope's giant salamanders, (*Dicamptodon copei*),
- Pacific tailed frogs, (*Ascaphus truei*),
- Dunn's salamander, (*Plethodon dunni*),
- Van Dyke's salamander, (*Plethodon vandykei*), and
- Western red-backed salamander, (*Plethodon vehiculum*).

Dependence on streams differs among species. According to Leonard et al (1993), PNW amphibians fall into three categories: pond-breeding, stream-breeding, and land breeding, and the stream-breeding salamanders possess a stream-adapted larval stage which usually lasts from two to four years, although some individuals may reach sexual maturity without ever metamorphosing into terrestrial forms (i.e., neotenes or paedomorphs).

Stream-breeding amphibians are *Rhyacotriton*, *Dicamptodon*, and *Ascaphus*.

*Dicamptodon* and *Ascaphus* metamorphose into adult terrestrial forms which may, on occasion, move some distance away from streams. Paedomorphs or neotenes are common in both *Dicamptodon* species, and until recently, metamorphosed Cope's salamanders were thought to be rare. Lungless salamanders of the family Plethodontidae, on the other hand, live most of their lives outside of the stream, but two species, Dunn's and Van Dyke's salamanders, appear to be strongly associated with near-stream riparian areas. All of these species are commonly sympatric, and all can be found living within and around the same stream. A complete literature review of Pacific Northwest stream-dwelling and riparian amphibians is beyond the scope of this report, but a bibliography is provided in section 10.

#### *Species Descriptions*

A basic synopsis of the life histories of these amphibians provides context for the habitat discussion that follows. Most of the information for these species description was taken from Nussbaum, et. al. [1983] unless otherwise noted.

#### Torrent or seep salamanders (*Rhyacotriton* spp)

Torrent salamanders are small creatures (38-55 mm snout to vent length [SVL] as adults) that are frequently found as larvae and adults in springs, seeps, and low-order

streams. Most of the few ecological data available on torrent salamanders are for the southern torrent salamander (*Rhyacotriton variegatus*). In *R. cascadae* and *R. variegatus* (Nussbaum and Tait, 1997), oviposition is thought to be possible throughout the year but peaks in May. Available data indicate that torrent salamanders have long embryonic and larval intervals. In *R. variegatus*, the egg hatching period averages about 290 days and is followed by a post-hatching pre-feeding period of about 70 days, so the total embryonic period is about one year. Larval *Rhyacotriton* vary from 9 to 45 mm SVL. In *R. variegatus* from coastal Oregon, metamorphosis occurs at 3 to 3 ½ years. Sexual maturity is not attained until 4 ½ to 5 years from oviposition. Torrent salamanders are thought to be sedentary; living their whole lives in the same section of stream. Nussbaum and Tait [1977] found that 70% of a population moves less than 2 m in a summer and measured average annual larval movement at 2.2 m. However, the available three studies addressing movement on *Rhyacotriton* all suffer from sampling covering small spatial scales (< 40 m), so no opportunity has existed to detect more extensive movements. Recent finds of torrent salamander at greater distances (> 50 m) from small streams or seeps during wet conditions (M. Hayes, pers. comm.) may indicate that unrecognized greater movements occur. Welsh and Lind [1992] found *R. variegatus* in streams ranging in temperature from 6.5 to 14.5 °C and found maximum densities in streams with temperatures around 11 °C. These temperature measurements were not controlled for time of day or time of year.

A growing body of literature and observation indicates that torrent salamanders appear to be found mostly in non-fish bearing waters high in the drainage network (e.g. Wilkins and Peterson 2000, Stoddard 2001, Cross 2001, Olson et al. 2000). Torrent salamanders are found in highest numbers in perennial streams with small basin areas, shallow water, low temperatures, and low percentages of fine sediments. Wilkins and Peterson (2000) have observed that when torrent salamanders are found along larger fish bearing streams, the specific habitat in which they are found is usually a seep on the wall of the valley, not in the stream itself.

#### Coastal and Cope's giant salamanders (*Dicamptodon tenebrosus* and *copei*)

Giant salamanders also inhabit headwater streams, but their distribution does not start as high as torrent salamanders and it extends into larger order streams (e.g. Stoddard 2001, Cross 2001, Olson et al. 2000). Most ecological data available addresses the larger of the two species, *D. tenebrosus*. Adult and large larval and paedomorphic *D. tenebrosus* are too large to be consumed by most fish and can co-exist with fish (e.g. Parker 1991; Hawkins et al. 1983; Murphy and Hall 1981). Based on data from Oregon, *Dicamptodon tenebrosus* also has a long embryonic period (lasting about 275 days), and the eggs are usually deposited in subsurface water courses. The larval period is also lengthy (18 to 24 months), and giant salamanders reach sexual maturity at about three years (Nussbaum and Clothier 1973). During the larval period, salamanders always remain within the flow of water. Neotenic or paedomorphic adults are not uncommon among Pacific giant salamanders and can be the norm among Cope's giant salamanders. Pacific and Cope's giant salamanders are sometimes found inhabiting the same streams, but it is difficult to distinguish their larvae in the field. Ambiguous identification of larval *Dicamptodon* has

hindered ecological characterization of the two taxa. In larger streams, *Dicamptodon tenebrosus* are almost always found under cover during the daytime (Parker, 1991), frequently under cobble in pools. Parker (1991) found that *Dicamptodon tenebrosus* emerge at night to feed. In contrast, Wilkins and Peterson (2000) surveyed small streams and usually found *Dicamptodon* in the open in pools. Parker [1993] found that *Dicamptodon* leave cover at night and move toward riffle areas of the stream to feed.

#### Pacific tailed frogs (*Ascaphus truei*)

Because Pacific tailed frogs are possibly the most primitive amphibian found in North America, and with the Rocky Mountain tailed frog (*A. montanus*) is thought to represent the sister taxon of all extant frogs worldwide (Ford and Cannatella, 1993). Because they have many unusual attributes, scientists have studied *Ascaphus truei* extensively and for many years. The *Ascaphus* tadpole has a suction mouth which it uses to adhere to rocks and by which it feeds on various organic material, principally diatoms, growing on the rocks. Altig and Brodie [1972] observed *Ascaphus* preferences for substrate material, and found they preferred smooth rocks above 55 mm. They inferred that a minimum rock size was needed for a combination of enough grasping area for the mouth disc and provision of large enough interstitial spaces for daytime retreats. This laboratory conclusion agrees with field observations that have repeatedly found *Ascaphus* associated with coarse gravel and cobble habitat. *Ascaphus* tadpoles tend to cling to the undersides of rocks during the daytime and move to the top of the rocks to feed at night [Altig and Brodie, 1972; Feminella and Hawkins, 1996]. Feminella and Hawkins (1996) found that Pacific tailed frog tadpoles could recognize the presence of adult *Dicamptodon* and salmonids, but that they could not recognize the presence of shorthead sculpins, *Cottus confusus*. Jones [personal communication, December 11, 1998] has found co-existence between *Ascaphus* and salmonids but rarely between *Ascaphus* and sculpins.

*Ascaphus* breed in early fall, and tadpoles develop for two to three years before metamorphosing into adult frogs. The adult males feature a small tail-like appendage, are voiceless, and have very rudimentary hearing. Adults feed both in and out of water, but are active out of water frequently only when the relative humidity is very high.

The temperature relations of *Ascaphus* have been studied extensively. Bury [1969] reported that temperatures in streams inhabited by *Ascaphus* range from 2 °C to 15.5 °C, and Putnam [1931] found no *Ascaphus* in streams above 16 °C. In laboratory tests, Vlaming and Bury [1970] found that first-year tadpoles preferred temperatures less than 10 °C, second year tadpoles preferred temperatures between 10 and 16 °C, and all tadpoles avoided temperatures above 22 °C. Brown [1975] evaluated the development of eggs and found the temperature range for normal egg development was 7.6 to 18.0 °C. Alternatively, Metter [1966 and 1969] suggested that relative humidity was a more important habitat factor than temperature and that *Ascaphus* could survive at 20 °C if relative humidity was high.

As will be discussed below, numerous researchers have inferred from cross-landscape comparisons that *Ascaphus* are very sensitive to timber harvest and are often

locally eliminated from streams by adjacent timber harvest. Both the requirements for a relatively unimbedded coarse substrate and cold water temperatures have been cited as the reasons for sensitivity to timber harvest.

#### Van Dyke's salamanders (*Plethodon vandykei*)

Van Dyke's salamanders are rare throughout their range and thus little is known about their life histories or habitat requirements. Wilson et al. (1995) showed that Van Dyke's salamanders occupy regions with > 150 cm (59 in) average annual precipitation, and have an upper altitudinal limit coinciding with the bound between temperate mesophytic and subalpine forests. In all of the past cross-landscape studies of amphibian populations, Van Dyke's salamanders have been either absent or too few to include them in statistical analyses. Van Dyke's salamanders are riparian-dwellers associated with seepages and streamside talus, often on slopes with northern aspects (Nussbaum et al. 1983, Brodie 1970). They are usually found in mossy riparian areas within a few meters of water. These salamanders have a low fecundity. Breeding records are scarce (Noble 1925; Stebbins 1951; Jones 1989) but they are thought to breed every other year (Leonard et al. 1993), producing gelatinous clutches of seven to fourteen eggs suspended under cover. They reach sexual maturity when they reach about 44-47 mm SVL (Petranka, 1998). Females oviposit in spring, and embryos hatch in late summer/early fall (Nussbaum et al. 1983).

#### Dunn's salamanders (*Plethodon dunni*)

Dunn's salamanders are riparian-dwellers usually found in wet talus or rocks adjacent to streams. They do not inhabit water, but seek moist substrate (Corkran and Thoms, 1996). The single known nest was 38 cm deep in a rock crevice where the female tended the eggs throughout a 70-day incubation (Dumas 1955). Based on Nature Conservancy, Oregon State classifications, and Washington State classification (as listed in Corkran and Thoms 1996), Dunn's salamanders are relatively common in the Oregon Coast Range but are uncommon to rare in the Willapa Hills. No Dunn's salamanders were encountered during this study.

#### Western red-backed salamanders (*Plethodon vehiculum*)

Western red-backed salamanders live in a variety of habitats and might be described as riparian-associated. They live in cool, damp, and shaded ravines with lots of organic cover, and they are also found in talus slopes. (Brodie 1970; Corn and Bury 1991; Nussbaum et al. 1983, Leonard et al. 1993, Corkran and Thoms 1996) Although they are found in saturated streamside substrate, they are also common in upslope environments. Females are thought to breed every other year (Peacock and Nussbaum 1973 according to Petranka 1998), producing clutches of 4 to 18 eggs deposited in rock crevices, logs, and under bark (Carl 1943; Hanlin et al. 1979 according to Petranka 1998). They reach sexual maturity at two to four years, and females reproduce every two years (Brodie 1970; Ovaska and Gregory 1989).

### *Amphibian Responses to Landscape Alteration*

Both stream- and land-breeding amphibians have become a concern for timber managers in the last decade because of research suggesting that clearcutting causes local temporary extirpation of some species and reduces population numbers for other species. Much of the recent interest in amphibians is motivated by the results of a cross-landscape comparison of amphibian populations by Corn and Bury [1989]. They found that amphibian populations in young managed forests (14 to 40 years old) in Northern California and Oregon featured lower proportion of presence, lower diversity, and lower density when compared to populations in old-growth forests. Specifically, Corn and Bury [1989] found lower densities of four amphibian species, *D. tenebrosus*, *R. variegatus*, *A. truei*, and *P. dunni*, in younger forests compared to old-growth. They also found a strong relationship between amphibian populations and channel gradient and substrate type. Because their harvested and unharvested streams featured different distributions of gradients, with the harvested sites possessing a higher proportion of low gradient sites, the results may say more about gradient than about harvest effects. They inferred that clearcut logging could lead to local extirpations of amphibians and that recovery might take longer than 40 years. Results of this study have also been published in [Bury et al, 1991a; Bury et al, 1991b; and Corn and Bury, 1991].

Welsh and Lind [1996] developed a regression model of habitat correlates for the Southern Torrent salamander (*Rhyacotriton variegatus*) and found this species occurred within a narrow range of habitat conditions and is associated with cold water, coarse substrates, and dense conifer forest with large trees and abundant moss. They infer from these habitat preferences that *R. variegatus* is dependent on habitat conditions best provided by late seral forests in Northwestern California. "While it is clear that this species occurs in some early seral habitats, its combined habitat preferences demonstrate an ecological dependence on conditions typically found in late seral forests." This study corroborated findings of earlier studies by these researchers [Welsh and Lind, 1988; Welsh, 1990; Welsh and Lind, 1992]. Welsh (1990) related amphibian abundance to the age of mature stands (ranging from about 30 to several hundred years) and found that Olympic salamanders [now southern torrent salamander], Del Norte salamanders, and [Pacific] tailed frogs occurred predominantly in older forests and were uncommon or absent on young sites. Welsh postulated that these species required the moderated microclimate of old-growth forests. This study included both terrestrial and aquatic sampling, but the results were not separated, so it is not clear how aquatic amphibian populations were affected by stand age. Nine other amphibian species did not appear sensitive to stand age.

Steele (2001) studied Cascade torrent salamanders in the southern Cascades of Washington State and found lowest densities in stands < 23 years old, but he also found low numbers in stands > 60 years old. Water temperature was the best predictor of torrent salamander abundance, but stream temperatures showed no relationship with forest age. Although Steele attempted to select geomorphically and geologically similar streams, amphibian densities were highly variable.

Kelsey [1995] used cross-landscape and pre-and post-treatment comparisons to assess the sensitivity of *D. tenebrosus* and *A. truei* populations to timber management. Kelsey found that average densities and biomass were similar between managed and unmanaged streams. Because of greater coefficient of variation of *A. truei* populations in managed streams (166%) relative to unmanaged streams (124%), Kelsey concluded that *A. truei* was vulnerable to habitat changes following harvest.

Research efforts to determine the effects of timber harvest on stream-dwelling amphibians have reached a wide spectrum of conclusions. The three studies discussed below portray more subtleties in the relationship between forest management and stream amphibians. Murphy and Hall [1981] found that *D. tenebrosus* preferred large crevice habitat in cobble-dominated channels, and that the effects of timber harvest varied with the channel gradient. In channels exceeding 8% slope, harvest increased the biomass of Pacific giant [now coastal giant] and Olympic torrent [now Cascade torrent] salamanders, and in channels with lower slope, harvest decreased the biomass of these salamanders. Murphy et al [1981] sampled sediment, organic detritus, algae, insects, and vertebrates in streams varying in streamside vegetative condition and stratified by gradient. They found higher invertebrate standing stocks and vertebrate biomass in streams with clearcut riparian areas. *D. tenebrosus* was the only amphibian analyzed. Because of lack of replication, the differences in *D. tenebrosus* biomass was not statistically significant, but the results suggested that riparian clearcutting increased the biomass of *D. tenebrosus*. Murphy et al [1981] reported a significant negative correlation between *D. tenebrosus* biomass and the percentage of sand in the channels. They also found *D. tenebrosus* biomass in streams of 1% gradient was only 55% of the biomass in streams of 10% gradient.

Diller and Wallace [1996] measured presence and abundance of the southern torrent salamander, *R. variegatus*, in Northern California Coast Range streams draining managed forests (0-80 years of age) and related salamander presence and abundance to landscape and stream habitat variables. Geology (grouped into consolidated and unconsolidated deposits) was the only landscape variable that predicted presence, with *R. variegatus* favoring consolidated lithologies. The landscape variables of forest age, cover type, stream aspect, and elevation did not predict presence. Stream gradient was the only reach variable that predicted presence; habitat variables that did not predict presence included woody debris, substrate, canopy closure, vegetative overhang, temperature, and flow. Pairwise comparisons indicated that reaches with salamanders had higher slope, more small boulders, and less sand. Presence and abundance of *R. variegatus* in these streams in managed forests were comparable to streams in old-growth forests surveyed by other researchers. Overall, the results suggest that *R. variegatus* favors steeper streams in consolidated lithologies with gravel substrates.

Wilkins and Peterson [2000] believed that many of these contradictory findings about stream-dependent amphibians and timber harvest might be resolved by "considering landform characteristics as independent variables while controlling for effects of recent timber harvest." They sampled amphibians in streams that varied in

landform but all of which drained managed forests of the same successional stage in the Coast Range (Willapa Hills) of Southwestern Washington. Again, they found presence and abundance of amphibians in these managed streams similar to that found in old-growth streams by other researchers. They also found species diversity to be greatest in steep channels with northern aspects, suggesting cool microclimates and certain channel forms typically bearing coarse substrate are favored by all stream-dependent amphibians. Abundance of *Dicamptodon* was positively related to pool frequency and riparian woody debris cover and negatively related to in-channel woody debris >60 cm dia. They surmised that large woody debris caused the accumulation of finer sediments to the detriment of the crevice habitat favored by giant salamanders and their prey. Abundance of giant salamanders in basalt streams was twice as great as in marine sediment streams. The likelihood of torrent salamanders was related to increasing channel gradient and decreasing basin area, while numbers of torrent salamanders were greatest on north-facing slopes and showed a negative relationship to proportion of active channel with flowing water. These results suggest that torrent salamanders favor seepage habitat and splash zones in first-order streams. Pacific tailed frogs were found in only 13% of 40 perennial streams and were found exclusively in basalt streams at elevations above 300m. Occurrence of western redback and Dunn's salamanders were related positively to channel gradient and maximum sideslope gradient, respectively. Van Dyke's salamanders were found on only three streams, all of which were on north-facing slopes in basalt basins.

Vesely and McComb (2001) evaluated relationships between amphibian species richness and abundance and width of riparian buffer strips using streams in 17 managed stands and 12 unlogged forests. They found clear positive relationships between species richness and buffer width ( $r^2 = 0.62$ ) and between total salamander abundance and buffer width ( $r^2 = 0.40$ ). Buffer strips 20 meters wide contained approximately 80% of detectable torrent salamanders, *Dicamptodon*, and Dunn's salamanders.

#### 4. SELECTION AND DESCRIPTION OF SITES

##### 4.1 Harvest Assessment Sites

Site selection for this project was a difficult and iterative process. Originally, it was our goal to find planned harvest units with four non fish-bearing perennial headwater streams and a nearby reference stream so that the effects of a clearcut and three alternative riparian treatments could be evaluated. Furthermore, logging had to be scheduled between August 1998 and June 1999 so that pre-harvest data could be collected in the summer of 1998 and post-harvest data could be collected in the summer of 1999. Most modern clearcut units, however, are not large enough to contain four perennial headwater streams. Therefore, the criterion for the number of in-unit streams was reduced to three. At least three suitable sites were needed for replication of treatment effects.

Summary of Minimum Criteria for an Acceptable Study Site:

1. At least three apparently perennial, non-fish-bearing streams within the unit,
2. Logging of the unit to occur between August 1998 and June 1999,

3. A nearby reference stream not planned for logging within three years,
4. Ability to include special riparian treatments, and
5. Presence of amphibians detected in 20-minute time-constrained search.

Because past studies have revealed that stream-dependent amphibian populations are sensitive to large-scale variables including basin geology, channel aspect, gradient, and elevation, it was originally hoped to find study sites similar in all these features to reduce the variability in relations constructed between amphibian density and wood loading or channel substrate. Controlling for these factors was deemed impractical as explained below.

In the spring of 1998, Champion Pacific Timberlands and Rayonier Timberlands agreed to cooperate with this study and to assist in site selection. The site selection criteria were communicated to these companies, and Rhett Jackson and Darold Batzer of the UGA research team visited these companies during April 20-25, 1998 to select sites. Maps for all of the Champion harvests planned for western Washington in 1998 and 1999 were reviewed, and six sites were visited in the field. None of the Champion sites met the basic requirements for three in-unit perennial streams, ability to include special treatments, and timing of harvest. Rayonier pre-screened their planned sales before our arrival, and we field-evaluated eight potential units and found four of them to meet the minimum requirements. Three sites within the Willapa Hills were selected for the study. Two sites lay within basaltic geology and the other within marine sediments. A fourth site founded on mixed marine-glacial sediments to the south of Grays Harbor was designated as a backup site. It was apparent from this site selection process that it would be impossible to control for basin geology, channel aspect, gradient, and elevation without developing harvest plans for this purpose.

Unfortunately, events arose to make site selection more difficult. Logging of one of the proposed study sites began in June 1998, prior to pre-harvest monitoring, so we added the Grays Harbor backup unit to the study. After monitoring began, the logging company to whom another unit's trees were sold informed Rayonier that they would not be able to leave buffers on any of the streams because of the marginal profitability of the site. We decided to continue to use the site for replication of clearcut effects, but we needed another site to replicate other treatments. We found another site meeting the minimum criteria in low-relief terrain within the Humptulips basin to the southwest of the Olympic Mountains. Unfortunately, once monitoring began it was learned that the Humptulips site harbored only one stream-dwelling amphibian species, *Dicamptodon copei*.

Each site was supposed to contain four streams which were to receive the following treatments: no adjacent harvest (reference stream), standard clearcut, full riparian buffer, and a non-merchantable tree buffer. In actuality, non-merchantable buffers were not practical on most sites, so the non-merchantable treatment was applied to only one stream in only one site. Furthermore, one of the "buffer" streams went dry early in both summers, and no data was collected. Table 4.1-01 summarizes locations and characteristics of the study sites.

**TABLE 4.1-01. Description of Riparian Treatment Sites**

		Unit 13.191	Unit 12.129	Unit 17.039	Unit 21.329
General location		Willapa Hills	Willapa Hills	S shore of Grays Harbor	SW border of Olympic National Forest
Drainage		Palix River	Willapa River	Newskah Creek (Grays Harbor)	Humtulpis River
Geology		Basalt	Basalt	Mixed marine sediments and glacial deposits	Weathered Crescent Basalts
Topography	Slopes	Steep (average 30°)	Steep (average 33°)	Steep (average 33°)	Low (average 12°)
	Channels	Moderate (average 12°)	Steep (average 20°)	Moderate (average 12°)	Moderate (average 9°)
Elevation	Top	600 ft.	1720 ft.	a. 500 ft.	Approx. 600 ft.
	Bottom	400 ft.	1200 ft.	300 ft.	400 ft.
Precipitation (cm/yr)		220	220	210	340
Summer stream temperature range (°C)		9.4-14.2	8.1-15.0	9.5-15.3	9.2-18.0
Average active channel width (m)		1.1	2.85	1.27	1.28
Existing Overstory		2 <sup>nd</sup> growth hemlock	2 <sup>nd</sup> growth hemlock	2 <sup>nd</sup> growth hemlock	2 <sup>nd</sup> growth hemlock
Legal Location, Willamette Meridian		T13NR09W Sec 09	T12NR08W Sec 08	T17NR09W Sec 28	T21NR09W Sec 07
Number of study streams in unit, by treatment type					
Reference		1	1	1	1
Clearcut		2	1	2	1
Partial Buffer			1		
Full Buffer		1	-	1	1
Non-merch. buffer		-	-	-	1
Harvest Date		June 1999	May 1999	March 1999	April 1999
Initial Post-harvest Survey		September 1999	July 1999	May 1999	June 1999

All four study sites were within 60 km of the Pacific Ocean in the Coastal Mountain Ranges of Washington and were Rayonier Timberlands units. Two were within the Willapa Hills (sites 12 and 13), one was on the south shore of Grays Harbor (site 17), and one was in the Humtulpis basin southwest of the Olympic Mountains (site 21). The Chehalis River, separating the northern site 21 from the southern sites, is an important biotic divide. It separates *Rhyacotriton olympicus* to the north from *Rhyacotriton kezeri* to the south. Furthermore, *D. copei* is the only *Dicamptodon* found

north of the Chehalis. Very steep side slopes (average 89.1%) were present at site 12, steep side slopes at sites 17 and 13 (average 63.2% and 64.5% respectively), and low side slopes (average 21.67%) at site 21. All sites had 50- to 60-year-old second growth hemlock overstory prior to harvest, and all the study streams were first order streams with channel widths of < 4 m. Summer daily water temperatures at these sites ranged from 8.1° to 18.0° C, and annual precipitation was between 210 and 340 cm per year. Major overstory species were western hemlock (*Tsuga heterophylla*) and Douglas-fir (*Pseudotsuga menziesii*). Western red cedar (*Thuja plicata*), and red alder (*Alnus rubra*) also were found in the overstory. Common understory vegetation at and near the streams included sword fern (*Polystichum munitum*), salal (*Gaultheria shallon*), salmonberry (*Rubus spectabilis*), devil's club (*Oplopanax horridum*), skunk cabbage (*Lysichiton americanum*), and huckleberries (*Vaccinium* sp.).

Because of the large differences in lithology and topography between these sites, they were poor replicates for one another for the amphibian portion of this project. Monitoring revealed that differences in amphibian richness and abundance were driven by lithotopo differences in the sites as was found by Wilkins and Peterson (2000). Because of the economic constraints and the limited number of suitable sites logged in any year, it is apparent that a short-term block design is not ideal for analyzing harvest impacts to amphibians. Ideas for alternative study designs are presented in the Discussion section.

#### 4.2 Data Set Augmentation To Improve Small Stream Geomorphic Characterization

To augment the channel morphology data collected in the 15 harvest-assessment streams, additional streams were surveyed and additional small stream data was gathered to create a database large enough for statistical analysis. The geomorphic characterization of Coastal Range headwater streams uses two roughly comparable sets of data collected mostly in landscapes managed for commercial timber production. Each of the two sets contains some streams located in virgin timber. Some of these virgin timber stands have not experienced large-scale disturbance in over 250 years and some experienced a large windstorm in 1921 and are described locally as "21 Blow". The 21 Blow stands feature mixed canopies with some very large trees that survived the storm and many 80-year old trees. Streams in the 21 Blow stands would have received large inputs of woody debris in the 1921 storm. The 1921 wind storm affected large areas of the west slope of the Olympic lowlands and foothills, and "21 Blow" stands comprise a large portion of virgin timber in this area. The virgin timber streams from both data sets comprise a third data set for comparison of managed versus old-growth streams. We sampled a total of 42 streams, 31 in managed landscapes and 11 in virgin timber, in the two data sets (Figure 1). All of the streams have bankfull channel widths of < 4 m (all but one < 3 m). Channel gradients range from 5% to 32% (mean = 18%), and basin areas range from 0.011 km<sup>2</sup> to 0.458 km<sup>2</sup> (mean = 0.118 km<sup>2</sup>). All of the streams are located either in the Willapa Hills in southwestern Washington, in the western and northern foothills of the Olympic Mountains, or on the southern margins of Grays Harbor. The lead author helped plan the surveys on all 42 streams. Most channel measurements in the

two data sets are identical, but a few differed, affecting how the data were treated. Basic descriptive data on all 42 streams are presented in the Appendix.

The first data set includes 23 streams that were monitored to provide baseline data for a study evaluating the effects of timber harvest on the morphology and ecology of small streams. Fifteen of these 23 streams were located in second-growth western hemlock ca. 50-65 years old. The remaining 8 streams were located in old-growth timber. This data set is referred to as the IHSR data (for integrated headwater stream riparian study). The second data set includes nineteen streams that were surveyed by Rayonier Northwest Forest Resources and Merrill and Ring Timber Company to provide basic data on non-fish-bearing streams in their managed landscapes. Sixteen of these streams were located in second growth varying in age from young plantations to 60-year old trees, and the other three streams were located in virgin timber. This second data set is referred to as the RMR data (for Rayonier and Merrill and Ring).

Methods for determining particle size distributions varied between the IHSR and RMR data sets. In the IHSR streams, zig-zag pebble counts (N=200) (Beverger and King, 1995) were used to compare reach scale differences in surface particle size distributions, while surface particle size distributions in the RMR streams were determined from Wolman pebble counts conducted on five separate riffles (N=50 in each riffle). Valley side slopes were measured in the IHSR streams, but not in the RMR streams.

## **5. METHODS**

### **5.1 Abiotic Surveys and Analysis**

#### 5.1.1 Channel Survey Methods

A reach of each stream was surveyed to determine overall change in elevation, reach gradient, overall length, individual habitat unit drop, individual habitat unit length, residual depth for pool habitat, dominant and sub-dominant particle class in each habitat unit, bankfull width, amount of functional LWD, amount of non-functional LWD, and substrate characteristics, including fine organic debris and small wood. Functional LWD was any piece that contributed to a step or jam, caused scour, trapped or sorted sediment, or protected the bank from erosion. LWD pieces within the bankfull channel that did not serve any of the above functions were classified as non-functional. The length of the reach surveyed was approximately equal to 20 times its channel width, with a minimum survey reach length of 20 meters. Habitat units were classified as one of the following types present in our streams: riffle, step, pool, bedrock cascade, subsurface (where the channel flows in a tunnel below a vegetated ground surface), and run. Steps were subdivided into LWD steps (keyed by a piece of LWD), organic debris dams (all wood < 10 cm diameter), inorganic steps (composed of boulders and cobbles), and mixed jams (keyed by inorganic material but including LWD).

Because relatively small wood functions well to alter hydraulics and habitat in these small streams, our definition of LWD was more liberal than has been used in most LWD literature. A minimum diameter of 10 cm and a minimum length of 50 cm (as opposed to 1 or 2 meters length used in most studies) were defined for LWD. Woody debris was classified by diameter and functionality. Although the functionality of woody debris is considered to increase, or at least change, with wood size, there have been no defined terms to distinguish between size classes of wood. For the purposes of this paper, the term big wood applies to 40-80 cm diameter debris, and the term very large wood applies to wood larger than 80 cm diameter.

The accepted definition of a pool in Pacific Northwest streams surveys requires a minimum residual pool depth of 10 cm (e.g. WFPB, 1996). This definition is based upon habitat requirements for salmonids and was used in the habitat surveys for the 19 Rayonier streams analyzed in this study. In non-fish bearing headwater streams, home to amphibians and macroinvertebrates, this definition is probably too restrictive. The IHSR stream surveys used a more liberal (and also more subjective) definition of a pool. At the beginning of the survey, average active width of the channel from ordinary high water mark to ordinary high water mark was estimated, and the minimum criteria for residual pool depth was set at 5% of the estimated active channel width, or 10 cm, whichever was less.

Habitat and wood frequency was reported in terms of number of units per length of stream equivalent to the channel's width. For example, a 10-m length of a 2-m wide channel encompasses five channel widths, and if there are two pools in this segment, pool frequency is 0.4 pools/cw and 0.2 pools/m. In this example, the metric has dimensions of pools. In streams of this size, most habitat units span the entire channel, and the scale of habitat units is on the order of the stream width. Bigger channels tend to have longer habitat units, so habitat frequency expressed as number per meter necessarily decreases as channel width increases. Expressing habitat frequency in terms of a variable length unit equal to each channel's width allows direct comparison of frequencies between channels of different size. Woody debris was also quantified in the same way because the wood data was analyzed primarily in terms of its role in habitat formation. The "correct" reporting of woody debris frequency depends on the analysis. For example, LWD/m<sup>2</sup> is appropriate for assessing macroinvertebrate density relationships, and LWD/m is appropriate for conducting wood budgets.

The drainage area for each stream was determined from USGS 1:24,000 scale maps. Mean annual flow was estimated using Weather Bureau isopluvial maps and assuming a uniform annual evapotranspiration of 20 inches. A stream power index was calculated as the mean annual flow multiplied by the field-measured reach-averaged channel gradient, and a unit stream power index was calculated as stream power index divided by channel width. These indices are equivalent to stream power and unit stream power divided by the specific weight of water.

### 5.1.2 Statistical Analysis

Based on PNW amphibian and macroinvertebrate literature and basic stream ecology concepts, the following habitat and geomorphic variables were deemed of ecological interest: percentage of fines (< 2 mm), median particle diameter, percentage of bedrock exposed in channel, channel width, pool frequency, percent pool area, total step frequency, LWD step frequency, and percent of channel drop in steps. Variables used in analysis, and their definitions are summarized in Table 1.

The relative importance of different factors influencing these habitat variables was explored using forward stepwise linear regression (SigmaStat). For each variable of interest, a set of predictive channel or landscape variables were hypothesized and forward stepwise regression was used to select the most important explanatory variables. In all cases,  $F$  to enter was 4.00 and  $F$  to remove was 3.90. All accepted  $p$  values were < 0.055. If two or more variables known to be structurally auto-correlated entered the regression, the regression was repeated with each of the auto-correlated variables individually, and the best resulting regression was chosen based on the adjusted R-square value. Because of the large number of variables in the analysis and because of known collinearity between "independent" variables, no attempt was made to transform variables for better linear regression. Fitting of non-linear relationships was done on a case-by-case basis after linear model selection. Any regression with an adjusted R-square < 0.4 was rejected. In some cases, relatively strong relationships yielded poor R-square values because the relationship was not linear. Because streams were not controlled for geology, topography, time since last disturbance, or climate, geomorphic relationships in these channels should exhibit high variability, and high R-square values were not expected for regression relationships.

## **5.2 Macroinvertebrate Surveys and Analysis**

### 5.2.1 Headwater stream macroinvertebrate communities

To establish reference conditions for macroinvertebrates at our study sites, we sampled macroinvertebrate assemblages in each of 15 perennial 1<sup>st</sup>- order streams using substrate baskets. One reason reference conditions needed to be established was that concerns have developed about the impact of timber harvest on 1<sup>st</sup> - order streams of Washington's Coastal range. At the time of the study, no buffers were required for fishless streams and we had observed that after logging stream channels were routinely covered in 1 - 2 m of slash. Net-based sampling techniques (Surber or kick), which are widely used elsewhere (Resh & Carter 2000), would not be effective in slash-covered streams. Thus, we used substrate basket samplers because they could be employed in streams both before and after logging.

Substrate basket samplers (30 x 30 cm) were fashioned from chicken wire mesh (2.5 cm mesh size). Baskets were filled with 1.5 L of wood and 1.5 L of cobble collected from the study channels. However, because wood is structurally more complex than cobble, the wood contributed an average of 283 cm<sup>2</sup> of surface per basket while the cobble only 153 cm<sup>2</sup>. Material was gathered directly from the channel so natural

substrate conditions would be present (i.e. the wood and cobble supported natural microbial communities). In June 1998, 3 baskets per stream were installed into pools and were left for 6 weeks. This period is sufficiently long for invertebrate assemblages to reach equilibrium (Rosenberg & Resh 1982). Baskets were placed into pools because riffles or runs in these very small streams were too shallow (i.e. maximum depths averaged 3.1 - 6.6 cm) to cover the baskets. However, baskets were placed at pool inlets to ensure a range of flow conditions were sampled.

In August, baskets were removed from pools and immediately emptied into buckets. As flows are at their lowest during this time of year, it was feasible to place buckets in the channel and quickly insert the basket into the bucket, thereby minimizing loss of invertebrates. Samples were partially processed on-site to reduce the bulk of material that would need to be transported from the remote sites set in steep terrain. First woody debris was removed from the basket and placed into plastic bags. Then, cobble was washed and scrubbed and the slurry was passed through a 1 - mm mesh net. The remaining sample residue collected in the net was added to the bags with the woody debris. All samples were preserved with 95% ethanol.

Calibration studies (Haggerty, unpublished data) found that field sieving of samples resulted in the loss of 17% of the invertebrates, consisting exclusively of small nemourids, chironomids, and amphipods. Appropriate correction factors were used when calculating those macroinvertebrate abundances. As organisms passing through the sieve were very small and comprised only 0.22% of a sample's biomass, corrections for biomass were unnecessary.

In the laboratory, all samples were washed through a 300 -  $\mu\text{m}$  sieve prior to removing macroinvertebrates under a dissecting microscope. Macroinvertebrates were identified to genus, when possible, using keys in Merritt & Cummins (1996), Stehr (1991), and Stewart & Stark (1993). Chironomid larvae were mounted and identified to subfamily using Epler (1995). Macroinvertebrate taxa richness (using lowest taxonomic level practical), abundance, and biomass were calculated for each sample. Biomass was estimated by measuring macroinvertebrates to the nearest 0.5 - mm, and converting lengths to biomass (AFDM) using length-weight regression equations provided by Benke et al. (1999). Macroinvertebrates were placed into functional feeding groups (FFGs; collectors, predators, scrapers, and shredders) following Merritt & Cummins (1996).

The 1<sup>st</sup>- order reaches in coastal Washington tend to be much smaller than perennial stream reaches studied elsewhere. Therefore, we decided to contrast them with downstream 2<sup>nd</sup>- and 3<sup>rd</sup>- order reaches, because those reaches were more comparable in size to typical 1<sup>st</sup>- order streams elsewhere. In addition, we wanted to determine if tenets of the River Continuum Concept applied to Coastal Washington streams. Two 2<sup>nd</sup>- order reaches and one 3<sup>rd</sup>- order reach were sampled in 3 of the 4 study watersheds (Sites 12, 13, and 17). The same substrate baskets described above were used with 2 baskets being installed into each of the 2<sup>nd</sup>- and 3<sup>rd</sup>- order reaches. Sampling schedules and processing were similar to those described for the 1<sup>st</sup>- order streams.

We observed an abundance of wood and very low biomass of algae in 1<sup>st</sup>- order streams, which suggested that food webs were wood based. To assess the importance of wood to macroinvertebrate assemblages in 1<sup>st</sup>- order streams, we manipulated wood volume using substrate baskets and measured assemblage responses. As described above, we had installed 3 baskets with equal volumes of wood (1.5 L) and cobble (1.5 L) into each of the study streams. At the same time, we also installed similar baskets with wood-only (3 L) or cobble-only (3 L) into 9 of the 1<sup>st</sup> order streams (3 streams x 3 watersheds), with 2 baskets of each type in each stream. The type of basket (mixed, wood-only, cobble-only) that was placed into individual pools was randomly determined. Thus, in each of the 9 streams, we had 3 mixed, 2 wood-only, and 2 cobble-only baskets. Wood-only and cobble-only baskets were processed in the same fashion as described above for mixed baskets.

### 5.2.2 Macroinvertebrate responses to timber harvest

We sampled macroinvertebrate assemblages in headwater study streams before (1998) and after harvest (1999, 2000, 2001) using the mixed cobble and wood substrate baskets described above. In June of each year, baskets (3 in each reference stream and 4 in each buffered and clear-cut stream) were installed into pools and were left for 6 weeks. Basket placement in clear-cut streams in 1999, 2000, and 2001 was stratified so that 2 baskets were placed in pools open to sunlight and 2 in pools covered by slash. In the latter case, we dug down through the slash, placed the basket, and then replaced the slash. Samples were processed as described above in the study assessing reference conditions.

Rock tiles (unglazed marble; 12.5 x 7.5 cm) were installed adjacent to substrate baskets to measure organic and inorganic matter accretion in stream reaches before and after logging. Upon removal in August, tiles were lifted straight out of the water, placed into plastic bags, and frozen until processed. Sediment was removed, then dried at 60°C for 24-48 hours, weighed to the nearest mg, ashed in a muffle furnace, and reweighed to determine ash-free-dry-mass (AFDM) and ash mass. The AFDM was used as an index of organic matter accumulation (detritus, aufwuchs) and the ash mass was used as an index of inorganic sediment accretion.

### 5.2.3 Statistical Analyses

Although crayfish comprised virtually all of the macroinvertebrate biomass (Haggerty, unpublished data), they posed problems for statistical comparisons because their biomass resulted from only a few individuals. Thus, while crayfish biomass was included for general descriptive statistics, it was omitted for comparisons among watersheds, stream orders, or substrates. Crayfish were included for all analyses of richness and abundance.

Variation among streams within watersheds and among watersheds was analyzed for mixed substrate basket values. One-way analysis of variance (ANOVA) and post hoc Tukey-Kramer means comparison tests were used to compare 1) 1<sup>st</sup> order streams within each of the 4 watersheds and 2) variation among the 4 watersheds (all mixed substrate

baskets samples in a stream were pooled for this latter analysis). Two-way ANOVAs and Tukey-Kramer tests were used to evaluate differences in taxa richness, relative abundance, and biomass among 1<sup>st</sup>, 2<sup>nd</sup>, and 3<sup>rd</sup> order streams (watershed x stream order). Chi-squared contingency tests were used to determine if the relative densities and biomass of the functional feeding groups varied significantly among stream orders, as predicted by the River Continuum Concept. When Chi-squared tests were significant, residuals were examined to determine where deviations from expected values were most pronounced. Differences in taxa richness, functional feeding group assemblage, and biomass among substrates (wood, cobble, mixed) were assessed using two-way ANOVAs (stream x substrate) with individual baskets as units of replication. Chi-squared contingency tests and residual analyses were used to determine if the biomass of the functional feeding groups varied significantly among substrates. Relationships between physical (average summer stream temperature, mean flow, average bankfull width, stream power, and channel gradient; data from Sturm 2000) and biological (macroinvertebrate taxa richness, abundance, and biomass with and without crayfish) measures were examined using Pearson product-moment correlation analyses. Procedure-wise error was addressed in those 15 analyses by using a critical value of  $p < 0.003$  ( $0.05 / 15$ ). When data were not normally distributed for any of the above parametric tests, distributions were normalized using  $\log_{10}(x + 1)$  transformations.

When assessing impacts of logging we ignored location effects when testing for temporal and treatment responses because variation in macroinvertebrate assemblages among the 4 watersheds was negligible. Two-way ANOVAs (year x logging treatment) followed by Tukey-Kramer post hoc mean comparisons tests were used to assess differences among years and logging treatments in: 1) macroinvertebrate taxa richness, abundance, and biomass; 2) relative abundance and biomass of individual functional feeding groups (FFGs); 3) abundance of individual taxa; and 4) organic and inorganic matter accretion. We also ran a series of one-way ANOVAs to assess year-by-year treatment differences for each of these variables. When data were not normally distributed for any of the above parametric tests,  $\log_{10}(x + 1)$  transformations were used to meet ANOVA assumptions.

### **5.3 Amphibian Surveys and Analysis**

#### **5.3.1 Amphibian Habitat Correlates**

##### *Site Descriptions*

The data for this study were collected at 4 general locations differing in geology and topography, all within the coastal hills and mountains of Washington (Table 4.1-01). Overstory species were western hemlock (*Tsuga heterophylla*) and Douglas-fir (*Pseudotsuga menziesii*). Western red cedar (*Thuja plicata*), and red alder (*Alnus rubra*) were also found in the overstory. Common understory vegetation near the streams included sword fern (*Polystichum munitum*), salal (*Gaultheria shallon*), salmonberry (*Rubus spectabilis*), devil's club (*Oplopanax horridum*), skunk cabbage (*Lysichiton americanum*), and huckleberries (*Vaccinium* sp.). Steep side slopes and steep channels

on basalt were present at Willapa Hills site 12; steep side slopes and moderate channels on basalt were present at Willapa Hills site 13; moderate side slopes and moderate channels on mixed marine and glacial deposits at Gray's Harbor site 17, and low side slopes and low channel slopes on highly weathered marine basalts at Olympic foothills site 21. All streams featured 50 to 60 year old second growth hemlock overstory. Summer daily water temperatures at these sites ranged from 8.1° to 18.0° C (Table 4.1-01).

The large differences in lithology and topography and the similarity of vegetation between sites allowed a qualitative evaluation of the importance of lithotopo units on amphibian diversity and abundance. Wilkins and Peterson (2000) demonstrated the importance of lithotopo units on stream-dwelling amphibian assemblages. Our study sites provided an opportunity to validate Wilkins and Peterson's findings.

#### *Determination of Reach-scale Habitat Associations*

Detailed geomorphic channel surveys and amphibian sampling were conducted on the 15 streams used in the related study of timber harvest. This data was used to drive a forward-stepwise regression analysis to determine the characteristics of stream reach habitat that are most closely correlated to amphibian densities in headwater streams. We conducted daylight amphibian surveys, choosing study streams segments for surveying that were:

- 1) in areas of stream that allowed for at least 10 m of searching and for placement of netting at the bottom of the survey stretch (Bury and Corn 1991),
- 2) virtually free of immovable impediments to amphibian capture (i.e. very large logs, deeply undercut banks, etc), and
- 3) consistent in habitat structure with the majority of the stream length.

Streams were systematically searched for amphibians using light-touch rubble rousing techniques described by Bury and Corn (1991). We placed a net across the width of the flow at the bottom of the survey and then searched for amphibians beginning upstream and working downstream toward the net. Observed amphibians were captured by hand and using dip nets, then all cover objects (>5 cm diameter) were removed, allowing for the collection of dislodged animals.

Captured individuals were identified to species except *Dicamptodon* species which were identified to genus. When the stream stretch had been searched, we checked the net for washed amphibians then returned to the water all removed cover objects and attempted to reconstruct habitat units. The captured amphibians, held in buckets of water to prevent recapture, were then released into the stream.

We initially considered 12 physical channel and environmental variables including stream gradient, drainage area, percent fines, percent pool area, number of pools per channel width, organic jams per channel width, total steps per channel width, functional large woody debris per channel width, total large woody debris per channel width, unit stream power index, seven-day-high average temperature, and median diameter particle size (D50). We used forward stepwise regressions to determine

relationships between habitat and amphibian density among the sites where amphibians occurred. We ran forward stepwise regressions on *Dicamptodon* and total amphibian densities (number/meter) against the 12 habitat variables. Variables were selected based on our field observations, literature reviews, preliminary regression analysis, and correlation matrices. For torrent salamanders we could test only 8 habitat variables: stream gradient, drainage area, percent fines, percent pool area, organic jams per channel width, total steps per channel width, functional large woody debris per channel width, and unit stream power index. The number of variables used was determined by the requirement that variables for regressions could not exceed  $n-1$  (the number of observations minus one). For torrent salamander regressions, the site 21 streams were removed from the data set, as torrents were not found in those streams. Moreover, had torrent salamanders been found in those streams, they could not have been included in the dataset anyway, as the Olympic torrent salamander (*Rhyacotriton olympicus*) would be represented rather than the Columbia torrent salamander (*R. kezeri*), which occurred in the remaining 3 units. Extremely high captures of young-of-year torrent salamanders occurred in the 13S stream, creating a definite outlier in our data. It was therefore removed from the data sets for all regressions.

### *Specific Habitat Surveys*

To compare the relative occurrence of amphibians across specific in-stream habitats, we conducted specific habitat searches during the summers of 1998, 1999, and 2001. Based on preliminary observations, we hypothesized that we would find torrent salamanders primarily in seeps and in jams made up of small organic debris, that giants would primarily use small organic debris jams as well, and that tailed frogs would typically be collected from riffle habitats. To test these hypotheses, undisturbed sections of 11 first-order streams with channel widths < 3 meters were selected (one each near sites 17 and 21, and the rest near sites 12 and 13). Within these sections, we searched for specific-habitat types that could be described as pool, riffle, bedrock cascade, log jam, wood jam, cobble jam, mixed jam, or seep. The specific habitat types are defined in Table 5.3.2.

Upon finding a particular specific habitat type, we conducted a thorough manual search for amphibians within it. Initially, we attempted to search at least five examples of each of the specific habitat types at each site. However, some of these habitat types did not exist within the first-order portions of all the selected streams. Furthermore, evaluation of the 1998 and 1999 data revealed statistical ambiguities regarding riffles and cobble jams. Additional surveys of riffles and cobble jams were conducted in 2001 to increase the statistical power of the tests.

We used 2x2 Chi-square analysis with Fisher's Exact test to evaluate differences in specific habitat occurrence by species. Presence/absence data were used for this analysis. We grouped wood jams, log jams, and mixed jams into a single habitat called organic jams, and we compared organic jams to pools, organic jams to riffles, organic jams to cobble jams, cobble jams to pools, all jams to pools, and riffles to pools.

**Table 5.3.2 Specific Habitat Unit Definitions.**

Specific Habitat Unit	Description
Pool	Relatively still water, no obvious surface current
Riffle	Shallow rapid flow moving over mostly gravel and cobble
Bedrock Cascade	Flow moves over exposed bedrock
Log Jam	An obstruction where water percolates through multiple pieces of woody debris
Wood Jam	Similar to log jam except obstruction agent is a single piece of wood.
Cobble Jam	A mass of cobble and gravel, sometimes cemented with sand and smaller fines, obstructs the flow
Organic Debris Jam	An obstruction formed by a interlocking small organic debris all less than 10 cm diameter.
Mixed Jam	Water percolates through a mixed matrix of large woody debris, small organic debris, boulders, cobbles, or gravel
Seep	An area adjacent to the main flow where water emerges from sub-surface sources and either forms a trickle to the stream or saturates the soil without forming a visible surface flow
<i>Organic Jams</i>	<i>Union of the set of log jams, wood jams, organic debris dams, and mixed jams. This pooled data was used in the chi-square analysis of occupancy frequency.</i>

#### *Upstream-downstream Surveys*

To evaluate changes in amphibian presence and abundance with increasing stream size and order, we sampled stream systems along longitudinal gradients. We hypothesized that while torrent salamanders would be less frequent in larger streams, giants would persist due to their ability to reach larger sizes and thereby avoid depredation. To test this hypothesis, we used similar amphibian and physical stream sampling techniques as in the reach-scale habitat investigation and surveyed several 10 m segments of a stream, beginning in the headwaters and ending where the water depth made our sampling technique ineffective (when significant areas of flow exceeded about 40 cm depth). We used graphical analysis of the data to consider relationships between amphibian density and several stream size metrics, including drainage area, estimated mean annual flow, stream power, and unit stream power. We analyzed differences in amphibian densities (numbers/m<sup>2</sup>) by species among stream orders using Kruskal-Wallis Analysis of Variance on Ranks.

#### 5.3.2 Pre- and Post-Logging Assessment

##### *Pre-treatment*

To conduct daylight amphibian surveys in the study streams, we selected stream segments that were:

- 1) downstream of invertebrate sampling baskets (see Haggerty 2000).

- 2) in areas of stream that allowed for at least 10m of searching and for placement of netting at the bottom of the survey stretch (Bury and Corn 1991),
- 3) virtually free of immovable impediments to amphibian capture (i.e. very large logs, deeply undercut banks, etc), and
- 4) consistent in habitat structure with the majority of the stream length.

Following the physical channel surveys, streams were systematically searched for amphibians using light-touch rubble rousing techniques described by Bury and Corn (1991). We placed a net across the width of the flow at the bottom of the survey and then searched for amphibians beginning upstream and working downstream toward the net. Any observed amphibians were captured by hand and using dip nets, then all cover objects (>5 cm diameter) were removed, allowing for the collection of dislodged animals. The same methodology and field crew members (identical crews used in 1998 and 1999 with the addition of one member in 1999; two of the original crew members led the 2001 surveys) were used at each site and in all summers, therefore we assumed that our capture densities reflect absolute densities and that comparisons among sites and seasons are valid.

Captured individuals were identified to species except *Dicamptodon* species which were identified to genus. Variables associated with the micro-habitat of their collection site (relative position along the survey, water depth, micro-habitat type, and cover/substrate type) were recorded. When the stream stretch had been searched, we checked the net for washed amphibians then returned to the water all removed cover objects and attempted to reconstruct habitat units. The captured amphibians, held in buckets of water to prevent recapture, were then released into the stream. To minimize the impact of pre-season surveying on post-season data, we collected few voucher specimens and did not rake the streambeds or destroy the stream banks in search of animals. Upon completion of the surveys, we marked the boundaries of the surveys so that the same stretches of stream could be sampled post-treatment.

In the pre-harvest surveys, two-meter wide bands of streamside habitat on both sides of the channels were surveyed for riparian-associated amphibians. This was a "light touch" survey in which all large cover was removed, holes were searched, and both moss and litter were lifted. The duff layer was not raked nor was moss removed. Maintaining the integrity of the banks was a determinant in the search method. Slash deposition on the unbuffered streams precluded streamside habitat searches in the post-harvest surveys.

#### *Post-treatment*

Post-treatment sampling in streams that were not covered in slash followed the same protocol as in the pre-treatment. In slash buried streams, we first cleared a space for the placement of the net at the base of the amphibian survey to collect any amphibians washed downstream in the flow during slash removal. We then removed the woody debris from the channels by hand using chainsaws to cut wood into manageable sizes. We removed the large boles and branches of the bucked timber that had come to rest in the channels until we had totally exposed the flow of the previous year, then conducted the amphibian surveys in the same manner as we had pre-treatment.

Because large amounts of conifer branches were resting in the flow of many post-harvest streams, we were concerned that their removal might displace from the stream amphibians taking refuge in the needles. To see if this was the case, we returned to streams where we had preformed extensive slash removal several days after initially clearing the streams. On this second visit, we re-conducted the amphibian surveys to see if any of these aquatic amphibians had returned to the streams.

With the riparian habitats of many of the post-season streams buried in slash, we did not attempt to conduct terrestrial amphibian surveys in the stream-adjacent habitats. When terrestrial amphibians were found during the stream surveys, we identified them and noted their measurements and specific habitat characteristics as with the aquatic amphibians.

### *Data Analysis*

We analyzed the treatment results three different ways: Analysis of Variance (ANOVA) using a Randomized Complete Block Design, ANOVA using a completely randomized design, and graphical assessment. Because of high inter and intra site variability in amphibian abundances, the uneven replication of treatments, and the low number of replicates, neither the RCBD nor the ANOVA revealed a significant treatment effect. Therefore we chose to rely upon a graphical analysis of the data.

## **6. RESULTS**

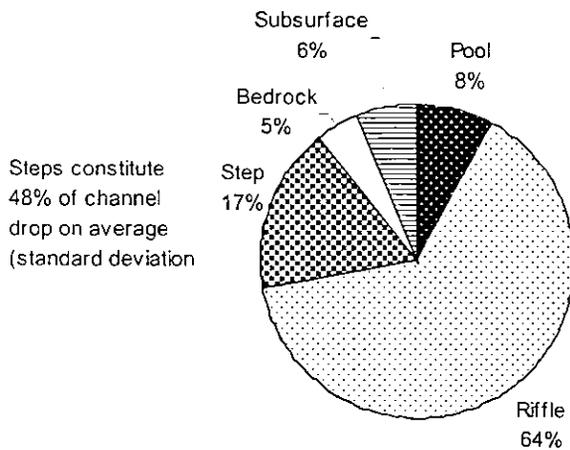
### **6.1. Channel Morphology**

#### 6.1.1. Physical Characteristics of Headwater Channels

##### *Gross Reach-scale Morphology*

Most of these streams featured what we called step-riffle morphology. While steps constituted 17% of channel length and 48% of channel drop on average, plunge pools were rare because these small streams lacked sufficient fluvial power to carve pools. Therefore, these streams do not fit into the Montgomery-Buffington classification system for mountain streams [Montgomery and Buffington, 1997] even though they fall into the stream gradient range for step-pool streams under this classification. With the 10-cm residual pool depth threshold used in the 19 RMR streams, no pools were identified in 18 of these streams. With the more liberal pool definition used for the 23 IHSR streams, pools constituted about 8% of the channel length on average. The cumulative distribution of habitat types over the 23 IHSR streams is illustrated in Figure 6.1-01.

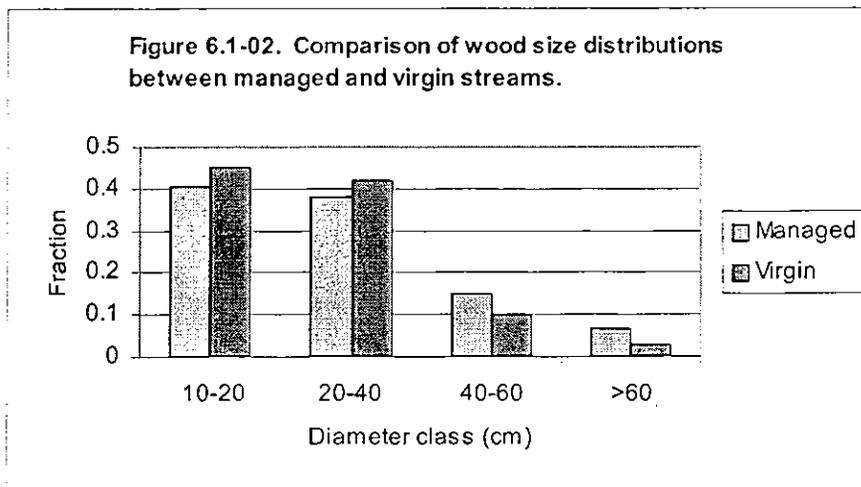
Figure 6.1-01. Cumulative Habitat Distribution for 23 IHSR Streams



The measured scarcity of pools is highly dependent on the definition of a pool. Using the standard requirement of a minimum residual pool depth of 10 cm, pools are almost non-existent in the small channels we surveyed. With a more flexible minimum residual depth of 5% of the active channel width, pools comprise about 6% of the channel length. This begs the question, what is the appropriate definition of a pool in a small stream? Since fish are not present, their habitat requirements cannot be used as a guide, and amphibian preferences for tiny pools are unknown. Many times during associated amphibian surveys, the amphibian catcher would report that an amphibian was caught in a pool, but the residual pool depth would be well below even our flexible requirement. In our judgement, a pool in the eyes of a torrent salamander is far smaller than most geomorphologists would be willing to count. The appropriate definition of a pool in these streams should probably be scaled to the size of the animals of interest. However, work needs to be done to determine species-specific pool use. Until then, the appropriate definition of a pool in a non-fish-bearing stream remains an open question.

### *Large Woody Debris*

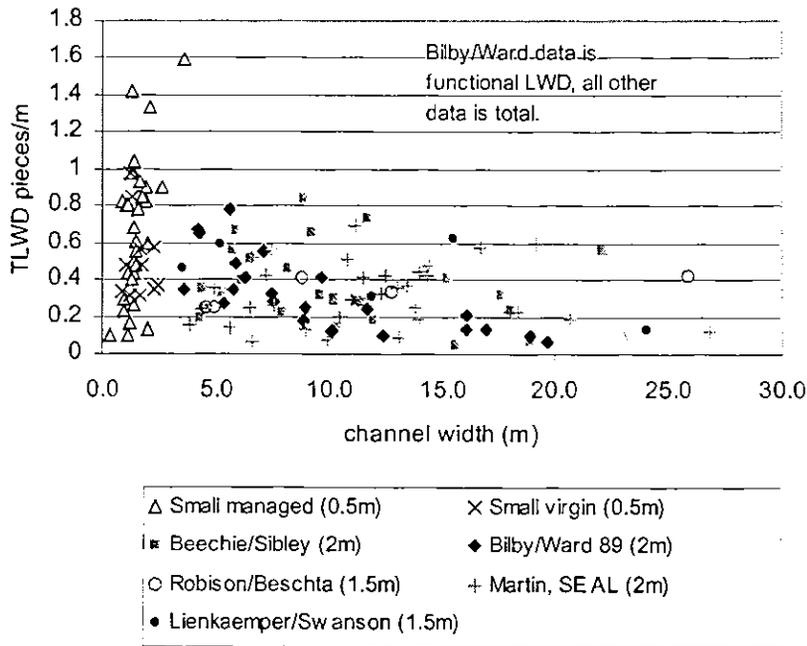
The frequency of total large woody debris (functional and non-functional) in these streams averaged 1.06 pieces/cw or 0.62 pieces/m, but was highly variable with a standard deviation of 0.95 pieces/cw or 0.36 pieces/m. Fifty-three percent of the total LWD in these channels was functional, but this number is almost meaningless as an average since the percentage of functional LWD in each stream was highly variable (median 60%; standard deviation 23%; maximum 100%; minimum 11%). Average total LWD frequency in 11 virgin timber streams was 0.80 pieces/cw or 0.51 pieces/m as compared to 1.15 pieces/cw or 0.67 pieces/m in managed streams, although this difference was not statistically significant. As shown in Figure 6.1-02, size distributions of woody debris in the virgin timber and managed streams were not different.



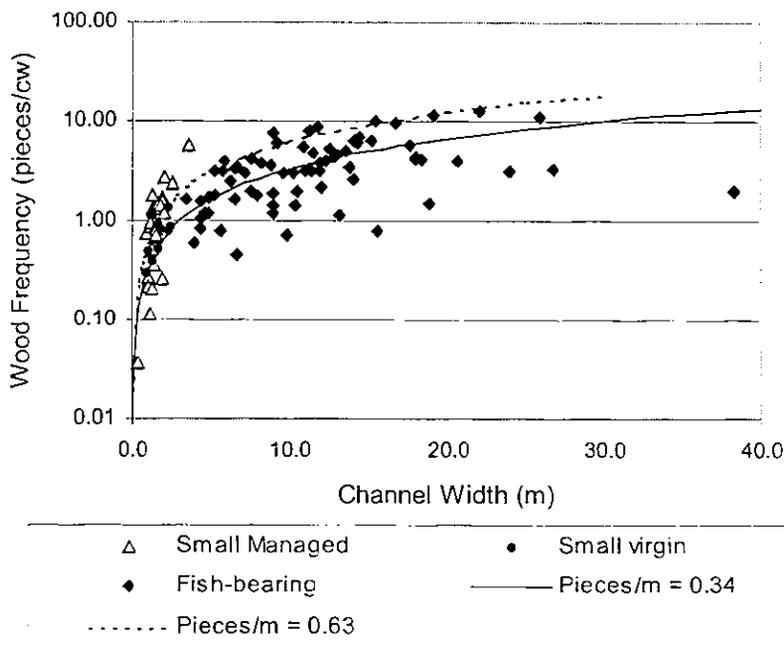
Frequencies of LWD in these small streams were generally lower than hypothesized by Keller and Swanson (1979), and also lower than would be expected by extrapolating Bilby and Ward's (1989) relationship between wood frequency and channel width. Figure 6.1-03 shows that only Bilby and Ward's data show a negative relationship with channel size. This is probably due to the fact that Bilby and Ward counted only functional wood, whereas the other data sets present total wood. Since larger channels need larger wood to store sediment, scour pools, and create habitat, the negative relationship with channel size shown by Bilby and Ward's functional wood data probably reflects fluvial sorting and flushing of smaller wood from larger channels and the fact that smaller wood is less likely to be functional in large channels. Also, the proportions of transportable and mobile wood increases with channel width (Martin and Benda, 2001). The other data sets of frequency of total wood per meter show no trends with respect to channel width.

When evaluating pieces/cw, the average total LWD frequency in our small streams (1.06 pieces/cw) was less than in the compilation of PNW fish-bearing stream total wood frequency (3.43 pieces/cw), reflecting the auto-correlation between this metric and channel width (Figure 6.1-04). Since pieces/m is invariant with channel width across the small stream data set, the pieces/cw metric necessarily increases as channel width increases. Therefore, if LWD were the dominant driver of habitat complexity in small streams, greater wood loading in pieces/m would be required to generate the same relative habitat unit frequency in small streams as in fish-bearing streams.

**Figure 6.1-03.** Relationship of woody debris frequency (pieces/meter) to channel width using data from this study and other published studies of woody debris.



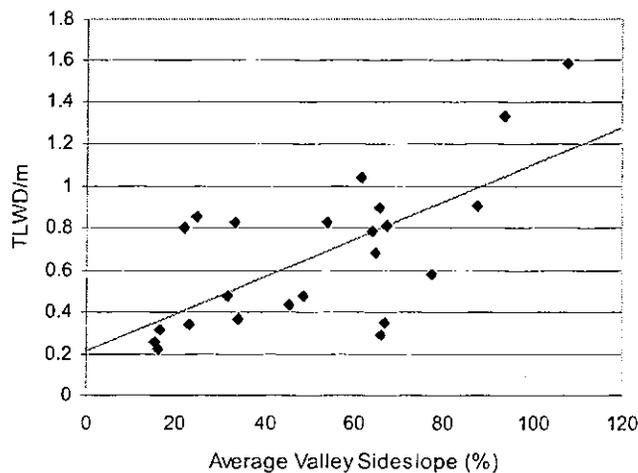
**Figure 6.1-04.** Relationship of normalized woody debris frequency (pieces/cw) to channel width from this study and other published studies of woody debris.



The high variability in LWD was not surprising given that the stands varied in sideslope and management status and history, and the channels varied in gradient and

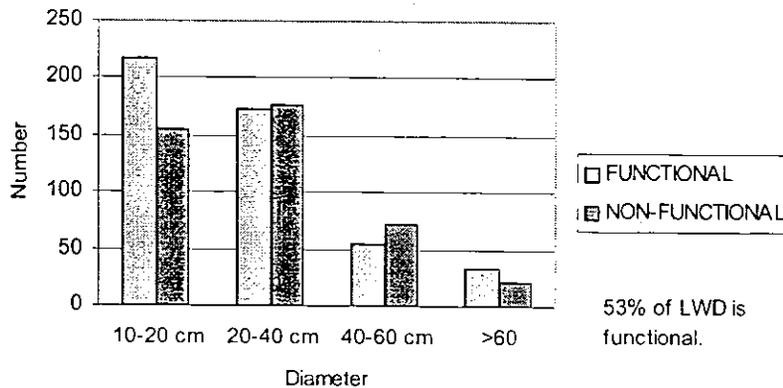
flow. Although no attempt was made to estimate time since last disturbance, less than five of these streams appear to have experienced a debris flow in the last century. Only a few of these streams were set in inner gorges, and there was evidence of recent landslides and debris flows in a small number of these streams. Bank erosion was inconsequential due to the low fluvial power of these streams. Most wood recruitment, therefore, appeared to come from limb senescence, blowdown, and chronic mortality.

Total LWD frequency in pieces/m was positively related to valley side slope in the 23 IHSR streams (Figure 6.1-05). It appears that steeper valley side slopes increase LWD recruitment distances. This observation is consistent with Fetherston et. al. (1995) and Froehlich (1973). Apparently, broken limbs and trees are more likely to fall downslope and to bounce or slide toward the stream when the sideslopes are steeper. Also, landslide contributions are likely to be greater as sideslopes increase beyond 45 percent. Eighty-one percent of functional LWD in these channels had a diameter between 10 and 40 centimeters (Figure 6.1-06). As discussed above, a principal effect of woody debris in these streams is to create steps. Since only 45 percent of steps are formed by LWD, and since only 19 percent of the functional LWD has a diameter exceeding 40 cm, it can be inferred that about 8.6 percent of steps should be created by wood larger than 40 cm in diameter. The actual percentage of steps created by wood larger than 40 cm in diameter was precisely 8.6 percent. The data on functional woody debris and step-forming agents strongly suggest that relatively small woody debris effectively functions in these streams to form steps and trap sediments. Larger wood is likely more effective in storing valley sediments (May and Greswell, 2001), but this role was not evaluated in this project. Comparing the distributions of functional and non-functional large woody debris (Figure 6.1-06) shows little relationship between the fraction of functional wood and the size class. The histograms also suggest that small wood is not preferentially flushed from these streams.



**Figure 6.1-05.** Relationship between total LWD frequency (pieces/m) and average valley sideslope (%) in the 23 IHSR streams. The regression  $y = 0.21 + 0.0089x$  was fitted with an  $R^2$  value of 0.45 ( $p < 0.001$ ).

Figure 6.1-06. Histogram of functional, non-functional, and total LWD; all 42 streams

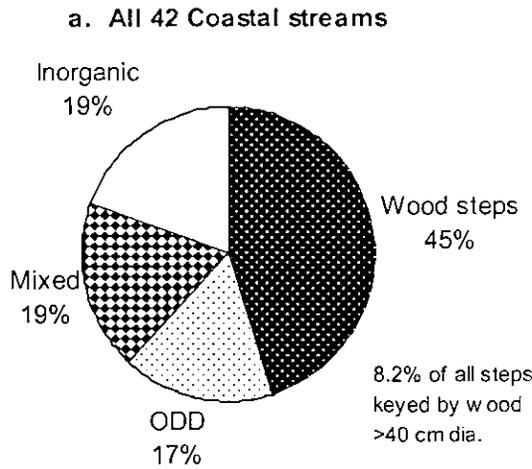


In the 23 IHSR streams, 12 subsurface habitat units existed where woody debris in excess of 40 cm diameter had stabilized so much sediment on the valley floor that the channel flowed essentially in tunnels beneath a vegetated ground surface. Of these 12 subsurface habitat units, four were keyed by wood 40-59 cm diameter, four were keyed by wood 60-79 cm diameter, and four were keyed by wood larger than 80 cm diameter. The biological value of these subsurface habitat units is unknown because we were physically unable to survey these channel units for amphibians or macroinvertebrates.

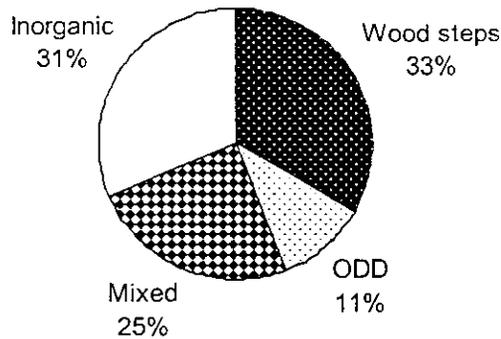
#### *Step Types and Frequencies*

Steps in these channels were formed by large woody debris (> 10 cm in diameter), organic debris dams (no key piece > 10 cm), mixed jams (cobble or boulders are the dominant step-forming agent, but wood or organic debris significantly adds to the jam), and by inorganic agents (cobbles and boulders). The distribution of step forming agents in all 42 streams is shown in Figure 6.1-07A. Fifty-five percent of the steps in these streams are formed by something other than large woody debris (LWD). Organic debris dams (all wood less than 10 cm diameter) comprised 17 percent of steps, and organic debris was an important contributor to many steps. Chesney (2000) also found that wood < 10 cm diameter was an important contributor to small stream morphology in eastern Cascade streams. The distributions of step types and wood frequencies varied little between the virgin timber and managed streams. The percentage of wood steps was actually lower and the percentage of inorganic steps was greater in the 11 virgin timber streams (Figure 6.1-07B)

Figure 6.1-07. Distribution of step types.



b. Virgin timber streams



### 6.1.2. Geomorphic Models of Headwater Streams

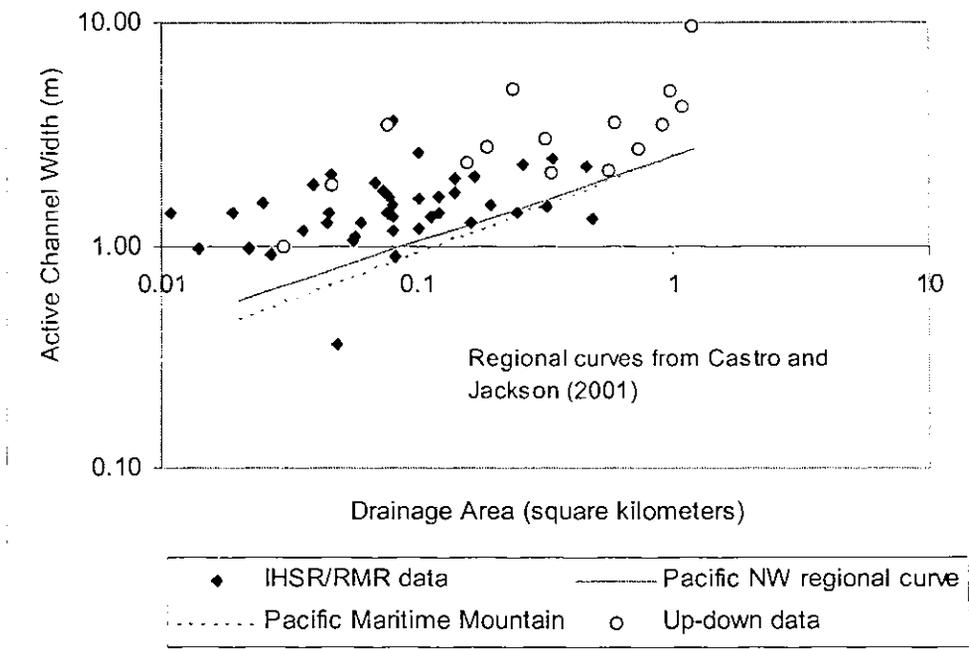
Models for active channel width, particle size characteristics, pool frequency, and step frequency were developed using forward stepwise regression. Variables used in the regression analysis are described in Table 6.1-01, and all regression models are summarized in Table 6.1-02 (at the end of this section) and described below.

#### *Channel Width*

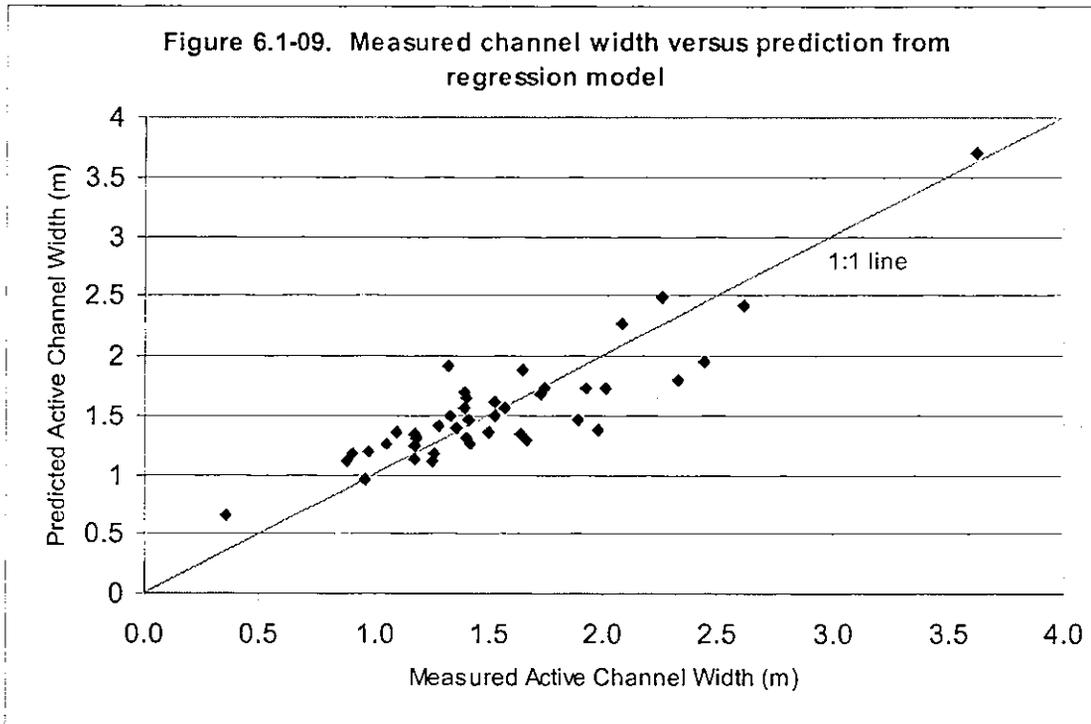
Active channel widths of headwater streams are greater than predicted by regional relationships for channel width derived from large streams. Castro and Jackson (2001) developed relationships between channel width and basin area for Pacific Northwest streams. As shown in Figure 6.1-08, channel widths for the IHSR and RMR headwater

streams are much larger than predicted by regional regression equations developed from larger stream data. Approximately 0.9 m seems to be a lower limit for headwater stream channel width.

Figure 6.1-08. Active Channel Width vs. Drainage Area



Channel width in headwater streams is determined by other factors besides drainage area. Five variables entered the forward-stepwise regression for channel width, and all five variables and their coefficients match intuitive expectations. Channel width increased in response to higher step frequency, higher functional large woody debris frequency, and larger drainage area. Basically, the channels become wider as flow increases and as the frequency of flow obstructions increases. Channel width decreased as the channel gradient and the percent drop in steps increased. Figure 6.1-09 shows the relationship between measured and predicted channel width in the IHSR and RMR using this five variable regression model.

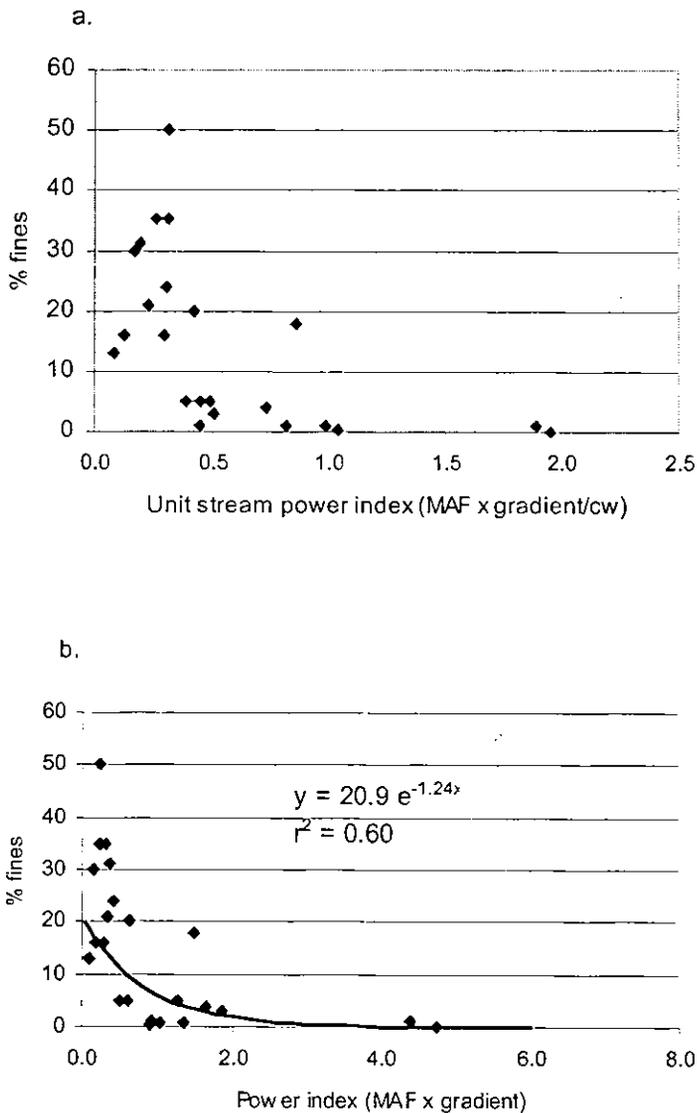


### Particle Size Statistics

Regressions for particle size statistics differed drastically between the zig-zag pebble counts and the Wolman pebble counts. Statistics from the zig-zag counts were explained by landscape-level variables while statistics from the Wolman counts were explained by the dominance of organic debris in creating steps.

The percentage of fines determined from zig-zag pebble counts ( $\%fines_{zz}$ ) was negatively related to total step frequency and the unit stream power index. The R-square value was low (0.45) but this is mostly due to the non-linear relationship of  $\%fines_{zz}$  to unit stream power as shown in Figure 6.1-10a. At low unit stream powers,  $\%fines_{zz}$  in the channel are quite variable, but at high unit stream powers,  $\%fines_{zz}$  are uniformly low. The same basic relationship holds true for the stream power index in this data set, but there is not a lot of practical difference in these two statistics in this analysis because our data set features little variability in channel width. Actually, it appears that the power index is a better predictor of  $\%fines_{zz}$  than is the unit power index, but the relationship is more non-linear (Figure 6.1-10b). Because of the obvious shape of the relationship between  $\%fines_{zz}$  and the power index, an exponential relationship was fitted as shown in Figure 6.1-10b. In some sense, the negative relationship between  $\%fines_{zz}$  and total step frequency is counter-intuitive, because steps increase the amount of sediment trapped in the channel. Apparently the surface shear stresses are higher in channels with high step frequency and thus fines sediments are flushed from these channels.

Figure 6.1-10. Relationship of percent fines (>2mm) to unit stream power and stream power.

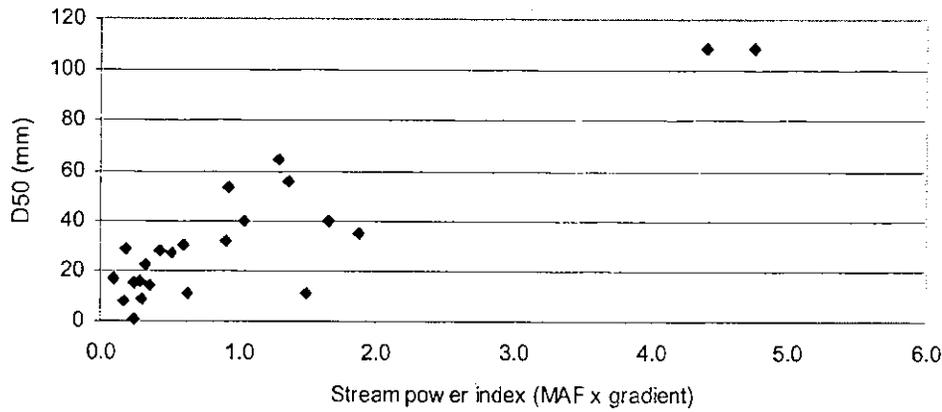


The heteroskedasticity of the relationship between percent fines and stream power may be useful for prioritizing efforts to reduce fine sediment production from forest road systems. At low stream gradients, there is a lot of variability in the percentage of fine sediment, from very high to very low. At low stream power, differences in management, creep rates, soils, etc., are likely to have a strong effect on the percentage of fine sediments. At high stream gradients, fine sediment concentrations are uniformly low, indicating that management is not likely to affect fine sediments in steep streams.

The median particle size determined from the zig-zag pebble counts ( $D50_{zz}$ ) was explained by only one variable, the stream power index, and Figure 6.1-11 indicates a relationship between these two variables although this data set lacks streams in the middle of the observed stream power range. Basically, these regressions indicate that

finer particles are flushed from streams with greater fluvial power. These streams therefore have a higher median particle size. Wood frequency did not help explain particle size metrics

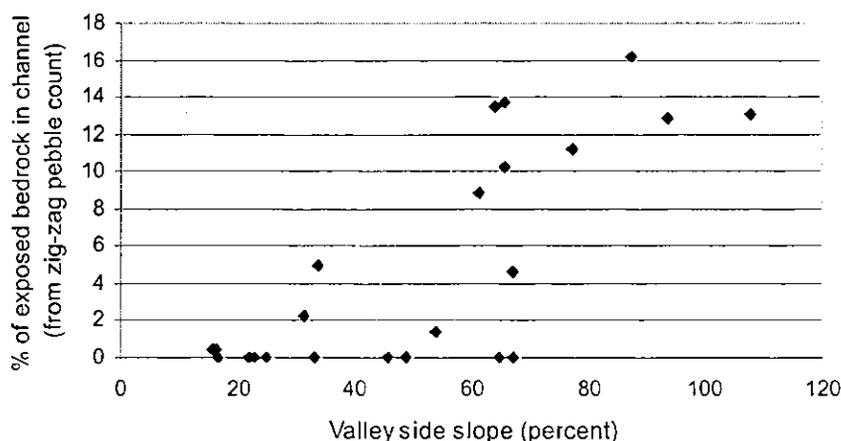
Figure 6.1-11. Relationship between median particle size (D50) and stream power.



determined at the reach scale.

The percent of bedrock determined from the zig-zag pebble count ( $\%BR_{zz}$ ) was explained only by sideslope. Figure 6.1-12 indicates a relationship between valley sideslope and the amount of exposed bedrock in the channel. It is not clear from this data set whether this is a linear relationship or whether it is a step function with greatly increased bedrock exposure when sideslopes exceed 60%. In debris flow channels, bedrock exposure is a function of time since last disturbance, and disturbance is more likely in steeper topography, so this result may reflect a partial auto-correlation between sideslope and time since last disturbance. Some streams are not going to feature bedrock, regardless of slope, because the parent material does not include bedrock or bedrock is far below the equilibrium channel. This would be the case for a first order stream that is founded in unconsolidated deposits. It would also be true of a channel founded on a debris fan (just a special case of alluvial deposits). Such streams generally do not evacuate by debris flow. None of the metrics of wood or step frequency entered the regression to explain bedrock exposure in the channel.

Figure 6.1-12. Relationship between percent bedrock exposure and valley side slope.



### *Pools*

The frequency of pools, expressed as pools/cw, was positively related to channel width, drainage area, and the percentage of drop in LWD steps. The dependence of pool frequency on channel width and drainage area shows that pool formation is more likely in streams with more flow and thus more fluvial power. Pool frequency also increases when a greater percentage of the channel drop occurs in LWD steps. It makes intuitive sense that the amount of drop in steps would influence pool formation, since a step allows the fluvial power to be concentrated at the base of the step. However, it is not clear why LWD steps would be more important than other steps. Wood steps may concentrate fluvial energy better than other steps with allow some flow to move through the step matrix itself.

The percent of the channel surface in pool habitat was positively related to TLWD/cw, the percent drop in steps, and the stream power index. Again, the dependence of percent pool habitat on power indicates that pool formation in these small channels is limited by fluvial power. In this case, pool habitat increases with the percentage of channel drop in all steps, but LWD steps are not singled out.

### *Step Frequency*

Total step frequency in these channels increases as the channels become steeper and as the channels become wider. Width and gradient were the only variables that entered the regression for total step frequency. Given the high proportion of non-LWD steps, it is not surprising that wood frequency did not enter the regression for total step frequency. Again, it appears that fluvial power dominates the morphology of small streams.

**Table 6.1-01.** Summary and definition of variables used in the statistical analysis.

<i>Variable Name</i>	<i>Definition</i>
%fines <sub>zz</sub>	% of particles less than or equal to 2mm dia. based on zig-zag pebble count
%fines <sub>w<sub>o</sub></sub>	% of particles less than or equal to 2mm dia. based on Wolman pebble count
D50 <sub>zz</sub>	median particle size (mm) based on zig-zag pebble count
D50 <sub>w<sub>o</sub></sub>	median particle size (mm) based on Wolman pebble count
%BR <sub>zz</sub>	% of channel bottom composed of exposed bedrock based on zig-zag count
Cw	bankfull channel width (m)
Pools/cw	pool frequency, pools per channel length expressed in channel widths
% pools	% of channel length composed of pool habitat units
TS/cw	total step frequency, steps per channel length expressed in channel widths
LWDS/cw	LWD step frequency, LWD steps per channel length expressed in channel widths
LWDS/TS	ratio of LWD steps to total steps
ODD/cw	organic debris dam frequency, dams per channel length expressed in channel widths
FLWD/cw	functional LWD frequency, pieces per channel length expressed in channel widths
TLWD/cw	total LWD frequency, pieces per channel length expressed in channel widths
Sideslope	average valley side slope (%)
% drop steps	ratio of total drop in steps to the total channel drop, expressed as a percentage
LWDS%drop	ratio of total drop in LWD steps to the total channel drop, expressed as a percentage
Power	stream power index, defined as (MAF x gradient). This has units of liters/s
unit power	unit stream power index, defined as (MAF x gradient / cw). This has units of liters/s/m
Runoff	estimated average rainfall minus estimated evapotranspiration (m)
DA	drainage area, km <sup>2</sup>
MAF	estimated mean annual flow (liters/second)
Gradient	channel gradient (%)

Table 6.1-02. Summary of Forward Stepwise Linear Regressions.

IHSR Data – particle size distributions based on zig-zag pebble counts; 23 streams				
	<i>Coefficient.</i>	<i>Variable</i>	<i>p-value</i>	<i>Std.error</i>
%fines <sub>zz</sub> =	+30.394	constant		4.793
	-21.733	TS/cw	0.035	9.623
	-12.579	unit power	0.019	4.918
	N = 23	R <sup>2</sup> = 0.450	Adj. R <sup>2</sup> = 0.395	Std. Error = 11.067
Variables not included in the model: LWDS/cw, LWDS/TS, ODD/cw, FLWD/cw, TLWD/cw, sideslope, cw, %drop steps, %drop LWD steps, power, runoff, DA, MAF, gradient.				
D50 <sub>zz</sub> =	<i>Coefficient.</i>	<i>Variable</i>	<i>p-value</i>	<i>Std.error</i>
	+11.779	constant		3.674
	+20.817	power	<0.001	2.294
	N = 23	R <sup>2</sup> = 0.797	Adj. R <sup>2</sup> = 0.787	Std. Error = 13.258
Variables not included in the model: TS/cw, LWDS/cw, LWDS/TS, ODD/cw, FLWD/cw, TLWD/cw, sideslope, cw, %drop steps, %drop LWD steps, unit power, runoff, DA, MAF, gradient.				
%BR <sub>zz</sub> =	<i>Coefficient.</i>	<i>Variable</i>	<i>p-value</i>	<i>Std.error</i>
	-3.858	constant		1.795
	+0.171	sideslope	<0.001	0.0311
	N = 23	R <sup>2</sup> = 0.589	Adj. R <sup>2</sup> = 0.570	Std. Error = 3.868
Variables not included in the model: TS/cw, LWDS/cw, LWDS/TS, ODD/cw, FLWD/cw, TLWD/cw, cw, %drop steps, %drop LWD steps, power, unit power, runoff, DA, MAF, gradient.				
pools/cw =	<i>Coefficient.</i>	<i>Variable</i>	<i>p-value</i>	<i>Std.error</i>
	-0.167	constant		0.0397
	+0.124	cw	<0.001	0.0281
	+0.00278	LWDS%drop	0.027	5.759
	+0.816	DA	<0.001	0.174
N = 23	R <sup>2</sup> = 0.853	Adj. R <sup>2</sup> = 0.829	Std. Error = 0.0687	
Variables not included in the model: TS/cw, LWDS/cw, LWDS/TS, ODD/cw, FLWD/cw, TLWD/cw, sideslope, %drop steps, power, unit power, runoff, MAF, gradient.				

	<i>Coefficient.</i>	<i>Variable</i>	<i>p-value</i>	<i>Std.error</i>
%pools =	-4.484	constant		2.412
	+1.920	TLWD/cw	0.025	0.792
	+0.016	%dropsteps	0.051	0.0512
	+4.765	power	<0.001	0.768
	N = 23	R <sup>2</sup> = 0.792	Adj. R <sup>2</sup> = 0.759	Std. Error = 4.206
Variables not included in the model: TS/cw, LWDS/cw, LWDS/TS, ODD/cw, FLWD/cw, sideslope, cw, %drop LWD steps, unit power, runoff, DA, MAF, gradient.				

RMR Data – particle size distributions based on modified Wolman counts in five riffles; no pools due to strict pool definition; 19 streams.

	<i>Coefficient.</i>	<i>Variable</i>	<i>p-value</i>	<i>Std.error</i>
%fines <sub>w<sub>0</sub></sub> =	+1.296	constant		2.827
	+57.228	LWDS/TS	<0.001	9.808
	-0.414	LWDS%drop	<0.001	0.102
	N = 18	R <sup>2</sup> = 0.713	Adj. R <sup>2</sup> = 0.675	Std. Error = 5.473
Variables not included in model: TS/cw, LWDS/cw, ODD/cw, FLWD/cw, TLWD/cw, cw, %drop steps, power, unit power, DA, MAF, gradient.				
	<i>Coefficient.</i>	<i>Variable</i>	<i>p-value</i>	<i>Std.error</i>
D50 <sub>w<sub>0</sub></sub> =	+21.9114	constant		3.451
	-25.555	LWDS/TS	<0.001	5.384
	-48.834	ODD/cw	<0.001	8.649
	+4.452	cw	0.061	2.173
	+0.149	LWDS%drop	0.022	0.0572
	N = 18	R <sup>2</sup> = 0.824	Adj. R <sup>2</sup> = 0.770	Std. Error = 3.001
Variables not included in model: TS/cw, LWDS/cw, FLWD/cw, TLWD/cw, %drop steps, power, unit power, DA, MAF, gradient.				

Merged Data – sideslope and particle size metrics not used in regressions; 42 streams

	<i>Coefficient.</i>	<i>Variable</i>	<i>p-value</i>	<i>Std.error</i>
<b>cw =</b>	+1.169	constant		0.155
	+1.079	TS/cw	<0.001	0.292
	+0.435	FLWD/cw	<0.001	0.0698
	-0.00440	%dropsteps	0.003	0.00138
	+1.480	DA	0.001	0.429
	-0.0139	gradient	0.033	0.000627
	N = 42	R <sup>2</sup> = 0.792	Adj. R <sup>2</sup> = 0.764	Std. Error = 0.764
Variables not included in model: LWDS/cw, LWDS/TS, ODD/cw, TLWD/cw, LWD step %drop, power, unit power, MAF.				
	<i>Coefficient.</i>	<i>Variable</i>	<i>p-value</i>	<i>Std.error</i>
<b>TS/cw =</b>	-0.063	constant		0.0827
	+0.240	cw	<0.001	0.0444
	+0.00639	gradient	0.040	0.00300
N = 42	R <sup>2</sup> = 0.507	Adj. R <sup>2</sup> = 0.481	Std. Error = 0.156	
Variables not included in model: FLWD/cw, TLWD/cw, power, unit power, DA, MAF.				

### 6.1.3. Channel Responses to Timber Harvest

#### *Habitat Distributions*

As reported in Jackson et al. (2001), the dominant effect of clearcutting adjacent to these small headwater streams was the introduction to the channel of large amounts of logging debris. Trees were felled downslope, and after bucking, tops and limbs covered the streams. This organic debris consisted of whole branches including needles, small twigs, and logs. Streams were covered or buried with half a meter to two meters of organic debris (average 0.94 meters).

Table 6.1-04 summarizes 1999 and 2001 data on channel burial and coverage for the clearcut streams. A buried channel has so much organic detritus in the flow cross-section that the channel is no longer definable. Instead, the water flows through a porous medium composed of organic debris and soil which completely fills the former channel cross-section. A covered channel is obscured by organic debris, but a recognizable channel still exists below the debris. Two years following harvest, the amount of buried channel in the clearcut streams had actually increased somewhat, due to remobilization of hillslope organic debris at site 17, but the average depth of burial had decreased slightly. The amount of covered channel had decreased greatly, as had the depth of coverage. The total amount of buried and covered channel in the clearcut streams decreased from 94% to 79% from 1999 to 2001. Furthermore, much of the covered channel length was now more recognizable and the habitat structure could be ascertained.

Habitat distributions for the buffered and reference streams are shown in Figures 6.1-13 and 6.1-14. Habitat distributions in the buffered and reference streams showed some variations but no large changes and no trends between 1998, 1999, and 2001. Even with large amounts of blowdown, thin buffers protected channel habitat structure in the near term. As the blown over trees break up and add big pieces of tree trunks to the streams, habitat is likely to change, but the time frame for this expected channel evolution is unknown.

#### *Sediment Particle Size Distributions*

Immediately after harvest, particle size distributions in the clearcut stream moved sharply toward finer particles due to the hydraulic roughness imposed by the organic logging debris (see Jackson et al. 2001). The average percentage of fines in the clearcut streams increased from 12 to 44 percent. In four of the six clearcut streams, particle size distributions moved dramatically toward finer particles. Compared to the buffered and reference streams, the clearcut channels featured significantly higher percentage of fines as determined by ANOVA analysis ( $p < 0.05$ ). The average percentage of fines in the clearcut streams increased from 12 to 44 percent. The median particle diameter ( $D_{50}$ ) therefore decreased in the clearcut streams as well, although high pre-harvest variability in  $D_{50}$  rendered the results less clear statistically. Pre-harvest  $D_{50}$  did not differ between the treatment groups, but post harvest  $D_{50}$  in the clearcut streams was less than pre-harvest  $D$  in the reference streams. Using a t-test, however, post harvest  $D_{50}$  in the

clearcut streams was not statistically different from pre-harvest  $D_{50}$  in the clearcut streams, due to high variability in  $D_{50}$ .

Sediment samples were taken in the summer of 2001 to determine if and how particle size distributions (PSDs) in the channels have changed in the last two years. Particle size distributions for each stream and each year are shown in Figures 6.1-15, 6.1-16, and 6.1-17. Results for the clearcut streams were mixed. PSDs were still unchanged in stream 12W, the only stream that showed no change in the year following harvest. Streams 17W still featured the extremely high fine sediments exhibited in the year following harvest. PSDs in streams 17M and 13M had improved since 1999 but had not returned to pre-harvest conditions. The PSD in stream 21E had become finer by 2001. Samples were not taken in stream 13S because the pins marking the sediment sampling locations had been lost.

PSDs in three of the five buffered streams basically did not change over time. One stream had become much finer in the year after harvest, and was still finer in 2001. Another stream showed a decrease in  $D_{50}$  but this could simply be due to a single cobble not being sampled when cobble material had been sampled in previous summers. Meanwhile, PSDs in the reference stream showed some variability but no trends and no ecologically significant variation in % fines.

The average percentage of fines in the clearcut streams increased dramatically in the summer following harvest, due to the increased channel roughness from the organic debris (Figure 6.1-18). By 2001, the percentage of fines in the clearcut streams had dropped somewhat, but was still highly elevated over pre-harvest conditions. In the buffered streams, percentages of fines changed little over time, but some decrease in median particle diameter existed (Figure 6.1-19).

**Table 6.1-04. Burial and coverage of streams by slash immediately and two years following harvest**

**Clearcut Streams - immediately following harvest**

Stream	% Buried	Burial Depth (m)	% Covered	Cover Depth (m)	Sum of Buried and Covered	Survey Length (m)	Length Buried (m)	Length Covered (m)
12E*	44%	1.8	28%	2.3	72%	27.7	12.3	7.7
12W	6%	1.0	91%	0.9	97%	27.0	1.5	24.5
17W	64%	1.0	36%	1.0	100%	21.0	13.4	7.6
17M	42%	1.6	58%	1.4	100%	23.0	9.6	13.4
21E	7%	0.5	85%	0.6	92%	27.6	2.0	23.4
13S	90%	0.6	8%	0.2	98%	50.0	44.9	4.2
13M	52%	1.0	48%	0.4	100%	35.0	18.3	16.7
						211.3	102.0	97.5

**Weighted Averages for Clearcut Streams**

% Buried	Burial Depth (m)	% Covered	Cover Depth (m)	% Buried or Covered
48%	0.97	46%	0.90	94%

\*Note: lower portion of surveyed section of 12E was within the buffer of fish-bearing stream below.

**Clearcut Streams - two years following harvest**

Stream	% Buried	Burial Depth (m)	% Covered	Cover Depth (m)	Sum of Buried and Covered	Survey Length (m)	Length Buried (m)	Length Covered (m)
12E*	NA	-	NA	-	NA	25.0	13.6	NA
12W	0%	0.0	64%	0.7	64%	30.0	0.0	19.3
17W	64%	1.4	36%	1.3	100%	34.0	23.2	10.8
17M	84%	1.1	16%	0.5	100%	25.0	21.0	4.0
21E	18%	0.4	29%	0.3	47%	36.0	6.6	10.3
13S	87%	0.8	13%	1.1	100%	35.0	30.3	4.7
13M	78%	1.0	11%	0.6	89%	26.0	20.4	2.9
						211.0	115.1	52.0

**Weighted Averages for Clearcut Streams**

% Buried	Burial Depth (m)	% Covered	Cover Depth (m)	% Buried or Covered
55%	0.91	25%	0.74	79%

\*Note: the previously covered portion of 12E was cleared for amphibian survey in 1999

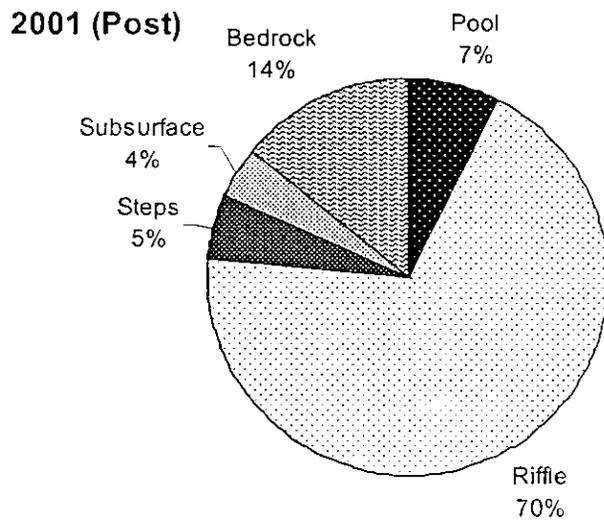
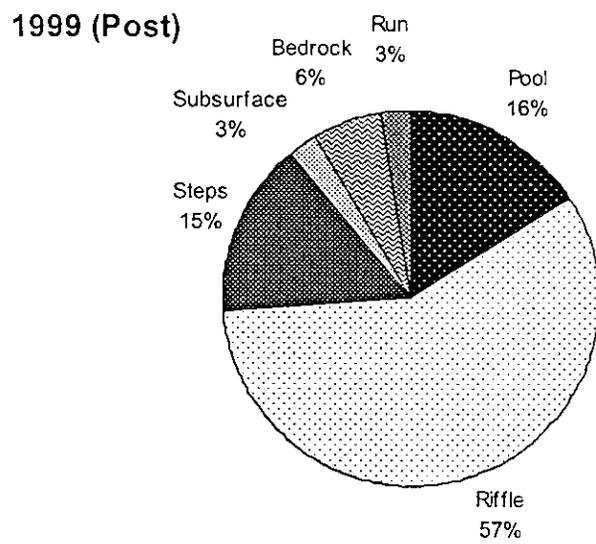
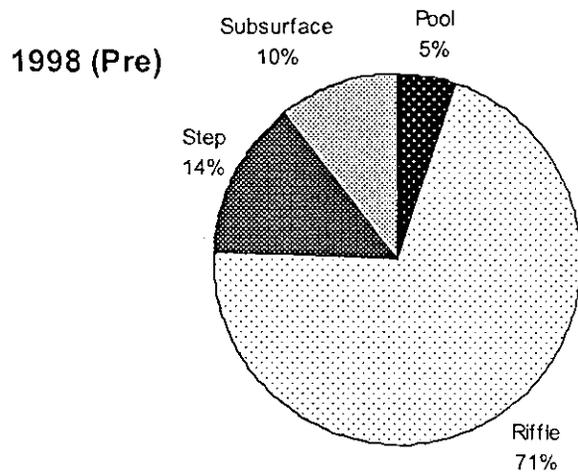


Figure 6.1-13. Cumulative Habitat Distributions in Buffered Streams, 1998, 1999, and 2001.

1998 (Pre)

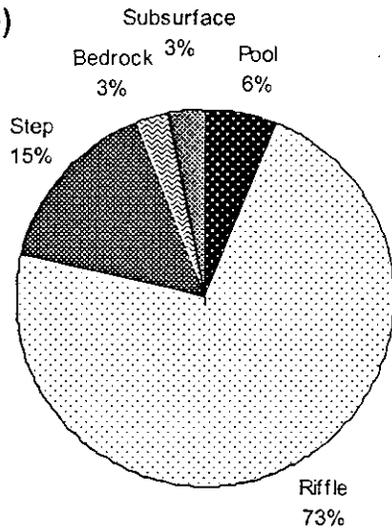
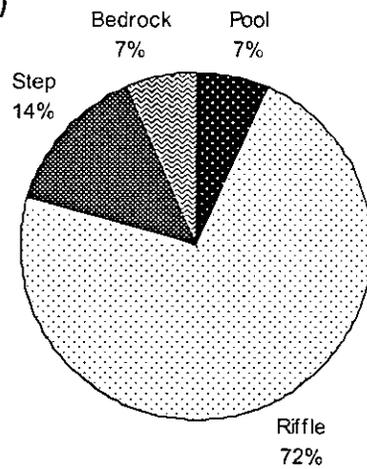
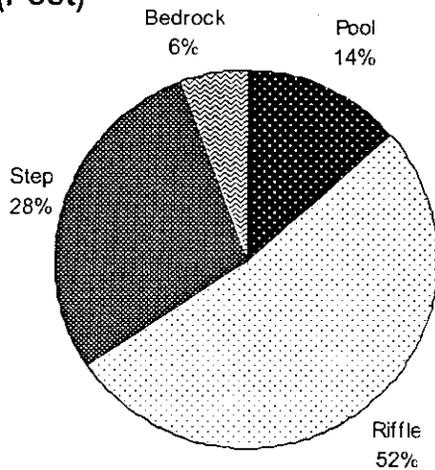


Figure 6.1-14. Cumulative Habitat Distributions in Reference Streams, 1998, 1999, and 2001. The reference stream on site 17 (Stream 17R) was logged in 2000, so the 2001 data reflects a shift simply due to the loss of data from 17R.

1999 (Post)



2001 (Post)



2001 (Post): Adjusted for loss of 17R

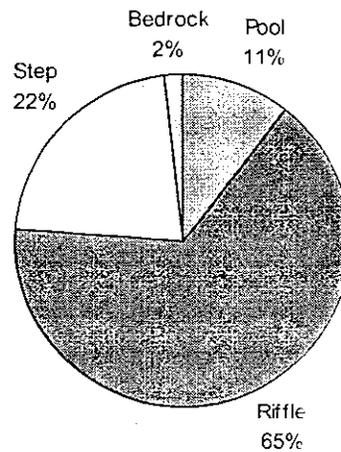


Figure 6.1-15. Particle size distributions in the reference streams; 1998, 1999, and 2000.

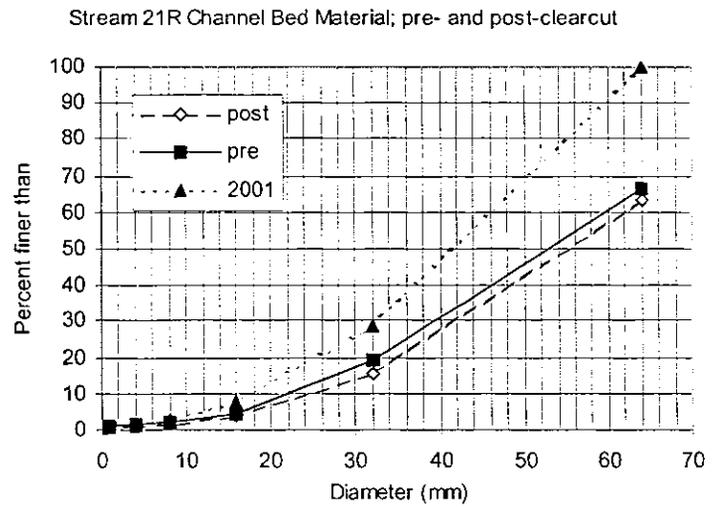
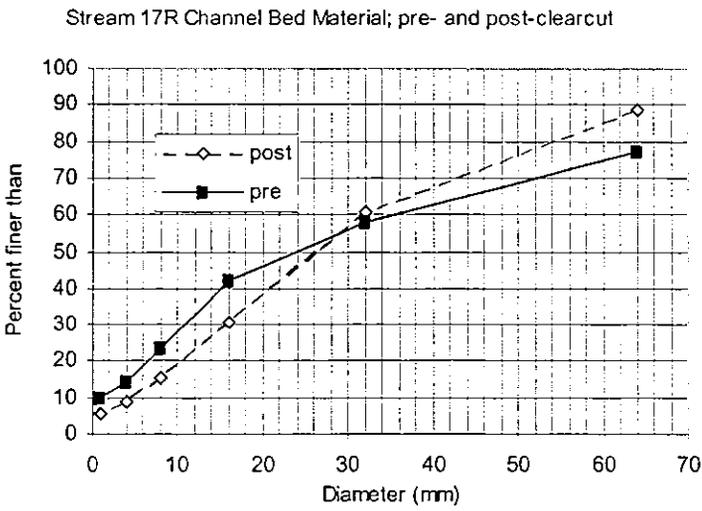
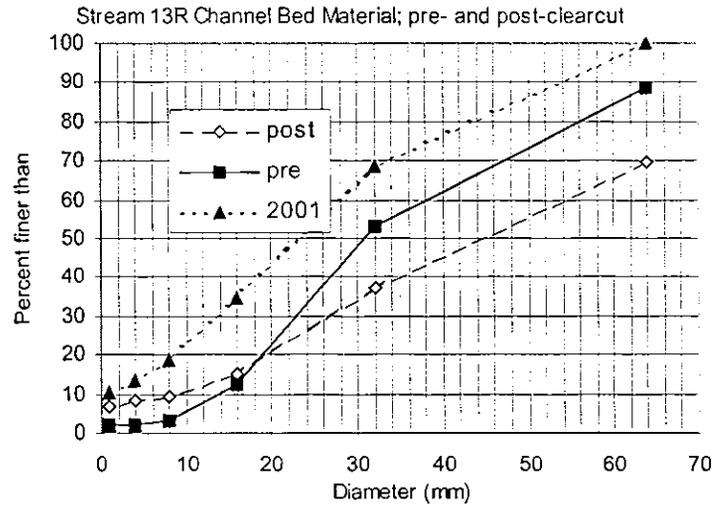
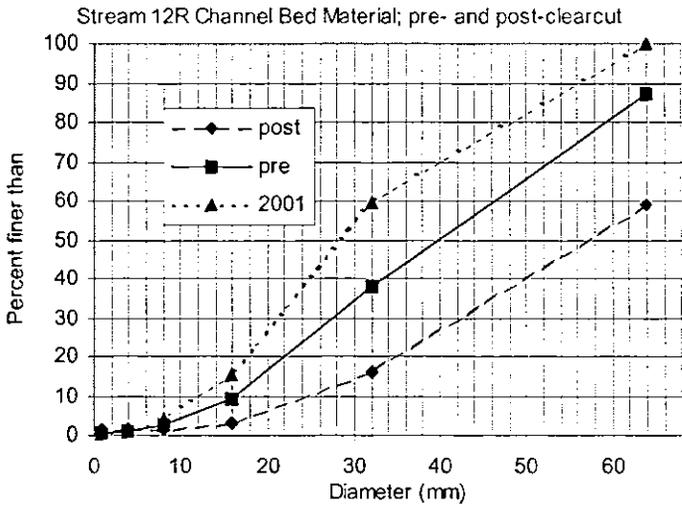


Figure 6.1-16. Particle size distributions in the buffered streams; 1998, 1999, and 2000.

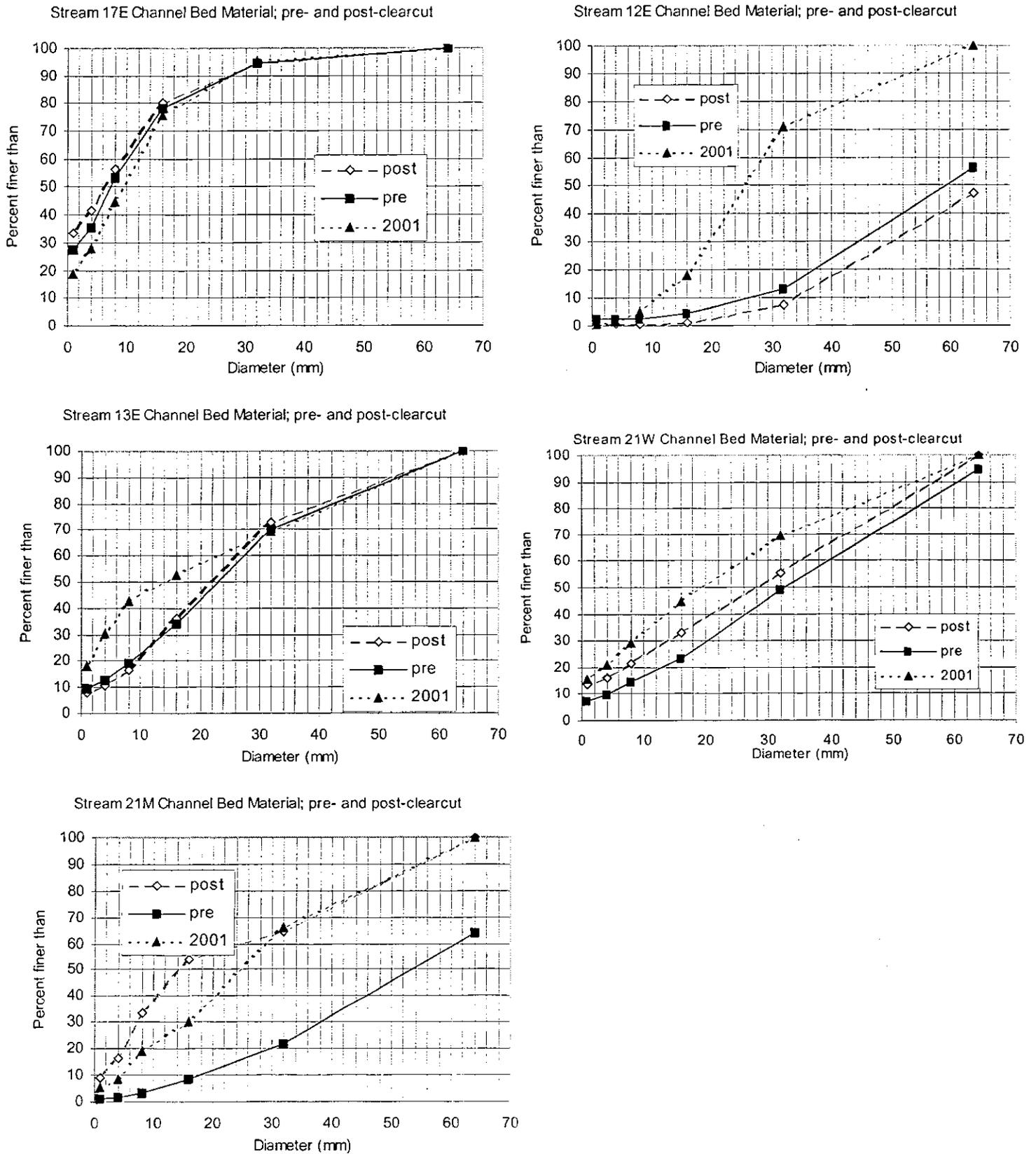


Figure 6.1-17a. Particle size distributions in the clearcut streams; 1998, 1999, and 2000.

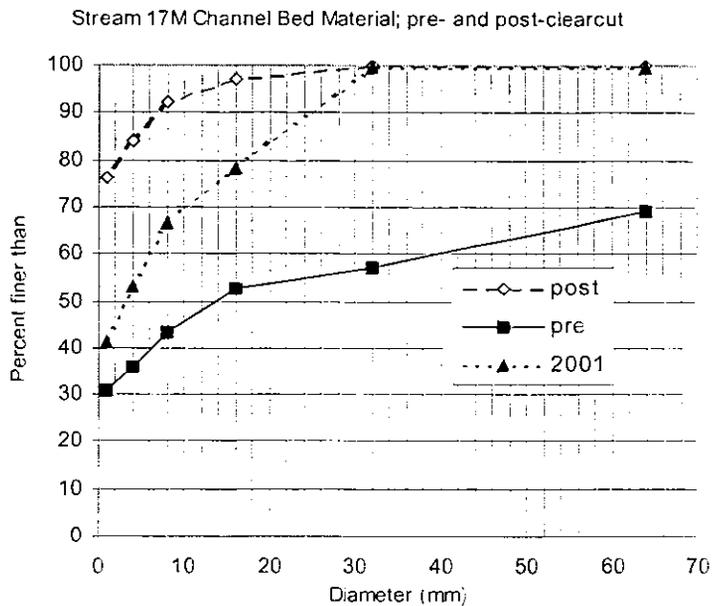
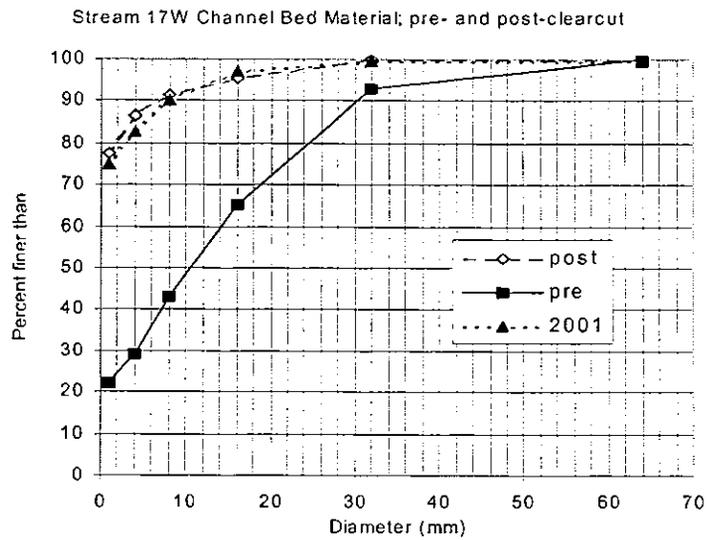
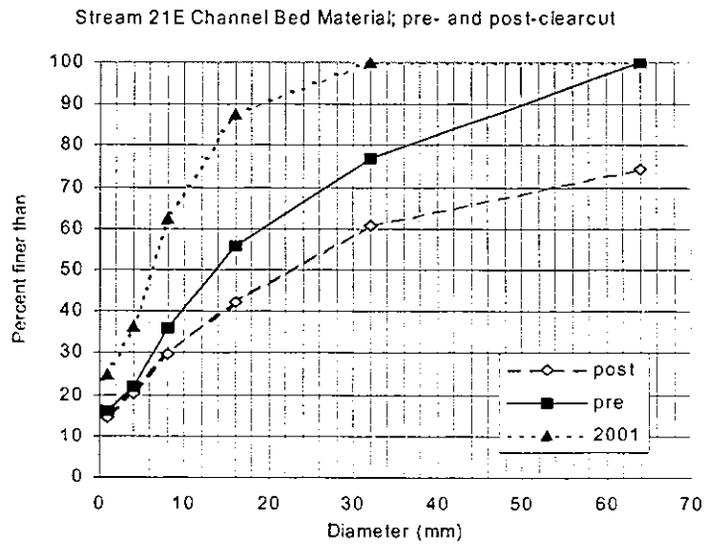


Figure 6.1-17b. Particle size distributions in the clearcut streams; 1998, 1999, and 2000.

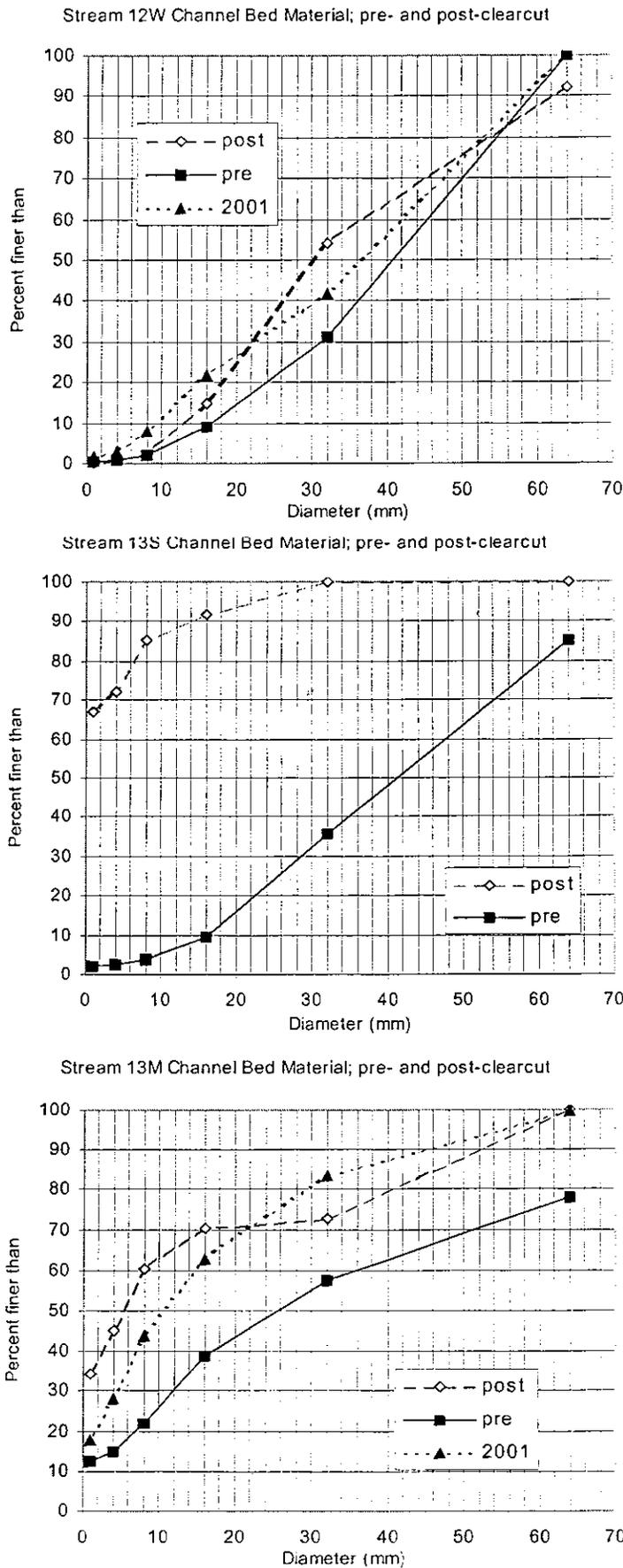


Figure 6.1-18. Percent fines by treatment; 1998, 1999, and 2000.

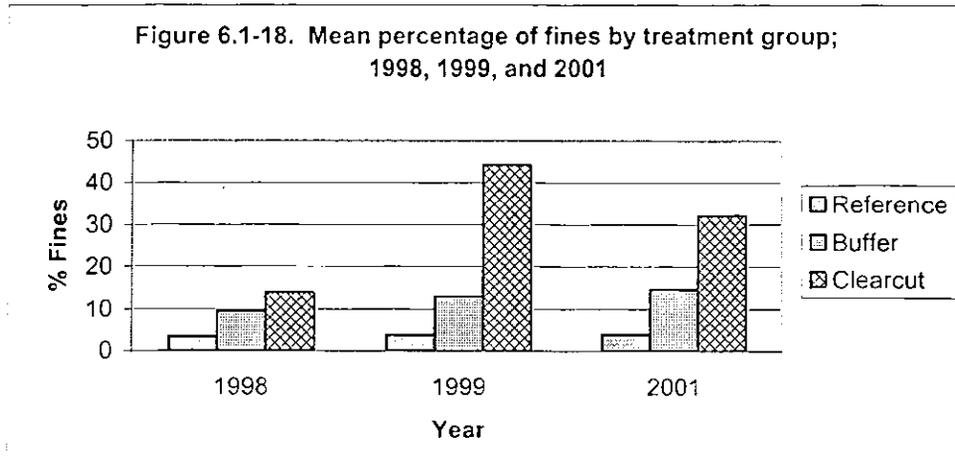
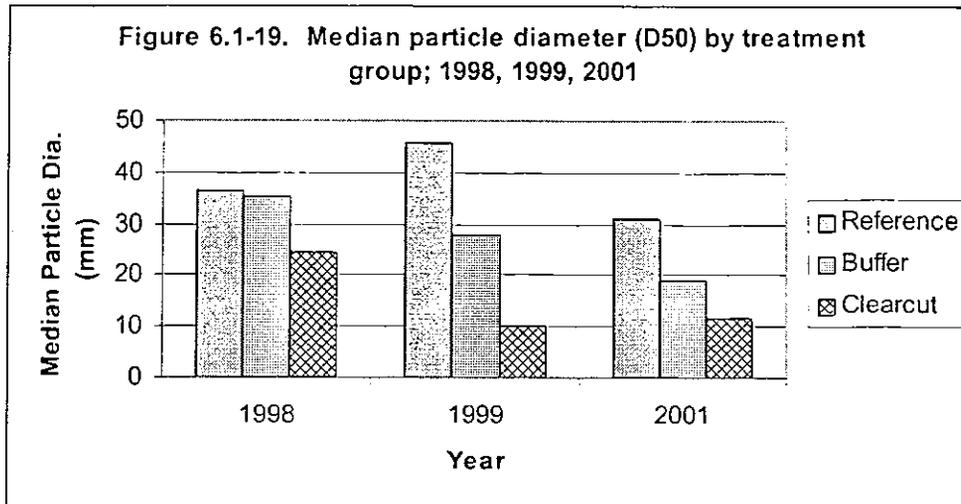


Figure 6.1-19. Median particle size by treatment; 1998, 1999, and 2000.



## Buffer Blowdown

Buffer blowdown was minimal in the first summer after timber harvest, but buffer blowdown was significant two years after harvest. Table 6.1-03 summarizes the amount of blowdown observed in 2001. Blowdown of buffer trees ranged from 33% to 64% with attendant affects on canopy cover. Blowdown of narrow buffers previously has been observed to be a problem in western Washington (e.g. Grizzel and Wolff, 1998). Blowdown of strip buffers may be more common along headwater streams than along fish bearing streams, however, because headwater streams rise higher on the ridges and do not have the topographic protection of being located in the bottom of a valley.

Table 6.1-03. Summary of buffer blowdown

Stream	Buffer Type	Blowdown (2001)	Canopy Cover <sup>†</sup>		
			1998 (pre)	1999	2001
21W	Non-merchantable	44%	90%	65%	10%
21M	Full	52%	93%	15%*	15%*
17E	Full	33%	92%	51%	35%
13E	Full	64%	87%	23%	72% <sup>ss</sup>
12E	Partial (within buffer of fish bearing stream)	42%; 32% along fish bearing stream	95%	NM	90%

<sup>†</sup> These canopy cover estimates should be used with caution. The densiometer readings were taken within the survey section. However, both the buffers and the blowdown of the buffers were patchy, so these numbers are not an average for the whole stream.

\* The buffer on 21M was much wider and denser downstream of the survey reach where these densiometer measurements were taken.

<sup>ss</sup> Canopy coverage on 13E in 2001 was provided by dense Scrub-shrub vegetation growing adjacent to the channel. On this stream the channel-adjacent herbaceous vegetation had grown to a height of 2 meters in many places.

Prior to harvest, sites 12, 13, and 17 featured dense western hemlock stands growing to the stream edge. Little understory existed due to the high crown closure. After harvest, the buffers did not provide a lot of overhead canopy coverage because the trees were tall with branches and needles only at the very top. A lot of light came through the sides of the buffers. Canopy coverage generally decreased after blowdown occurred, although herbaceous vegetation growing along the channels provided shade that was difficult to quantify. By the summer of 2001, the herbaceous vegetation along channel 13E had grown to a height of 2 meters in many places, and densiometer measurements were taken below this herbaceous cover on this stream. In the first year after harvest, water temperatures increased somewhat in the buffered streams, but not in the clearcut streams (Jackson et al. 2001). Temperature increases is described in section 6.2. Water temperatures were not measured in 2001 due to an equipment failure (see section 6.2).

As described in Jackson et al. (2001), the thin riparian buffers effectively prevented the introduction of slash from the harvest area, and they maintained channel habitat structure. After blowdown, all of the newly fallen trees spanned the channels, so until they break down, the blown down trees are not adding woody debris to the channels or altering channel structure.

## 6.2 Stream Temperatures

We used HOBO Tidbit dataloggers to monitor pre- (1998) and post-harvest (1999) temperatures in the study streams to assess the effect of harvest on water temperature. HOBO Tidbit dataloggers were deployed in the streams in summer of 2001, but due to a software glitch between the HOBO dataloggers and Windows 98 operating systems, the dataloggers failed to record data in 2001. Detailed analysis of immediate post-harvest shifts in temperature due to canopy changes are presented in Jackson et al. (2001).

The summer of 1999 was much cooler than the summer of 1998, and this was reflected in the stream temperatures. Using the hourly stream temperatures from the reference streams, relative frequency histograms were made for 1998 and 1999 (Fig 6.2-01). The temperature histograms for the reference streams clearly demonstrate the natural differences in the two summers.

Because of the large differences in 1998 and 1999 summer temperatures, the temperature ranges observed before and after harvest often had little to no overlap, which made it more difficult to assess the effect of timber harvest on temperature. From the hourly temperature observations, daily maximum temperatures were calculated for the pre-harvest summer of 1998 and the post-harvest summer of 1999. The daily maximum stream temperatures were then plotted against the appropriate reference stream and a regression equation was calculated (regression analysis can be found in Jackson et al. (2001)). The slopes of the pre- and post-harvest regression lines were compared using a Student's t-test in a method analogous to using a paired t-test to compare two population means (Zar, 1984). This test allowed us to determine if pre- and post-harvest years have significantly different slopes. Using the regression equations, changes in stream temperature were calculated based on a reference stream temperature of 11 °C which was generally within or near the overlapping range of the 1998 and 1999 data. This approach was used to minimize the amount of extrapolation used to assess the temperature shifts in the streams.

Temperature impacts of timber harvest are summarized in Table 6.2-1. Of the 7 clearcut streams, three exhibited no statistically significant difference in stream temperature, one became cooler (-1.1 °C), one became slightly warmer (+0.8 °C), and the remaining two streams became both cooler and warmer depending on location in the stream. The 12W stream and the 21E stream were longer than the other streams, and two HOBOS were placed in these streams. In both streams, the temperatures at the upper HOBOS became cooler (-2.2 and -1.7 °C) while the temperatures at the lower HOBOS became a lot warmer (+5.2 and +15.1 °C). In short, the temperature in only partial sections of only two of seven clearcut streams increased enough to have a deleterious effect on amphibians. Four of the seven streams either did not change or became cooler. This is strongly counter to the conventional wisdom regarding the effects of clearcutting on stream temperature. In this case, temperatures of the clearcut streams were protected by the layer of slash that was deposited over these streams. The slash effectively shaded the stream and acted as an insulating blanket. Temperature changes in the three buffered streams were not dramatic. Two became warmer (+1.6 and +2.4 °C) and one became slightly cooler (-0.3 °C). The buffers prevented slash from burying these streams, but the remaining overhead canopy prevented large temperature changes.

Despite the initial study design, only one non-merchantable buffer was created in the study. It did prevent slash burial of the stream, but it provided less overhead canopy than the full buffers. Two HOBOs were placed in this stream, and they both detected sizeable temperature increases (+3.7 and +6.6 °C).

**Table 6.2-01.** Estimates of Temperature Impacts. Using the regression equations, changes in stream temperature are calculated based on a reference stream temperature of 11 °C. This temperature is generally within or near the overlapping range of the 1998 and 1999 data and was chosen to avoid extrapolating either the pre- or post-harvest regression relationships beyond their data range.

Stream	Treatment	Pre-harvest temp.	Post-harvest temp.	Temperature change °C
12W – high	clearcut	10.69	8.17	-2.5
12W-middle	clearcut	10.71	27.46	+16.8
12E	clearcut	No significant difference in reference stream relationship		-
13S	clearcut	No significant difference in reference stream relationship		-
13M	clearcut	11.98	10.23	-1.8
17W	clearcut	12.85	14.04	+1.2
17M	clearcut	No significant difference in reference stream relationship		-
21E – high	clearcut	11.99	10.69	-1.3
21E – low	clearcut	9.35	13.28	+3.9
13E	buffer	10.21	12.78	+2.6
17E	buffer	13.64	13.07	-0.5
21M	buffer	11.31	13.30	+2.0
21W – high	non-merch. buffer	12.50	15.34	+2.8
21W – low	non-merch. buffer	11.72	16.62	+4.9

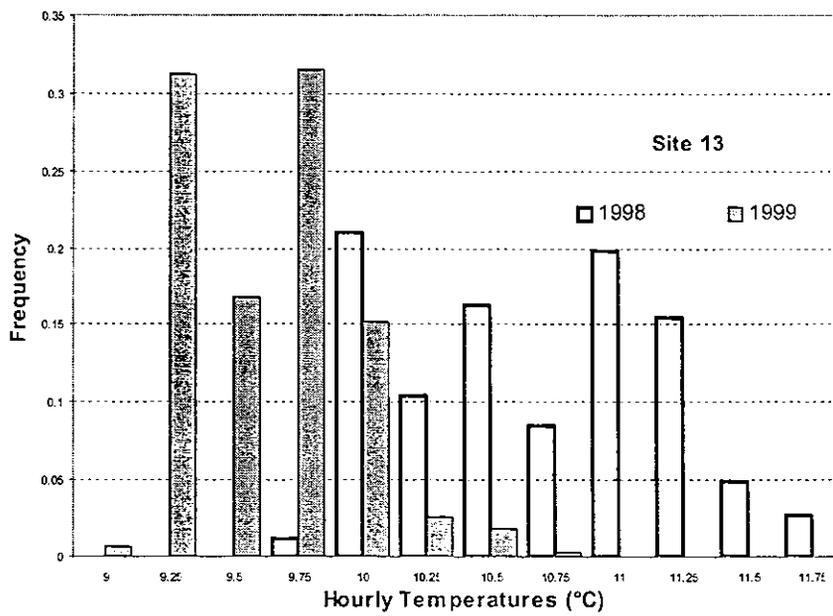
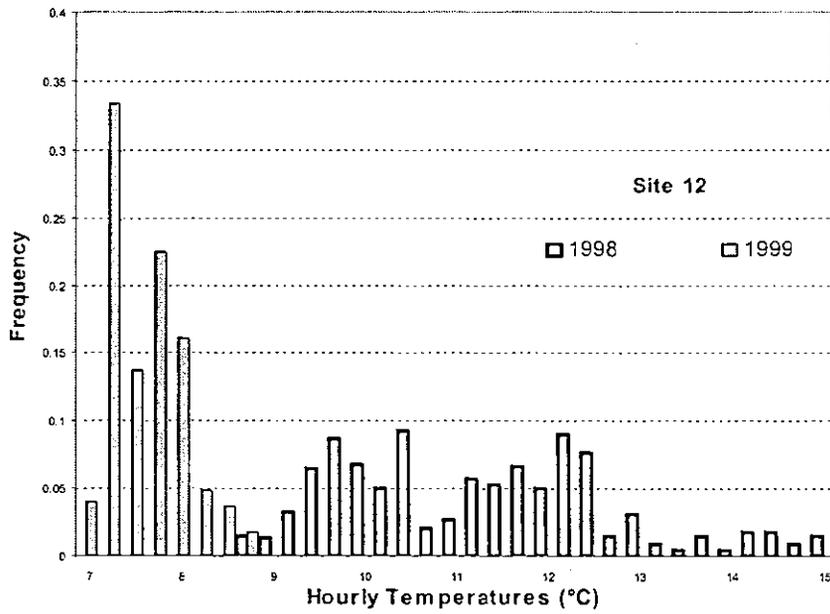


Figure 6.2-01. Selected histograms of hourly stream temperatures in reference streams in 1998 and 1999.

## 6.3 Macroinvertebrates

A complete description of macroinvertebrate communities in these streams and their response to harvest in the first two summers following harvest (1999 and 2000) will soon be available in Haggerty et al. (in press) and Haggerty et al. (in review). Monitoring for macroinvertebrates only was conducted in 2000 and was funded by Rayonier.

### 6.3.1 Headwater stream macroinvertebrate communities

Thirty-one insect and 4 non-insect taxa (genera, subfamily, or family) representing 9 orders and all functional feeding groups were collected from 1<sup>st</sup> order streams (Table 6.3-01). The relative merits of species, genus, and family level identifications are the topic of hot debate in the field of invertebrate bioassessment of streams, with no consensus at this time. No one argues that species level identifications are preferable, but many feel that the large costs required to obtain this taxonomy may not be justified in all cases. Taxonomists argue that species level identification requires verification by experts on the group of interest, and maintain that any species level identifications that are published without such validation should be viewed skeptically. For this project, we did not have a sufficient budget to pay for species level identifications and thus only report a taxonomy that was reliable given that personnel were ecologists rather than taxonomists. Comparisons of our invertebrate taxonomy to other regions were done at comparable levels of classification.

Numerically dominant taxa included Gammaridae (31.4%), Chironomidae (18.6%), Elmidae (7.6%), Nemouridae (7.0%), Empididae (6.8%), Rhyacophilidae (4.7%), and Leuctridae (3.8%). Astacidae (92.7%), Elmidae (3.0%), Gammaridae (1.5%), and Perlidae (0.9%) contributed most to the total biomass. Biomass was probably underestimated for crayfish (Astacidae) because mostly small individuals occurred in substrate baskets. Biomass estimates from the combination salamander-crayfish sampling indicated that crayfish probably composed > 99% of the biomass.

Within each of the 4 watersheds, macroinvertebrate assemblages were similar among streams (ANOVA, all  $p > 0.26$ ). Thus, it was appropriate to use streams as replicates for subsequent analyses. Site 17 had greater taxa richness than Site 12 (Figure 6.3-01A;  $p = 0.02$ ) and both Site 13 and Site 17 had significantly higher mean macroinvertebrate abundances than Site 12 (Figure 6.3-01B;  $p = 0.04$ ). Substrate basket values indicated mean biomass was similar among all 4 watersheds (Figure 6.3-01C). Correlation analyses between physical and invertebrate measures suggested that macroinvertebrate biomass (crayfish included) was positively correlated to stream power (calculated as the product of estimated mean annual flow and channel gradient;  $r = 0.73$ ;  $p = 0.0013$ ) and channel width ( $r = 0.73$ ;  $p = 0.0013$ ). Otherwise, no significant relationships between physical and invertebrate measures were detected.

Taxa richness did not differ significantly among 1<sup>st</sup>-, 2<sup>nd</sup>-, and 3<sup>rd</sup>- order streams (Figure 6.3-02A;  $p = 0.077$ ). However, abundances differed among stream orders (Figure 6.3-02B;  $p = 0.0125$ ), with 2<sup>nd</sup>- order streams having a higher abundance of macroinvertebrates than 1<sup>st</sup>- or 3<sup>rd</sup>- order streams. Abundances in 1<sup>st</sup>- and 3<sup>rd</sup>- order streams were similar. Macroinvertebrate biomass did not significantly differ among 1<sup>st</sup>-, 2<sup>nd</sup>-, and 3<sup>rd</sup>- order streams (Figure 6.3-02C;  $p = 0.8043$ ). Relative abundance and biomass for the different functional feeding groups (Figure 5) varied among stream orders (abundance:  $\chi^2 = 77.6$ ;  $df = 6$ ;  $p < 0.0001$ ; biomass:  $\chi^2 = 78.2$ ;  $df = 6$ ;  $p < 0.0001$ ).

Shredders dominated 1<sup>st</sup>-order streams both in terms of abundance and biomass, but their importance declined in downstream reaches (Figure 6.3-03). In contrast, scraper abundance and biomass increased from 1<sup>st</sup>- to 3<sup>rd</sup>-order reaches. In terms of abundance, macroinvertebrate predators were most numerous in 3<sup>rd</sup>-order reaches, but their biomass remained equal among stream orders. Collectors were equally abundant in all stream orders, but their biomass was much lower in 3<sup>rd</sup>-order than 1<sup>st</sup>- or 2<sup>nd</sup>-order streams, indicating that smaller-bodied collectors dominated downstream reaches.

Macroinvertebrate richness differed significantly among substrate basket types (Figure 6.3-04A;  $p = 0.0118$ ). Post hoc tests indicated that the wood-only and mixed substrate baskets had similar taxa richness, but both of these treatments supported significantly more taxa than did the cobble-only baskets. Macroinvertebrate biomass differed between mixed and cobble-only baskets (Figure 6B;  $p = 0.0449$ ). Relative functional feeding group biomass also differed among substrates (Figure 6.3-05,  $\chi^2 = 51.99$ ;  $df = 6$ ;  $p = <0.0001$ ), with shredders contributing proportionally more and collectors proportionally less biomass in the 2 treatments that included wood than in cobble-only baskets. We chose not to analyze abundance data in this experiment because field sieving of cobble could have differentially affected density estimates from each treatment.

Table 6.3-01. Macroinvertebrate taxa collected in substrate basket samplers (1.5 L wood and 1.5 L cobble) from 15 perennial, 1<sup>st</sup>-order streams in 4 study watersheds of Washington's Coastal Mountains. <sup>a</sup>C = Collectors; P = Predators; Sc = Scrapers; Sh = Shredders.

<i>Taxon</i>	Functional feeding group <sup>a</sup>	Substrate baskets #/m <sup>2</sup> ( $\pm 1$ SE)	<i>Taxon</i>	Functional feeding group <sup>a</sup>	Substrate baskets #/m <sup>2</sup> ( $\pm 1$ SE)
<b>Gastropoda</b>			<b>Trichoptera</b>		
Pleuroceridae			Brachycentridae		
<i>Juga</i>	Sc	3.80 (3.25)	<i>Micrasema</i>	Sh	0.58 (0.29)
<b>Oligochaeta</b>			Hydropsychidae		
Tubificidae	C	2.05 (0.63)	<i>Parapsyche</i>	C	0.59 (0.50)
<b>Amphipoda</b>			Philopotamidae		
Gammaridae			<i>Wormaldia</i>	C	0.29 (0.25)
<i>Gammarus</i>	Sh	49.00 (27.82)	Polycentropodidae		
<b>Decapoda</b>			<i>Polycentropus</i>		
Astacidae			Rhyacophilidae	P	0.29 (0.25)
<i>Pacifastacus</i>	P	1.46 (0.95)	<i>Rhyacophila</i>	P	7.32 (1.44)

Ephemeroptera			Coleoptera		
Baetidae			Elmidae		
<i>Paracloeodes</i>	Sc	0.29 (0.25)	<i>Lara</i>	Sh	11.79 (3.24)
Heptageniidae			Psephenidae		
<i>Cinygma</i>	Sc	0.59 (0.50)	<i>Acneus</i>	Sc	2.63 (1.31)
<i>Ironodes</i>	Sc	0.59 (0.29)	<b>Diptera</b>		
<b>Plecoptera</b>			Athericidae		
Chloroperlidae			<i>Atherix</i>	P	0.58 (0.30)
<i>Haploperla</i>	P	4.10 (1.32)	Ceratopogonidae		
<i>Neaviperla</i>	P	1.17 (0.41)	<i>Bezzia</i>	P	1.70 (1.00)
<i>Suwallia</i>	P	0.29 (0.25)	Chironomidae		
Leuctridae				C	8.78 (4.35)
<i>Despaxia</i>	Sh	5.85 (0.58)	<b>Chironominae</b>		
Nemouridae			<b>Diamesinae</b>		
<i>Soyedina</i>	Sh	8.77 (4.41)		C	13.75 (2.96)
<i>Zapada</i>	Sh	2.05 (1.43)	<b>Orthocladinae</b>		
Peltoperlidae			<b>Tanypodinae</b>		
<i>Soliperla</i>	Sh	0.31 (0.31)	Dixidae		
<i>Yoraperla</i>	Sh	0.87 (0.59)	<i>Dixa</i>	C	3.51 (0.41)
Perlidae			Empididae		
<i>Calineuria</i>	P	0.87 (0.50)	<i>Chelifera</i>	P	10.53 (2.12)
<i>Doroneuria</i>	P	0.87 (0.50)	Tipulidae		
			<i>Dicranota</i>	P	1.17 (0.58)
			<i>Hexatoma</i>	P	2.63 (1.32)
			<i>Pedicia</i>	P	0.88 (0.75)

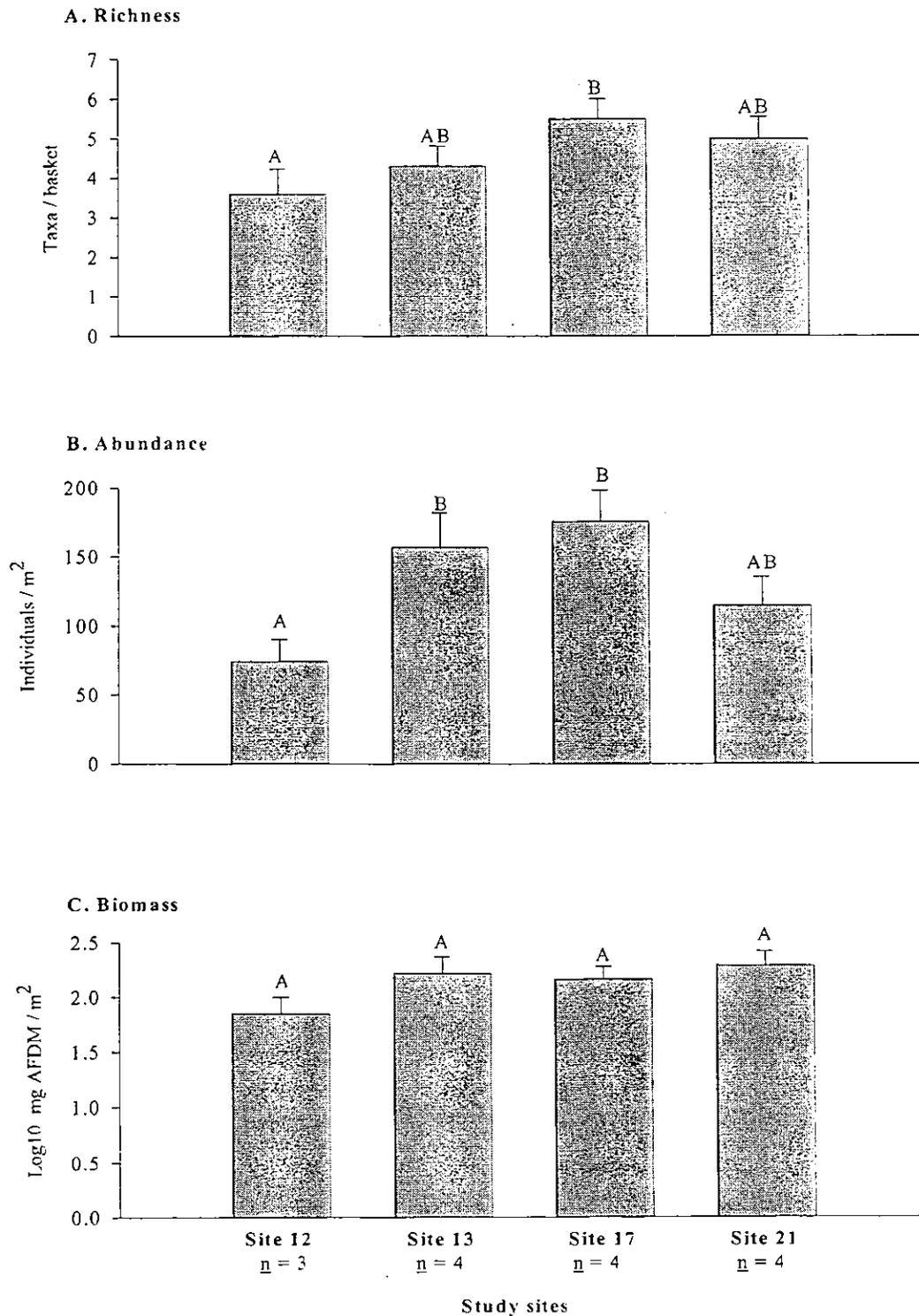


Fig. 6.3-01. A) Taxa richness, B) abundance, and C) non-crayfish biomass of macroinvertebrates from fifteen 1<sup>st</sup> order streams in the 4 study watersheds. Substrate basket samplers (30 x 30 cm) were used for macroinvertebrate collection. Bars indicated by the same letters do not differ significantly (ANOVA and Tukey-Kramer tests). Error bars indicate  $\pm 1$  SE.

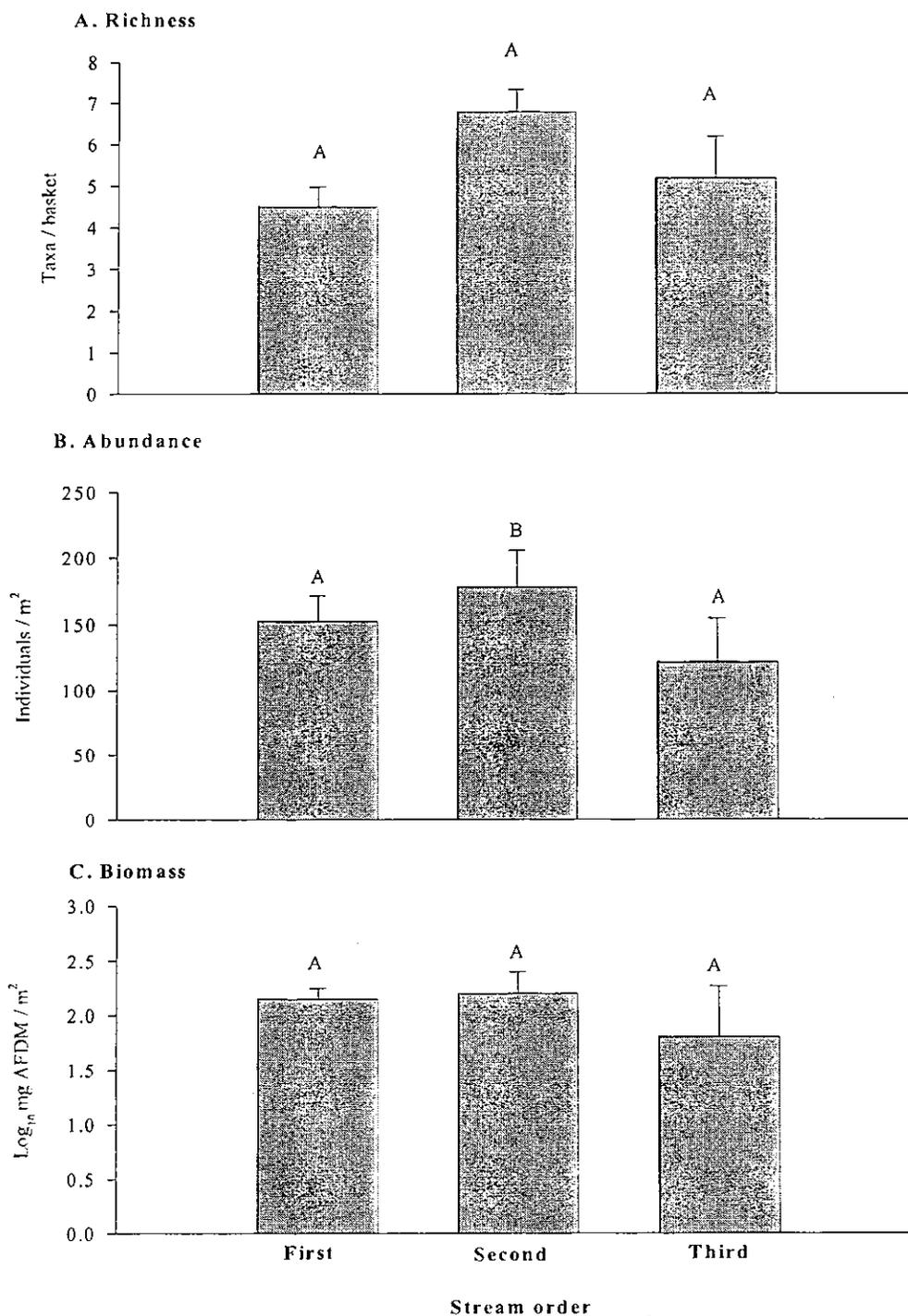
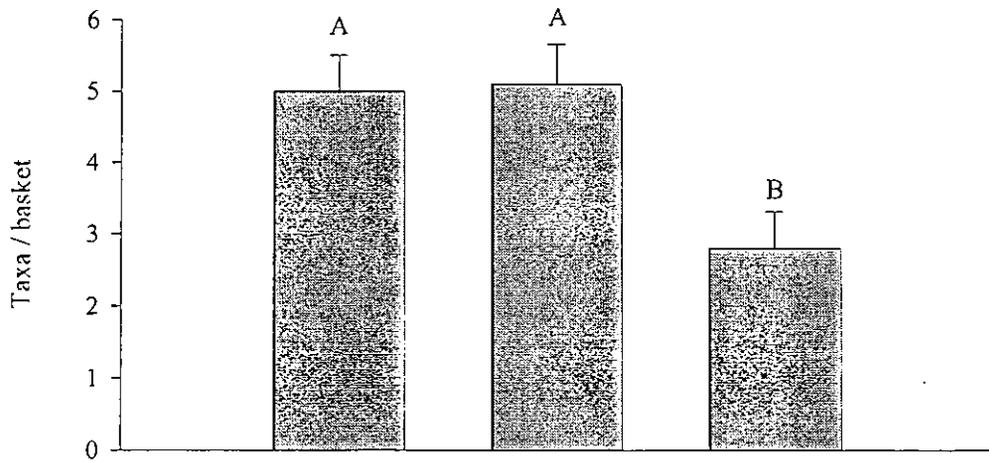


Fig. 6.3-02. A) Taxa richness, B) abundance, and C) non-crayfish biomass of macroinvertebrates among 1<sup>st</sup>, 2<sup>nd</sup>, and 3<sup>rd</sup> order streams. Substrate basket samplers (30 x 30 cm) were used for macroinvertebrate collection. Bars indicated by the same letters do not differ significantly (ANOVA and Tukey-Kramer tests). Error bars indicate  $\pm 1$  SE.

**A. Richness**



**B. Biomass**

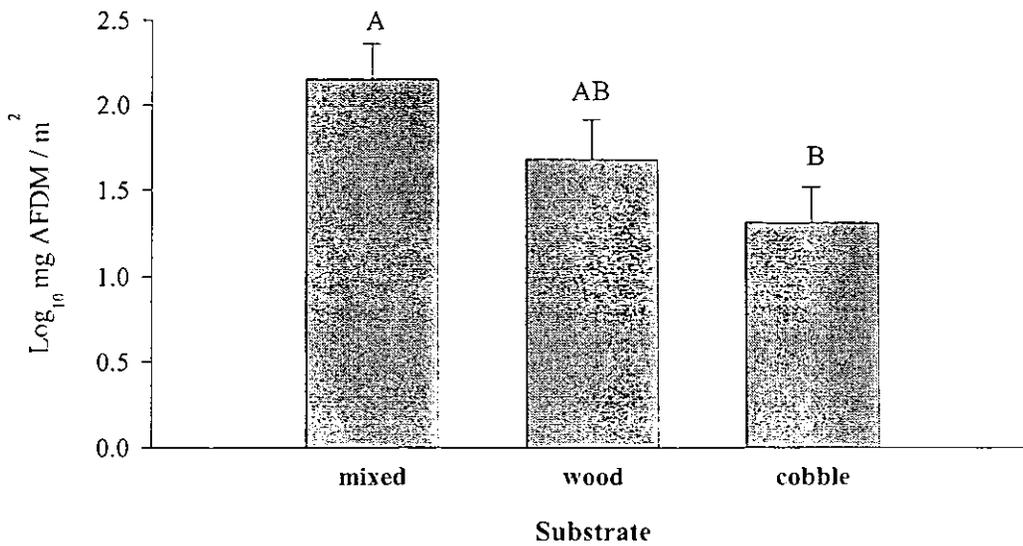


Fig. 6.3-03. A) Taxa richness and B) non-crayfish biomass of macroinvertebrates in mixed wood and cobble, wood-only, and cobble-only samples (substrate basket samplers [30 x 30 cm] each with 3 L of substrate). Bars indicated by the same letters do not differ significantly (ANOVA and Tukey-Kramer tests). Error bars indicate ±1 SE.

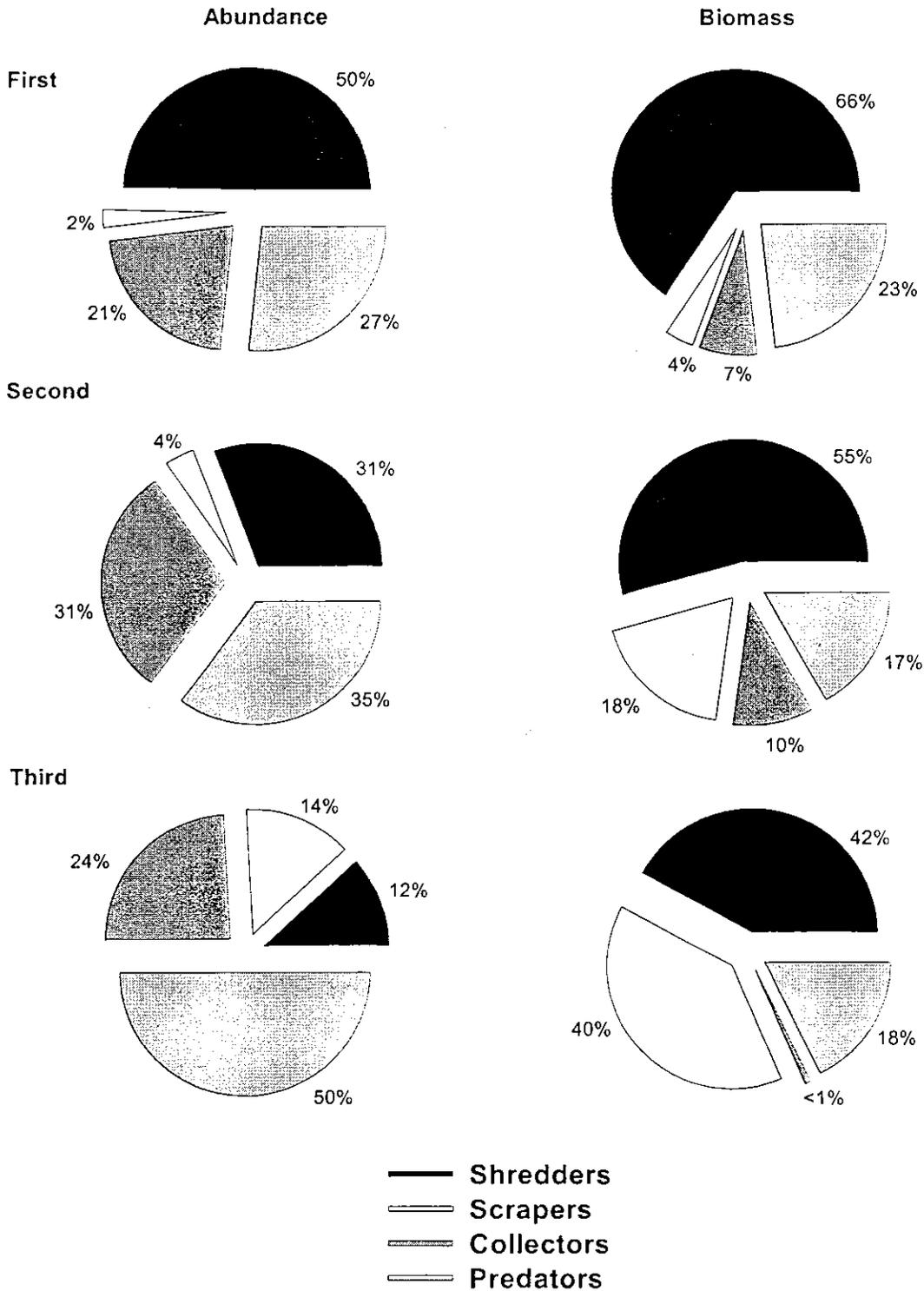


Fig. 6.3-04. Percent composition of functional feeding groups by total abundance and non-crayfish biomass for A) 1<sup>st</sup> order streams (152 individuals / m<sup>2</sup>; 138.0 mg AFDM / m<sup>2</sup>), B) 2<sup>nd</sup> order streams (177 individuals / m<sup>2</sup>; 163.4 mg AFDM / m<sup>2</sup>), and C) 3<sup>rd</sup> order streams (121 individuals / m<sup>2</sup>; 181.8 mg AFDM / m<sup>2</sup>).

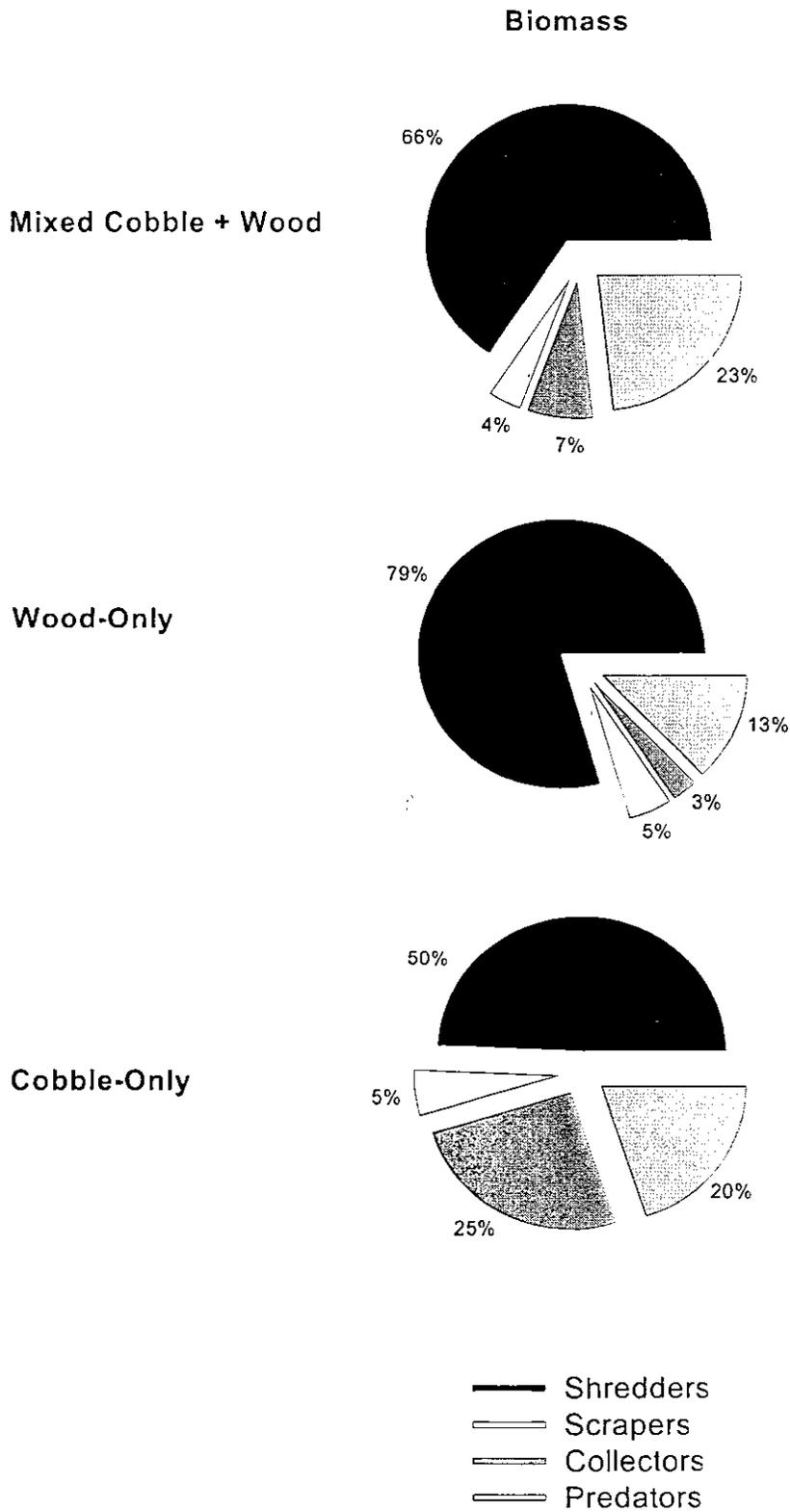


Fig. 6.3-05. Percent composition of functional feeding groups by non-crayfish biomass for A) mixed baskets (167.27 mg AFDM / m<sup>2</sup>), B) wood-only baskets (130.2 mg AFDM / m<sup>2</sup>), and C) cobble-only baskets (59.7 mg AFDM / m<sup>2</sup>).

## 6.3.2 Macroinvertebrate response to timber harvest

### 6.3.2A Initial responses (1999 and 2000)

In reference streams, we did not detect any significant change among years (pre-harvest year and 2 post-harvest years) for: 1) macroinvertebrate taxa richness, abundance, or biomass (Fig 6.3-06); 2) FFGs in terms of relative abundance or biomass (Fig. 6.3-07); or 3) amount organic and inorganic matter accretion (Fig. 6.3-08). The variation among logging treatments reported below was unlikely to result from simple year-to-year variation.

In the initial years after harvest, two-way ANOVAs indicated that macroinvertebrate richness ( $F_{2,33} = 30.3; p < 0.0001$ ), abundance ( $F_{2,33} = 5.8; p = 0.007$ ), and biomass ( $F_{2,33} = 5.1; p = 0.001$ ) varied, with 1999 being significantly higher than the 1998 baseline year or the 2000 summer, one year after harvest (Fig. 6.3-06). Logging treatments did not influence macroinvertebrate richness. However, macroinvertebrate abundance differed significantly among logging treatments ( $F_{2,33} = 4.6; p = 0.02$ ), with abundances in clear-cut streams being higher than in reference streams. After conducting one-way analyses, we found that this treatment response in abundance was only pronounced for 1999 (Fig. 6.3-06B;  $F_{2,12} = 4.23, p = 0.04$ ). Overall abundance in all streams was similar during both 1998 and 2000. Macroinvertebrate biomass also differed significantly among logging treatments ( $F_{2,33} = 5.8; p = 0.02$ ), with buffered streams supporting more biomass than reference or clear-cut streams. After conducting one-way analyses, we found that this treatment response in abundance was only pronounced for 1999 (Fig. 6.3-06C;  $F_{2,12} = 6.3, p = 0.005$ ). Overall biomass in all streams was similar during both 1998 and 2000.

Collector abundance ( $F_{2,33} = 23.5; p = 0.0005$ ) and biomass ( $F_{2,33} = 21.2; p = 0.005$ ) varied among years, with 1999 being higher than 1998 or 2000. While the overall two-way ANOVA did not detect treatment effects, one-way analyses indicated that in 1999 collectors were more abundant in clear-cut streams than reference or buffered streams (Fig. 6.3-07A;  $F_{2,12} = 7.8, p = 0.01$ ). Collector response in 1999 was due almost solely to chironomid increases, as they constituted 90.3% of that FFG.

Shredder abundance was similar among years. Significant logging treatment effects were evident in both two-way and one-way analyses. One-way ANOVAs indicated that shredders were more abundant in clear-cut and buffered streams than reference streams in 1999 (Fig. 6.3-07B;  $F_{2,12} = 6.2; p = 0.02$ ). Shredder biomass was similar among years. Significant logging treatment effects were evident in both two-way and one-way analyses. One-way ANOVAs showed clear-cut and buffered streams supporting higher biomass of shredders than reference streams immediately after logging ( $F_{2,12} = 7.4; p = 0.01$ ). Gammarid amphipods (46.9%) and nemourid stoneflies (36.5%) dominated the shredder group.

Scraper abundance ( $F_{2,33} = 9.9; p = 0.0008$ ) and biomass ( $F_{2,33} = 9.7; p = 0.0008$ ) varied among years, with 1999 being higher than 1998 or 2000. While the overall two-way ANOVA did not detect treatment effects, one-way analyses indicated that in 2000 scraper biomass was higher in clear-cut streams than reference streams (Fig. 6.3-07C;  $F_{2,12} = 4.5, p = 0.05$ ). Predator abundance and biomass were similar among logging treatments. Yearly differences were detected for abundance ( $F_{2,33} = 10.7; p = 0.0005$ ) and biomass ( $F_{2,33} = 3.4; p = 0.05$ ), with 1999 being higher

than 1998 or 2000. In clear-cut streams relative densities and biomass of all 4 FFGs were similar in baskets covered with slash or open to sunlight.

Two-way ANOVAs for individual taxa indicated that *Paracloedes*, *Sweltsa*, Chironominae, Orthocladinae, Tanypodinae, and *Dixidae* (all  $p < 0.05$ ) were more abundant in 1999, than in 1998 or 2000. While the overall two-way ANOVA did not detect logging treatment effects for any taxa, one-way analyses indicated that *Leucrocuta* mayflies were more abundant in buffered streams than in clear-cut or reference streams in 1999 ( $F_{2,12} = 4.3$ ;  $p = 0.03$ ). All other taxa had similar densities in all streams during all years (Table 1).

Following logging, 48% of the clear-cut channel length was covered by logging debris (i.e. a recognizable channel still existed below the debris) and 50% of the clear-cut channel length was buried (i.e. flow moved through a matrix of organic debris and sediment without a recognizable channel). The average depth of coverage and burial was 0.97 meters. The logging debris trapped fine sediments in the channels, increasing the percentage of fines from an average of 12% to 44%.

Organic and inorganic matter accretion on tiles was similar among years (Fig. 6.3-08). Significant logging treatment effects were evident in both two-way and one-way analyses for both organic and inorganic matter. One-way ANOVAs showed clear-cut and buffered streams contributed more organic matter (Fig 4A;  $F_{2,12} = 25.5$ ;  $p < 0.0001$ ) and inorganic matter (Fig 6.3-08B;  $F_{2,12} = 6.3$ ;  $p = 0.01$ ) accretion on tiles than reference streams in 1999. Organic matter accretion remained elevated in buffered streams in 2000 ( $F_{2,9} = 4.6$ ;  $p = 0.04$ ).

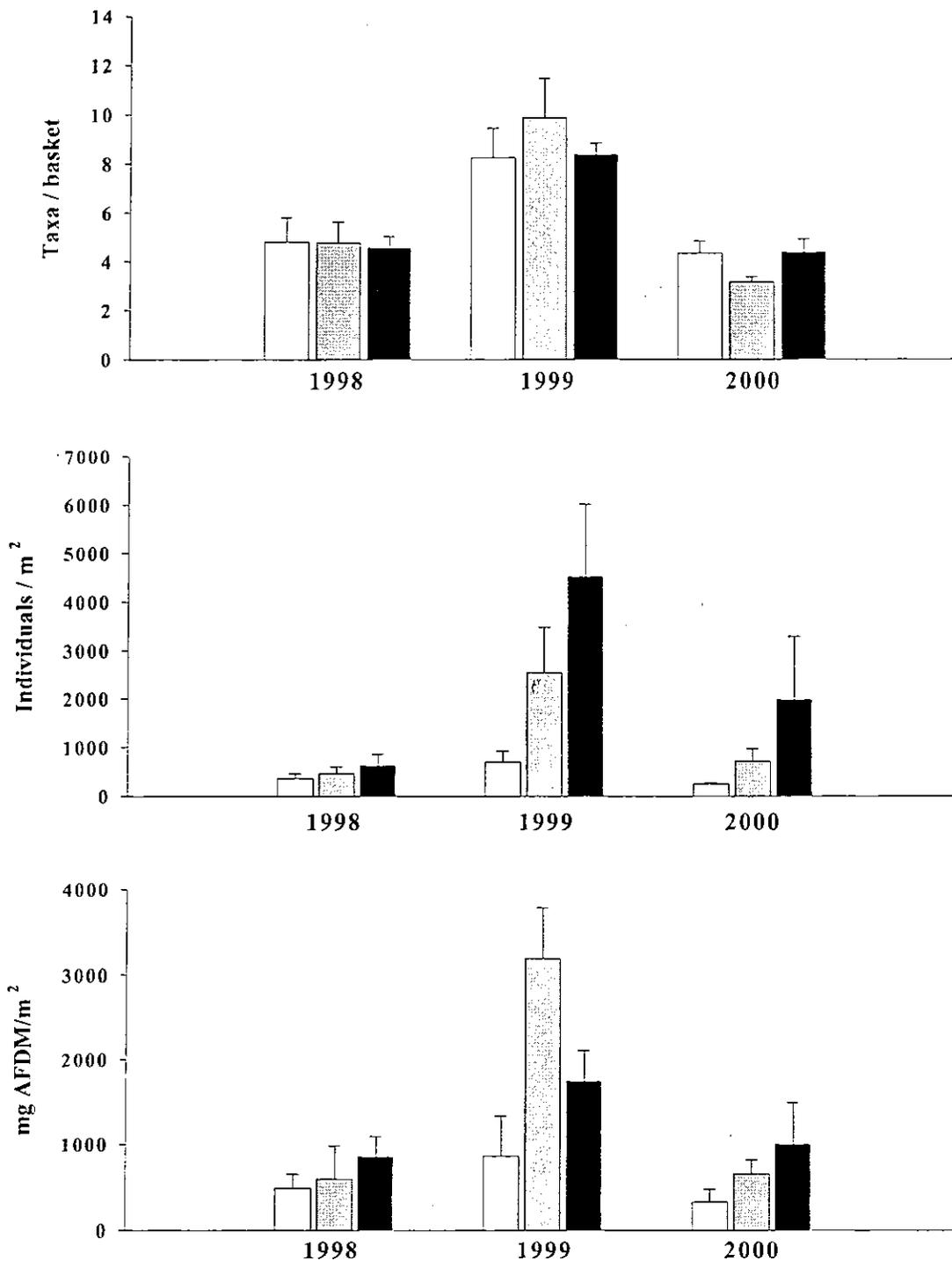


Fig. 6.3-06. Comparison of macroinvertebrate (A) taxa richness, (B) abundance, and (C) non-crayfish biomass in clear-cut, buffered, and reference streams in 1998, 1999, and 2000. Substrate basket samplers (30 x 30 cm) were used for macroinvertebrate collection. Error bars indicate  $\pm 1$  SE. Richness, abundance, and biomass were significantly higher in 1999 than in 1998 or 2000. During 1999, clear-cut streams supported significantly more macroinvertebrates than reference streams. Otherwise, treatments were similar within and among years.

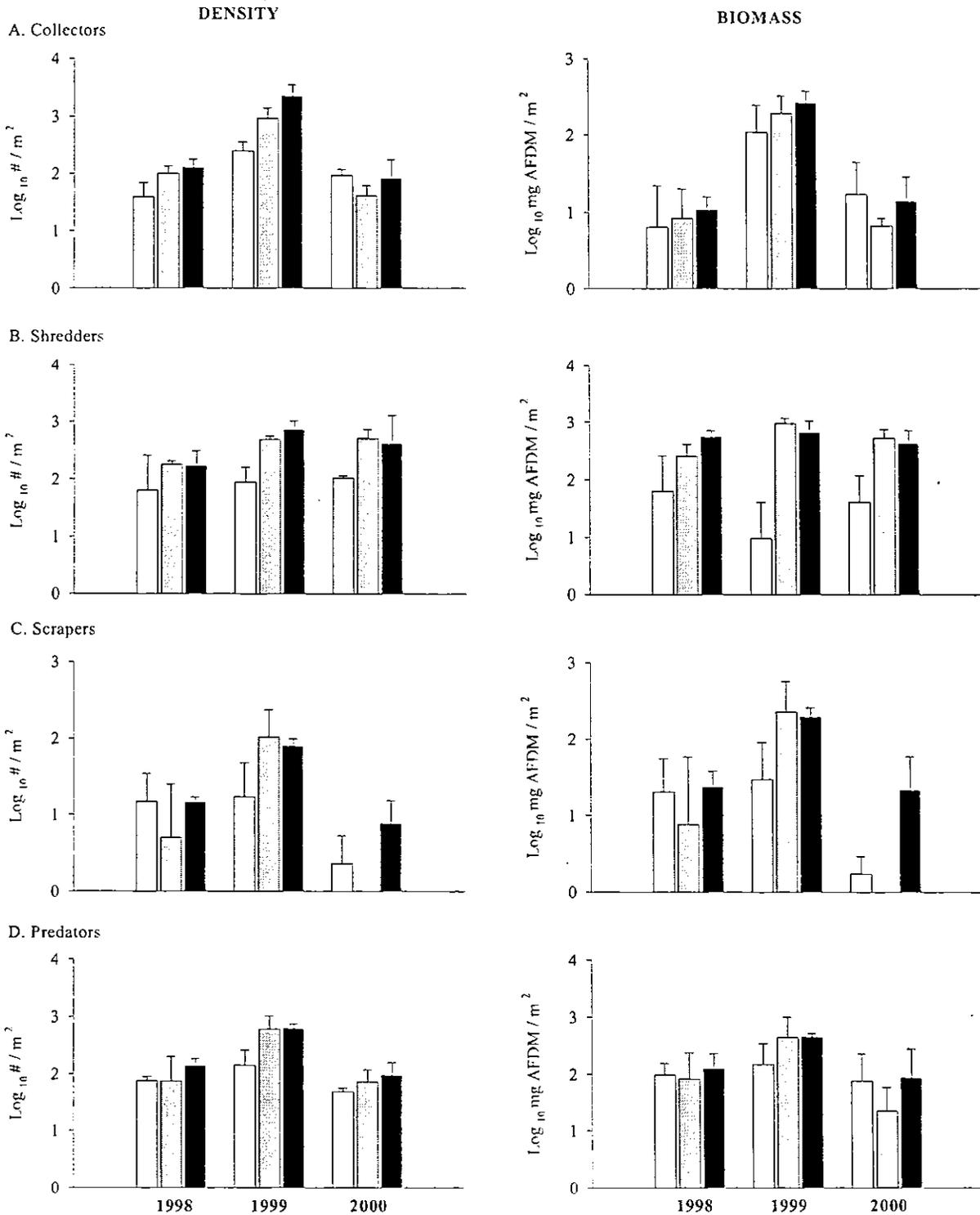


Fig. 6.3-07. Relative densities and biomass of (A) collectors, (B) shredders, (C) scrapers, and (D) predators in clear-cut, buffered, and reference streams in 1998, 1999, and 2000. Error bars indicate  $\pm 1$  SE. The abundance and biomass of each group was significantly higher in 1999 than 1998 or 2000. Collector and shredder abundance and biomass was significantly higher in clear-cut and buffered streams as compared to reference streams in 1999. Shredder biomass remained elevated in treatment streams in 2000. Logging treatments influenced neither scrapers nor predators.

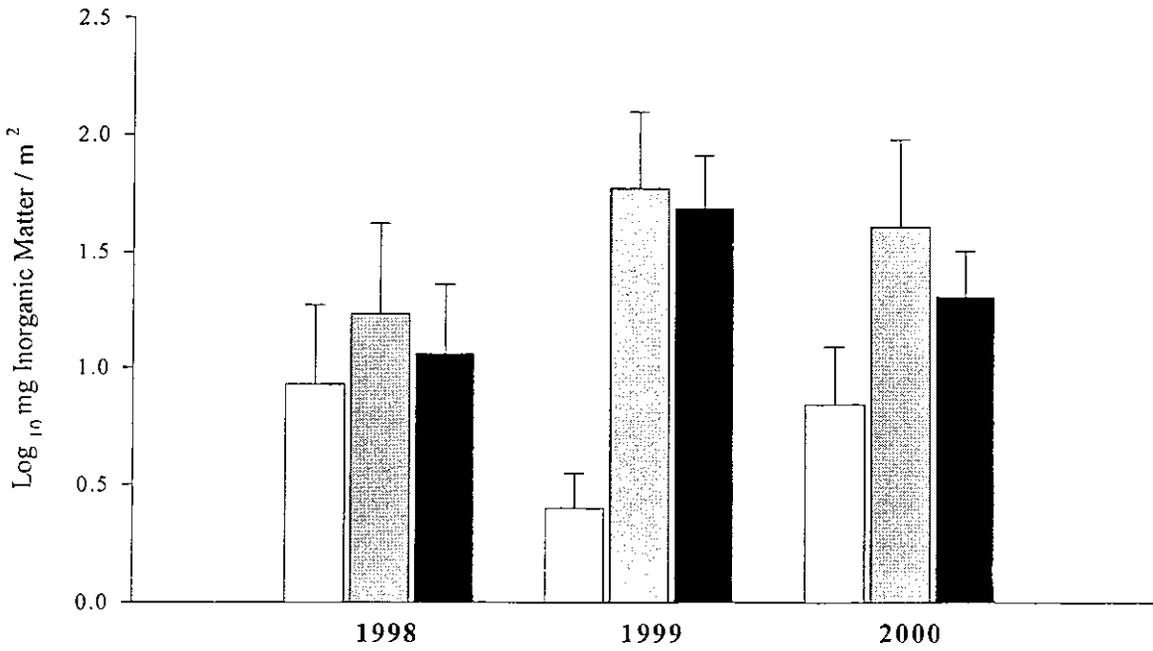
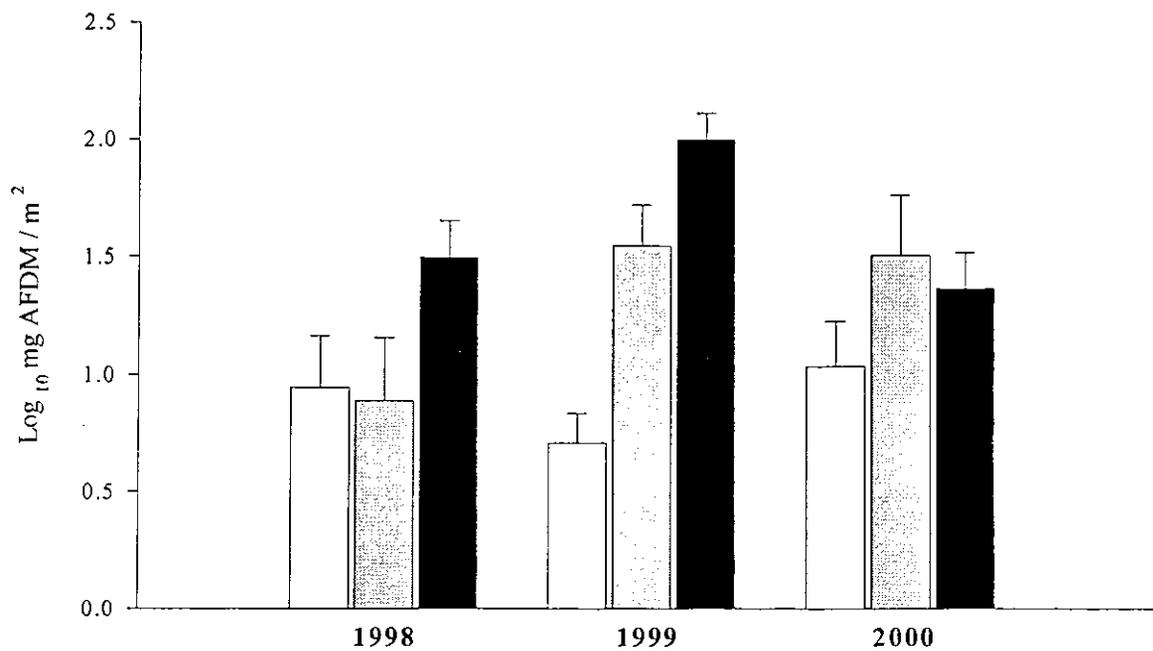


Fig. 6.3-08. (A) Amount organic and (B) inorganic matter accretion on stone tiles placed in clear-cut, buffered, and reference streams in 1998, 1999, and 2000. Error bars indicate  $\pm 1$  SE. Organic and inorganic matter accretion was similar in each year. Organic matter accretion was significantly higher in clear-cut and buffered streams than reference streams in 1999, and accretion remained elevated in buffered streams in 2000. Inorganic matter accretion was higher in buffered and clear-cut streams than reference streams in 1999, but was similar among treatments in 2000.

### 6.3.2B Two-year Response ( 2001)

Two years post-harvest, statistical analyses contrasting invertebrate abundances among reference, clearcut, and buffered streams were not significant (for total number responses see Fig. 6.3-09). An overall response was not found because responses by macroinvertebrates were not consistent among watersheds. In Watershed 12, the reference stream continued to support low densities of macroinvertebrates, as in past years. In one of the two clearcut streams in that watershed, higher densities of fly larvae (Diptera, mostly Chironomidae midges) were observed, but the remaining fauna of this stream was similar to the reference. In the second clearcut stream, higher numbers of fly larvae were coupled with large numbers mayfly (Ephemeroptera), stonefly (Plecoptera), and caddisfly (Trichoptera) immatures. These "EPT" organisms are typically considered to be environmentally sensitive and ecologically desirable.

Fig. 6.3-09. Mean densities of macroinvertebrates in reference, clearcut, and buffered streams. Differences among treatments are not statistically different ( $P = 0.303$ , 2-way ANOVA, log-transformed data). Y-axis is macroinvertebrates/ $m^2$ .

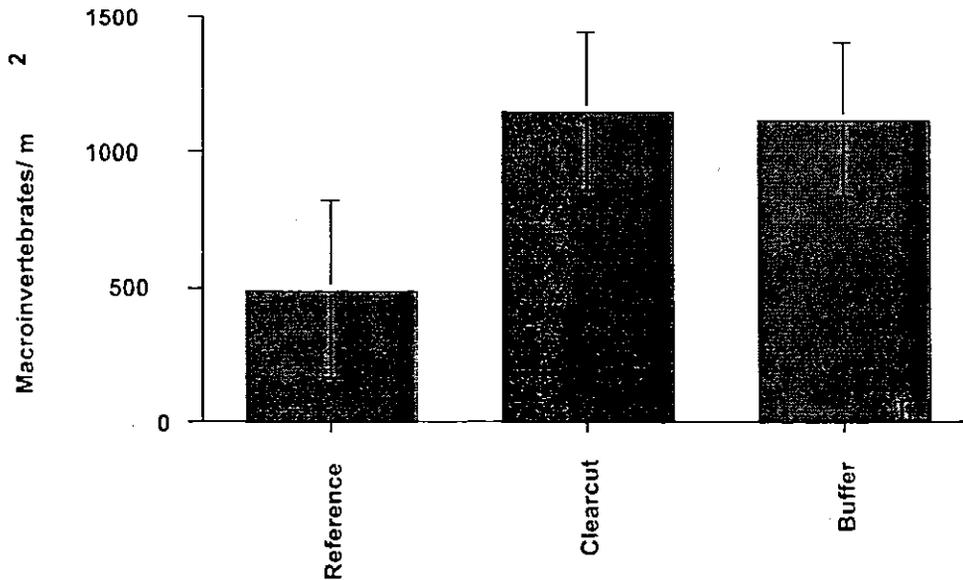


Table 1. Densities (number · m<sup>-2</sup> ± 1 standard error) of macroinvertebrates collected from perennial 1<sup>st</sup> - order streams in 4 study watersheds of Washington's Coastal Mountains, USA. Substrate basket samplers (1.5 L wood and 1.5 L cobble) were used to sample macroinvertebrates in these streams prior to logging (1998), immediately after logging (1999), and 1 yr following logging (2000). *Leucrocuta* was more abundant in buffered streams than clear-cut or reference streams in 1999. All other taxa were similar among logging treatments. *Paracloeodes*, *Sweltsa*, Chionominae, Orthocladinae, Tanypodinae, and *Dixa* were significantly more abundant in 1999 than 2000. <sup>a</sup> C = Collectors; P = Predators; Sc = Scrapers; Sh = Shredders.

<i>Taxon</i>	FFG <sup>a</sup>	Clear-cut 1999	Buffer 1999	Reference 1999	Clear-cut 2000	Buffer 2000	Reference 2000
Gastropoda							
Pleuroceridae							
<i>Juga</i>	Sc	2.43 (1.56)	12.03 (7.17)	3.33 (3.33)	0.47 (0.47)	0	0
Oligochaeta							
Tubificidae	C	11.80 (2.33)	6.48 (3.22)	7.78 (2.37)	2.78 (1.92)	0.92 (0.92)	3.67 (2.62)
Amphipoda							
Gammaridae							
<i>Gammarus</i>	Sh	46.66 (17.0)	82.55 (30.77)	35.11 (30.77)	194.44 (79.99)	105.55 (49.33)	6.18 (4.89)
Decapoda							
Astacidae							
<i>Pacifastacus</i>	P	0.34 (0.34)	1.86 (1.36)	0	1.39 (1.39)	0	1.23 (1.23)
Ephemeroptera							
Bactidae							
<i>Paracloeodes</i>	Sc	1.39 (0.66)	4.63 (2.56)	6.67 (3.78)	0	0	0
Heptageniidae							
<i>Ironodes</i>	Sc	5.55 (2.34)	1.86 (1.86)	6.67 (2.97)	0.47 (0.47)	0	0
<i>Leucrocuta</i>	Sc	2.78 (1.73)	13.89 (5.82)	0	0	0	0
Leptophebiidae							
<i>Paraleptophebia</i>	C	4.17 (4.17)	0	0	0	0	0
Plecoptera							
Capniidae							
<i>Paracapnia</i>	Sh	2.10 (1.67)	0	0	1.86 (1.09)	0	0
Chloroperlidae							
<i>Haploperla</i>	P	5.55 (2.96)	0.92 (0.92)	2.22 (2.22)	0	0	0
<i>Neaviperla</i>	P	3.70 (0.79)	1.86 (1.36)	18.89 (11.10)	0	0	0
<i>Sweltsa</i>	P	9.38 (2.50)	10.19 (5.88)	0	0	0	0
Leuctridae							
<i>Despaxia</i>	Sh	4.89 (1.87)	18.62 (7.05)	16.66 (10.43)	10.70 (3.24)	0	6.18 (1.96)

<i>Taxon</i>	FFG <sup>a</sup>	Clear-cut 1999	Buffer 1999	Reference 1999	Clear-cut 2000	Buffer 2000	Reference 2000
<i>Paraleuctra</i>	Sh	2.78 (2.78)	1.86 (1.24)	0	0	0	0
Nemouridae							
<i>Nemoura</i>	Sh	0	0.92 (0.92)	0	7.41 (6.11)	0.92 (0.92)	0
<i>Soyedina</i>	Sh	21.62 (10.12)	2.89 (2.89)	0	45.77 (26.55)	5.67 (4.63)	16.89 (3.78)
<i>Zapada</i>	Sh	28.54 (9.47)	15.00 (7.33)	22.22 (7.00)	13.89 (6.90)	1.89 (1.89)	11.11 (8.55)
Peltoperlidae							
<i>Soliperla</i>	Sh	1.04 (0.58)	0.92 (0.92)	4.44 (2.22)	0	0	0
<i>Yorapera</i>	Sh	2.09 (1.76)	0	2.22 (1.48)	1.39 (0.77)	0	1.22 (1.22)
Perlidae							
<i>Calineuria</i>	P	0	0	1.11 (1.11)	0	0	2.44 (1.67)
<i>Doroneuria</i>	P	0	0	3.33 (1.70)	1.86 (1.09)	0	0
Perlodidae							
<i>Isoperla</i>	P	0.34 (0.34)	0	0	0	0	0
<i>Rickera</i>	P	1.04 (0.77)	1.86 (1.24)	0	0	0	0
Taeniopterygidae							
<i>Doddsia</i>	Sc	0.34 (0.34)	0	0	0	0	0
Trichoptera							
Brachycentridae							
<i>Eobrachycentrus</i>	Sh	0	0.92 (0.92)	0	0	0	0
Hydropsychidae							
<i>Parapsyche</i>	C	0	0	7.78 (5.51)	0	0	0
Lepidostomatidae							
<i>Lepidostoma</i>	Sh	0	1.86 (1.36)	0	0	0	0
Limnephilidae							
<i>Hydatophylax</i>	Sh	0.34 (0.34)	0	2.22 (1.48)	0.92 (0.64)	0	2.44 (2.44)
<i>Psychoglypha</i>	C	1.04 (0.77)	4.63 (3.20)	1.11 (1.11)	0	0	0
Psychomyiidae							
<i>Psychomyia</i>	C	1.39 (0.58)	1.86 (1.36)	4.44 (3.4)	0	0	0
Rhyacophilidae							
<i>Himalopsyche</i>	P	1.04 (0.58)	6.48 (3.20)	0	0	0	0
<i>Rhyacophila</i>	P	7.99 (2.44)	19.44 (8.44)	17.78 (8.55)	8.8 (2.77)	6.48 (4.82)	1.22 (1.22)
Coleoptera							
Dryopidae							
<i>Helichus</i>	Sh	1.39 (0.66)	0.92 (0.92)	0	0	0.92 (0.92)	0
Elmidae							
<i>Lara</i>	Sh	7.66 (2.66)	10.20 (3.98)	2.22 (1.44)	3.24 (1.24)	17.61 (8.55)	2.44 (1.63)

<i>Taxon</i>	FFG <sup>a</sup>	Clear-cut 1999	Buffer 1999	Reference 1999	Clear-cut 2000	Buffer 2000	Reference 2000
Psephenidae							
<i>Acneus</i>	Sc	1.98 (0.89)	0.92 (0.92)	5.55 (3.78)	0.92 (0.64)	0	1.22 (1.22)
Diptera							
Athericidae							
<i>Atherix</i>	P	1.39 (0.66)	1.86 (1.86)	0	0	0	0
Ceratopogonidae							
<i>Bezzia</i>	P	0.69 (0.48)	0	0	1.39 (1.02)	0.92 (0.92)	1.22 (1.22)
<i>Ceratopogon</i>	P	0.69 (0.48)	0	0	0		
Chironomidae							
<b>Chironominae</b>	C	82.23 (16.55)	66.66 (30.86)	13.33 (5.5)	8.80 (5.09)	3.67 (2.09)	1.22 (1.22)
<b>Diamesinae</b>	C	30.11 (7.87)	10.19 (5.71)	22.22 (2.22)	0	3.67 (3.67)	2.44 (2.44)
<b>Orthocladinae</b>	C	264.40 (72.94)	115.66 (31.44)	64.44 (9.44)	16.22 (6.66)	2.78 (2.78)	20.89 (8.33)
<b>Tanypodinae</b>	P	36.20 (7.88)	79.55 (37.55)	3.33 (2.37)	6.02 (3.28)	1.86 (1.24)	0
Dixidae							
<i>Dixa</i>	C	15.99 (5.89)	36.11 (17.54)	7.78 (3.78)	2.31 (0.94)	0.92 (0.92)	4.89 (3.78)
Empididae							
<i>Chelifera</i>	P	12.19 (4.33)	8.33 (4.37)	7.78 (2.38)	2.31 (0.94)	10.19 (3.16)	4.89 (3.78)
Pelecorhynchidae							
<i>Glutops</i>	P	0.69 (0.48)	0	2.22 (2.22)	0	0.92 (0.92)	2.44 (1.67)
Psychodidae							
<i>Pericoma</i>	C	0	0	0	0	0.92 (0.92)	0
Tipulidae							
<i>Dicranota</i>	P	1.39 (0.97)	3.67 (3.67)	0	0.92 (0.64)	0	1.22 (1.22)
<i>Hexatoma</i>	P	5.55 (1.93)	2.78 (1.99)	3.33 (2.37)	0.47 (0.47)	0.92 (0.92)	1.22 (1.22)
<i>Pedicia</i>	P	0	0.92 (0.92)	0	0	0.92 (0.92)	0
Sciaridae							
<i>Sciara</i>	C	1.04 (0.58)	0.92 (0.92)	2.22 (1.44)	0	0	0

In Watershed 13, fly larvae were generally higher in the clearcut and buffered streams than the reference, but no response by EPT organisms was evident. Instead amphipod crustaceans became very abundant in clearcut and buffered streams, a response largely unique to this watershed. In Watershed 21, no differences among streams was evident, but for unknown reasons the references was much more productive for macroinvertebrates than it had been in past years. Responses in Watershed 17 could not be related to reference conditions because that the trees around that watershed's reference stream had been harvested. However, streams there were the only ones encountered where oligochaete worms were a dominant organism. The clearcut stream in Watershed 17 had very low numbers of EPT organisms, while the buffered stream support relatively high densities of these organisms.

We have yet to analyze diversity or responses of individual taxa, some of which might be particularly sensitive to harvest induced change. We are also working towards assessing biomass responses, which will provide a better picture of ecosystem response than abundance data. Finally, population and biomass response by different functional feeding groups (shredders, collectors, scrapers, predators) must yet be assessed; functional response is a particularly useful measure of ecosystem response.

## 6.4 AMPHIBIANS

### 6.4.1. Headwater Stream Amphibian Habitat Correlates

#### *Variation in Amphibian Communities Across Lithotopo Units*

As shown in Figure 6.4.1, species composition and abundance varied greatly among the four sites. The steep basalt sites, sites 12 and 13, featured all three of the study genera and the highest abundances of animals. Site 21, with gentle topography and highly weathered marine basalts, featured only giant salamanders in low numbers. The Grays Harbor site (site 17), with intermediate channel slopes and sideslopes and a mixed marine sediment and glacial deposit geology (see Table 4.01), featured both torrents and giants, but in relatively low numbers. All four of these sites are within the mapped distributions of the three genera (Corkran and Thoms 1996, Leonard et al. 1993), but local geologic and topographic conditions appear to drive presence and abundance. This finding is consistent with the observation by Wilkins and Peterson (2000) regarding the importance of lithotopo units in stream-dwelling amphibian assemblages.

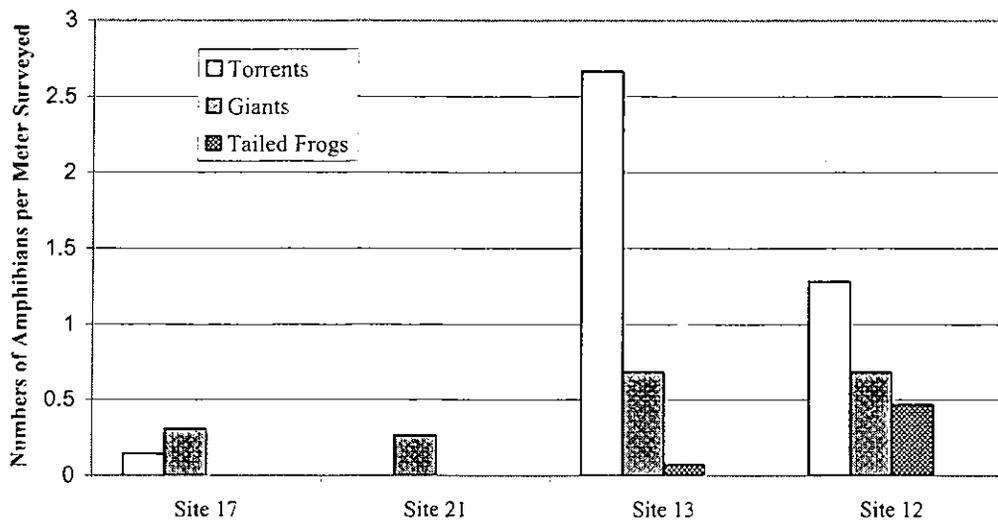


Figure 6.4.1. Species of Amphibians Found Pre-treatment per Site in Numbers per Meter Surveyed

#### Reach-Scale Habitat Associations

During the surveys of the 15 streams used to evaluate relationships between amphibian densities and reach-scale habitat characteristics, we collected 257 amphibians within the channel banks. This total included 158 torrents, 78 giants, 18 tailed frogs and 3 “other” terrestrial amphibians. Although these “other” amphibians were collected from within the channel survey area, so few numbers existed that we removed them from further “by species” analysis. In addition, too few streams contained tailed frogs to run regressions against reach-scale characteristics. Both tailed frogs and “other” salamanders were included in calculations of total amphibians.

In regressions of *Dicamptodon* densities against reach-scale channel characteristics, no variables were significant. It is possible that this result is due to generalist behavior by *Dicamptodon* as a genus, or that this is an artifact of merging *D. tenebrosus* and *D. copei* in the statistical analysis. This analysis is predicated on the assumption that *D. tenebrosus* and *D. copei* require and prefer similar habitat. Total amphibian density was positively related to functional large woody debris per channel width ( $p=0.003$ ; Table 6.4.1 and Figure 6.4.2).

Percent pool area ( $p=0.014$ ) was the only independent variable that entered the model for torrent salamander densities (Table 6.4.1). The relationship between torrent salamanders and percent pool area was surprising, given that our observations and the specific habitat analysis strongly suggested that torrent salamanders are under-represented in pool habitats relative to jam and riffle habitats. We infer that percent pool area is indicative of the type of channel complexity that creates good habitat for torrent salamanders.

Table 6.4.1. Results of stepwise linear regression of amphibian densities versus reach-scale habitat metrics.

<i>All Amphibians</i>				
Step	Variable	p-value	Standardized Structure Coeff.	Cumulative R <sup>2</sup>
1	Functional LWD/cw	0.003	0.777	0.503
<i>Torrent Salamanders</i>				
Step	Variable	p-value	Standardized Structure Coeff.	Cumulative R <sup>2</sup>
1	% Pool Area	0.014	0.741	0.549

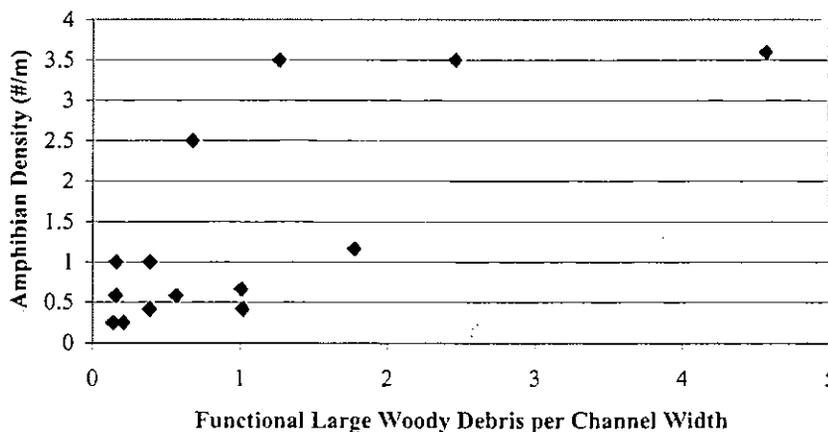


Figure 6.4.2. Aquatic Amphibian Density vs Functional Large Woody Debris per Channel Width

### *Specific Habitat Censuses*

During three seasons of specific habitat surveys, we collected 189 amphibians from 199 specific habitat units surveyed in eleven streams located near our four sites (but not including the streams used in the reach-scale habitat analysis). Tailed frog larvae were found in only 12 of the 199 units, therefore their numbers were eliminated from further analysis. Tailed frogs were only found in riffles, cobble jams, and pools, not in organic jams or bedrock cascades. 133 Torrents were found and were collected from all of the seven habitat types. Forty-three *Dicamptodon* were collected from all the habitat types except bedrock cascades. Table 6.4.2 presents the presence/absence data for giant salamanders and torrent salamanders in each habitat type. Pools featured the lowest percent occupancy by torrent salamanders. Torrents were most likely to be found in organic jams and riffles. *Dicamptodon* showed no particular preference for any habitat, although no *Dicamptodon* were found in bedrock cascades. Again, it is possible that this result is due to generalist behavior by *Dicamptodon* as a genus, or that this is an artifact of merging *D. tenebrosus* and *D. copei* in the statistical analysis. This analysis is predicated on the assumption that *D. tenebrosus* and *D. copei* require and prefer similar habitat. We were only able to survey nine bedrock cascades in the specific habitat streams. Insufficient numbers of channel-adjacent seeps were found to include in the analysis.

As 117 of the 199 habitat units sampled contained no amphibians, ANOVA analysis of amphibian densities across habitats was not possible. We instead considered presence/absence and conducted Chi-square analysis on the habitat data by species. We combined the habitat units of log, wood and mixed jams to make an organic jam unit, and found that torrents are present in these more than in pools ( $p < 0.05$ ,  $df = 1$ ) and cobble jams ( $p < 0.05$ ,  $df = 1$ ). Torrent salamanders are also present in riffles more than in pools ( $p = 0.052$ ,  $df = 1$ ). This analysis indicates that torrent salamanders spend a relatively small proportion of time in pool habitat and that torrent salamanders prefer jams with organic material to fully inorganic jams. We found no differences in the presence of *Dicamptodon* among habitat types.

Table 6.4.2. Percent occupancy of specific habitat types by torrent and giant salamanders.

#### Torrent Salamanders

Habitat Type	Units Surveyed	Present	Absent	% present
Organic Jams	61	24	37	39.3%
Cobble Jams	48	14	34	29.2%
All Jams	109	38	71	34.9%
Riffles	65	28	37	43.1%
Pools	26	5	21	19.2%
Bedrock Cascades	9	3	6	33.3%

#### Giant Salamanders

Habitat Type	Units Surveyed	Present	Absent	% present
Organic Jams	61	11	50	18.0%
Cobble Jams	48	6	42	12.5%
All Jams	109	17	92	15.6%
Riffles	65	15	54	23.1%
Pools	26	3	23	11.5%
Bedrock Cascades	9	0	9	0.0%

#### Upstream-Downstream Surveys

We collected 245 *Rhyacotriton* from 21 of the 38 (55%) streams sampled. Torrents were found in first, second, and third order streams, and were found at three of the four sites (none were found at site 21). In site 12, 11 of 13 streams (85%) contained *Rhyacotriton*. Three of the six site 17 streams (50%), and 63% (7 of 11) of the streams at site 13 contained torrents. The highest torrent density was  $4.85/m^2$ , and was at a site 13 stream. Torrent salamander numbers and densities decrease as order increases ( $P = 0.084$ ,  $H = 4.957$ ,  $df = 2$ ) as shown in Table 4. *Rhyacotriton* densities are highest in streams where the flow and drainage area are low (first order streams), and their densities decrease as these stream variables increase (Figure 6.4.3).

We collected 188 *Dicamptodon* from 33 (87%) of the streams. These salamanders were present in first, second, and third ordered streams and at all four sites. The highest density of giants

was 1.59/m<sup>2</sup> and was in the same stream (13S) that contained the highest torrent density. ANOVA analysis across stream orders showed no relationship between *Dicamptodon* and stream size, as their numbers and densities were similar among all stream orders. The previously stated cautions about pooling *D. tenebrosus* and *D. copei* apply here as well. *Dicamptodon* densities are plotted against drainage area and estimated mean annual flow in Figure 6.4.4.

Over the two field seasons, we found 138 *Ascaphus* within the 38 stream surveys that comprised the Up-Down portion of the project. Of the 32 streams that could potentially harbor tailed frogs, 16 (50%) contained tailed frog larvae. These streams were first, second, and third order, and were distributed across the 12, 13, and 21 sites. *Ascaphus* densities seem to increase with increases with stream order (Table 6.4.3) but this result was not statistically significant. *Ascaphus* densities appear to show a weak positive relationship with stream power and unit stream power (Figure 6.4.5).

Table 6.4.3. Average numbers and densities of *Rhyacotriton* sp., *Dicamptodon* sp., and *Ascaphus truei* in first, second, and third order streams surveyed for the Upstream-Downstream investigation.

Stream Order	Average number of individuals collected in sample reaches			Average number per unit length of stream (#/m)		
	Torrents	Giants	Tailed Frogs	Torrents	Giants	Tailed Frogs
1 <sup>st</sup> n = 18	10.1	4.9	2.4 n = 9	1.046	0.484	0.245 n = 9
2 <sup>nd</sup> n = 10	5.4	4.6	3.6 n = 7	0.52	0.425	0.338 n = 7
3 <sup>rd</sup> n = 10	1	5.3	9.3 n = 8	0.09	0.507	0.926 n = 8

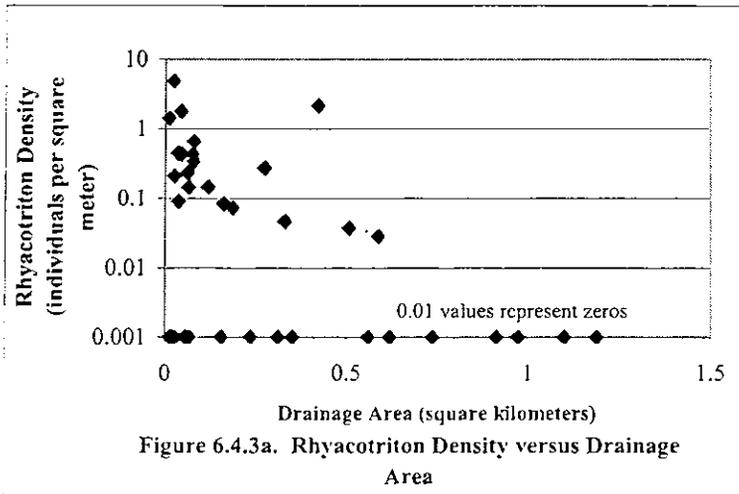


Figure 6.4.3a. Rhyacotriton Density versus Drainage Area

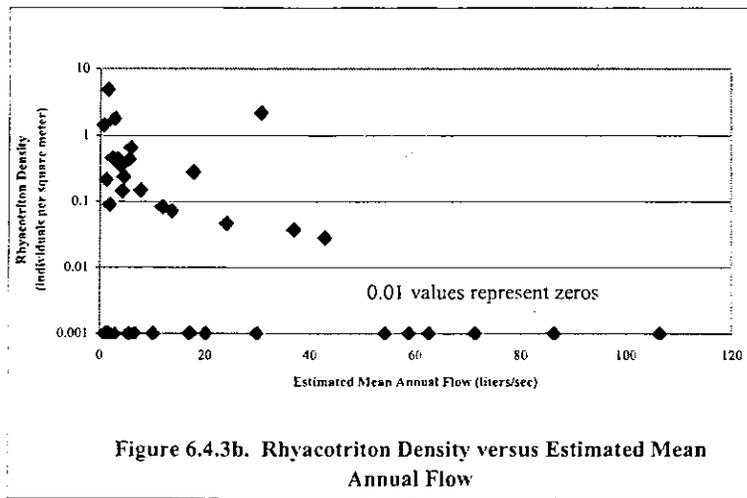


Figure 6.4.3b. Rhyacotriton Density versus Estimated Mean Annual Flow

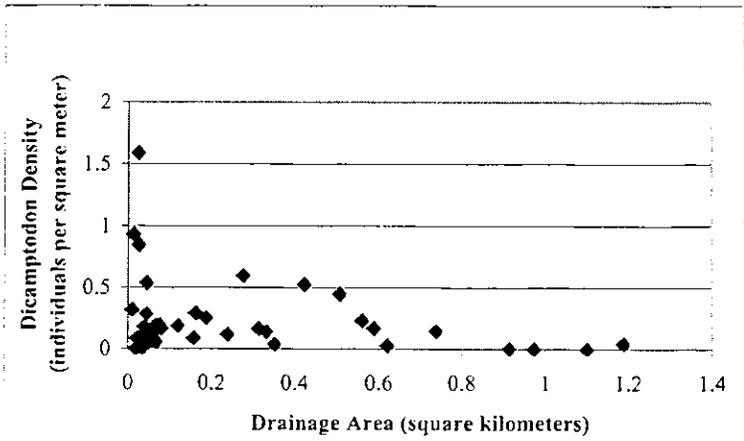


Figure 6.4.4a. Dicamptodon Density versus Drainage Area

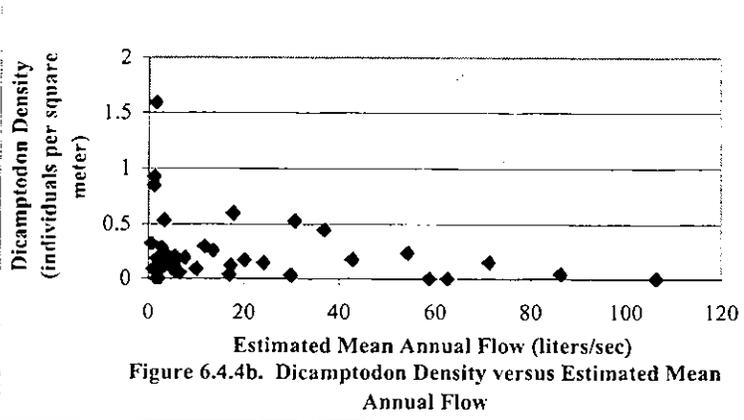


Figure 6.4.4b. Dicamptodon Density versus Estimated Mean Annual Flow

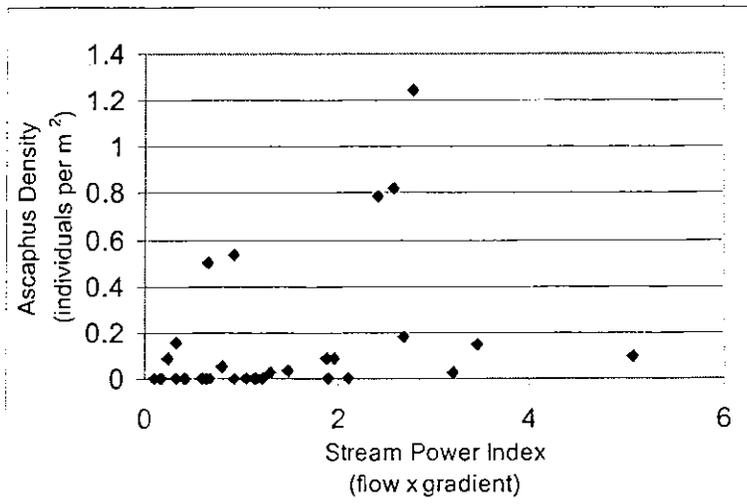
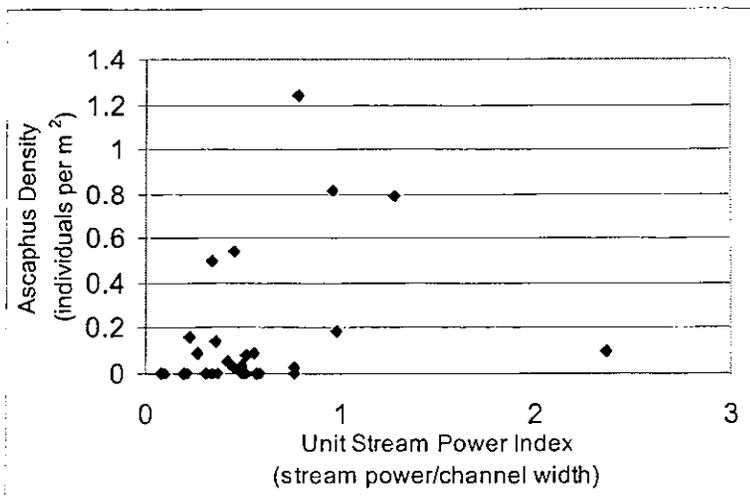


Figure 6.4.5a. Ascaphus density vs stream power index



#### 6.4.2. Headwater Amphibian Responses to Timber Harvest

The high variability in densities between sites (driven by lithotopo differences) and between streams within sites resulted in very low statistical power for the comparisons of amphibian densities prior to and following logging. Furthermore, tailed frogs were present in too few of the study site streams to support any sort of statistical analysis. Therefore, the effects of logging on stream-dwelling amphibians were assessed through graphical methods only.

##### Giant Salamanders (*Dicamptodon*)

*Dicamptodon* disappeared or nearly disappeared in five of six clearcut streams immediately after harvest (Figure 6.4-02). Meanwhile, *Dicamptodon* numbers showed no consistent trends in either the buffered or reference streams. Clearcut harvest either caused significant direct mortality or motivated emigration, presumably (but not necessarily) downstream, or both. Two years following harvest, *Dicamptodon* appeared to be repopulating clearcut streams, although habitat structure in these streams was still dominated by changes wrought by the introduction of logging debris.

Although the reductions in *Dicamptodon* immediately following harvest in clearcut streams appear consistent and dramatic, statistical analysis revealed no significant differences in treatment effects because of the large variability in abundance of these creatures (Cross, 2001). Alternative study designs that reduce the pre-harvest variability or that survey a large number of streams would be necessary to discern treatment effects statistically. Possible alternative study designs are presented in the discussion section.

While *Dicamptodon* appear very sensitive to the immediate effects of clearcut harvest on small headwater streams, long-term viability of populations in these streams may not be a problem due to the presence of *Dicamptodon* in the buffered fish-bearing streams into which these headwater streams drain. Longitudinal surveys of amphibian populations revealed that *Dicamptodon* are present in significant numbers in larger fish-bearing streams up to ten meters wide (Cross 2001). Other researchers have found *Dicamptodon* while conducting snorkel surveys in even larger streams (Murphy and Hall 1981; Hawkins et al. 1983; Parker 1991).

##### Torrent salamanders (*Rhyacotriton*)

*Rhyacotriton* showed no response to clearcutting relative to the other treatments (Figure 6.4-03). *Rhyacotriton* populations varied significantly from year to year in most streams, but population trends appeared random in clearcut, buffered, and reference streams. Numbers in one of the clearcut streams decreased substantially, but a large number of young-of-year were present in this stream prior to harvest, so its pre-harvest numbers were affected by a large torrent salamander hatch. Torrent salamander numbers increased in the other clearcut streams at sites 12 and 13. At site 17, torrents had always been present in only low numbers, following logging, and torrents disappeared in the clearcut stream in the second year after logging.

Fig. 6.4-01. Torrent Densities Pre/Post Treatment by Site

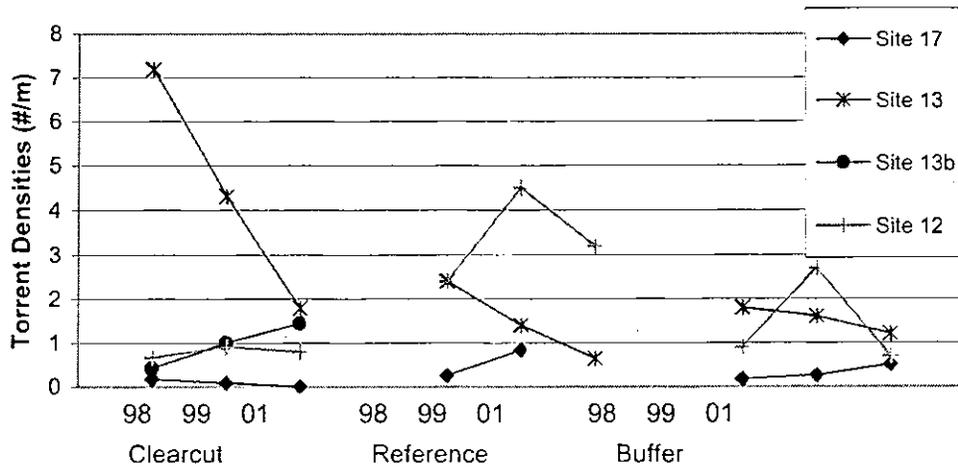


Fig. 6.4-02. Dicamptodon Densities Pre/Post Treatment by Site

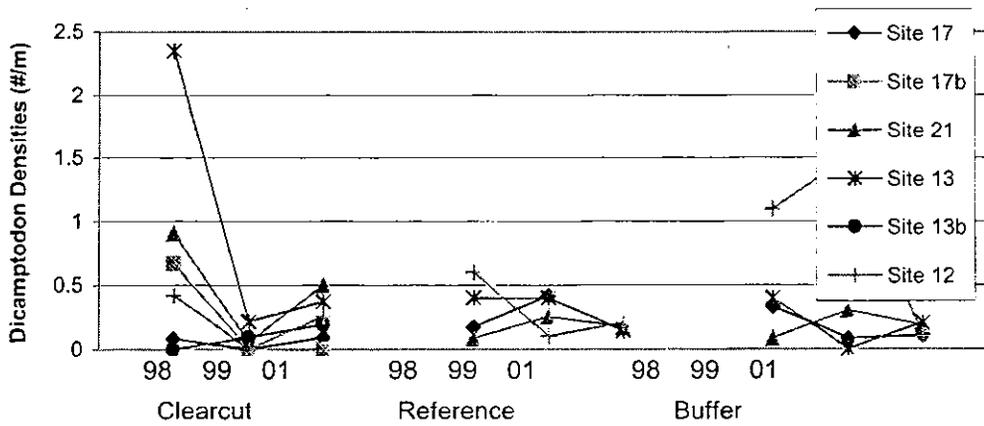
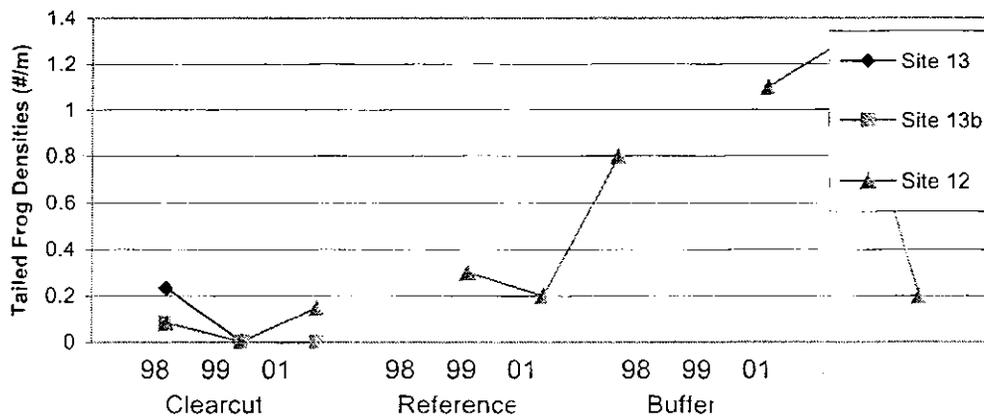


Fig. 6.4-03. Tailed Frog Densities Pre/Post Treatment by Site



### *Pacific tailed frogs (Ascaphus truei)*

Pacific tailed frogs were present in only 5 of the 15 streams monitored including three clearcut streams, one buffered stream, and one reference stream. Immediately after harvest, tailed frogs were undetected in all three clearcut streams. Two years following harvest, tailed frogs reappeared in one of the clearcut streams. In the buffered stream and reference stream, numbers were highly variable from year to year, but presence was maintained in all years.

Within their ranges, tailed frog distributions are much more patchy than *Dicamptodon* or *Rhyacotriton*. Studies of tailed frog response to timber harvest should pre-screen streams for tailed-frog presence. Although this study suffers from low numbers of streams and low numbers within streams, the results suggest that Pacific tailed frogs are sensitive to clearcut harvest on small headwater streams. Given the dependence of tailed frog larvae on coarse bed sediment for feeding and given the fining of the clearcut streams due to slash introduction, it is not surprising that tailed frogs fare poorly in unbuffered headwater streams.

It is not clear, however, that headwater streams are a favored habitat for *Ascaphus*. Pacific Northwest headwater streams with mature forest canopies have low primary productivity due to low light levels, and this potentially limits the availability of diatoms on which tailed frogs feed. Our limited upstream-downstream surveys for tailed frogs suggest they are more abundant in larger streams.

### *Other amphibians*

In the pre-harvest surveys, species captured included western redback salamanders, northern red-legged frogs, roughskin newts, and Van Dyke's salamanders. Western redbacks were ubiquitous and found on every site. The locations of Van Dyke's salamander captures were mostly consistent with the Wilkins and Peterson's general description of Van Dyke's habitat (Wilkins and Peterson, 1999). Van Dyke's were found in mossy stream-adjacent habitat along streams with coarse substrate in basalt geology in the Willapa Hills. Unlike Wilkins and Peterson's captures, however, some Van Dyke's were found on south-facing slopes.

In the post-harvest clearcut streams, searching streamside habitat was not possible due to the accumulation of logging debris. However, changes in streamside habitat can be described. After harvest, mossy and shady stream-adjacent habitat ceased to exist along the clearcut streams. Incidental captures of riparian-associated amphibians were rare in the post-harvest clearcut streams and consisted almost exclusively of western redbacks salamanders.

## 7. DISCUSSION

### *Descriptive Headwater Geomorphology*

The low fluvial power of these streams allows organic debris and relatively small LWD to play important roles in creating steps and affecting the morphology and habitat. If fluvial transport of LWD were an important part of wood mechanics in these streams, the size distribution of functional wood should be skewed to the right of the distribution for non-functional wood, but that

is not the case. In fact, the distributions suggest that wood between 10 and 20 cm diameter is more likely to function than is larger wood. Organic debris dams, which lack any LWD, comprised 17percent of the steps in these streams. The low fluvial power also means that bank erosion is an inconsequential component of wood recruitment. We hypothesize that the major sources of wood to these streams are chronic mortality, limb senescence, and windthrow.

While obviously important, the role of big and very large woody debris in small stream habitat creation may have been overstated in recent decades, partly because LWD is one of the few controlling geomorphic variables that land managers can manipulate. Consequently, the role of other landscape variables and the natural variability in habitat quality may have been undervalued. The morphology and behavior of any channel is affected by eight inter-related major factors: 1) climate, 2) soils and geology, 3) topography (and network topology), 4) upland and riparian vegetation, 5) sediment loading, 6) flows, 7) LWD loading, and 8) time since disturbance. Since LWD loading can be partly controlled by riparian silvicultural practices and since previous studies have repeatedly demonstrated the beneficial role of wood in channels, a push has occurred toward a philosophy that all streams need big wood to provide good habitat. There are several dangers associated with this trend. It fuels a perception that all streams should support good habitat and that wood will cure any stream with poor habitat. This discounts the importance of the other dominant landscape variables that control habitat characteristics in a stream. Until recently, very little data exists on streams < 4 m width, and it is clear that large stream wood relationships should not be extrapolated downward to small streams. Other researchers (e.g. May and Greswell, 2001) have demonstrated that very large woody debris is important in trapping sediment and rebuilding valley floors after debris flows.

Many stream researchers describe their study streams as small, medium, or large, but no standard definitions exist of what these terms mean. In this data set, a two-component definition of small streams as first- or second-order and as having channel widths < 4 m was not incompatible, but no reason exists that there would be similar correspondence between stream width and stream order in other physiographic regions. Using channel width as a descriptor of stream size can be problematic because two streams with identical flows might have different average widths due to differences in gradient or LWD loading. Using an estimate of mean annual flow might be a better way to classify stream size, but it is sometimes difficult to get good climatic data with which to estimate flow. Furthermore, in landscapes with significant groundwater fracture flow, little relationship exists between basin area and flow. In the regression analysis conducted here, channel width often entered the channel morphology regressions instead of other metrics of channel size, so channel width may have advantages over inaccurate estimates of mean annual flow and power in basins of this size.

Forward-stepwise multiple linear regression is a problematic methodology for evaluating geomorphic processes in streams. Important relationships are likely to be non-linear, but the number of possible independent variables and the large amount of covariance make it difficult to pre-screen and linearize the independent variables. Stream power and unit power are useful variable combinations with theoretical justifications, but it is likely that other useful variable combinations exist which could simplify stream assessments. Several important "variables", including geology, disturbance history, and management history are difficult or impossible to quantify in a regression analysis. The results of the regressions developed in this study are not

meant to serve as predictive equations, but instead were used to elucidate the relative importance of the many geomorphically important variables.

### *Headwater stream macroinvertebrate communities*

Headwater streams from Washington's coastal mountains share some similarities with 1<sup>st</sup>- order streams elsewhere. Shredders dominated the 1<sup>st</sup>- order study streams in terms of both abundance and biomass, but their relative importance decreased in downstream reaches. Conversely, scraper numbers and biomass were negligible in 1<sup>st</sup>- order reaches and increased downstream. These shifts in assemblage structure correspond closely with the predictions of the River Continuum Concept (Vannote et al. 1980), which indicates that energy resources in 1<sup>st</sup>- and 2<sup>nd</sup>- order reaches are largely detrital based (Grubaugh et al. 1997) and mid-order reaches (here, 3<sup>rd</sup>- order streams) move towards an autotrophic system (i.e., scraper and collector dominated).

While invertebrate assemblages in headwater streams of Washington's coastal mountains are functionally similar to streams elsewhere, the study streams differed in terms of taxonomic richness, abundance, and biomass when compared to headwater streams of other regions. The 1<sup>st</sup>- order streams in this study supported 39 taxa, whereas researchers in the Appalachian and Cascade mountain ranges of the U.S.A. have reported 55 (Grubaugh et al. 1997) and 58 (Hawkins et al. 1982) taxa, respectively. Furthermore, macroinvertebrate densities were much lower in coastal Washington than elsewhere. Densities in this study's streams averaged 134 individuals m<sup>-2</sup> as compared to densities of 2000 individuals m<sup>-2</sup> (Murphy et al. 1981) to 5750 individuals m<sup>-2</sup> (Hawkins et al. 1982) in streams of the Oregon Cascades, and 62,000 to 110,083 individuals m<sup>-2</sup> in Appalachian streams (Wallace et al. 1999). Our estimates accounted for individuals lost from sieving samples in the field, so differences in densities between coastal Washington and elsewhere are not simply a sampling artifact. The accuracy of our estimates was further supported by kick-net samplings from each stream, which collected very similar assemblages (Haggerty 2000).

Biomass estimates without crayfish (167.27 mg AFDM m<sup>-2</sup>) were also low in this study's 1<sup>st</sup>- order streams. In comparison, in Cascade streams Hawkins et al. (1982) reported 7 to 2287 mg AFDM m<sup>-2</sup> and Wallace et al. (1999) reported 1400 to 2200 mg AFDM m<sup>-2</sup> in Appalachian streams. When biomass calculations included crayfish, our estimates were considerably higher (2037.10 mg AFDM m<sup>-2</sup>), as crayfish comprised 93% of the sample biomass. Most studies elsewhere either do not include crayfish in biomass calculations or crayfish are absent. In headwater Appalachian streams, Wallace et al. (1999) reported crayfish biomass estimates of 35 to 2671 mg AFDM m<sup>-2</sup> with densities ranging from 1 to 8 crayfish m<sup>-2</sup>. When crayfish are included, biomass estimates in coastal Washington streams are more similar to levels elsewhere.

A combination of bottom-up and top-down factors probably explains the low taxonomic richness, density, and non-crayfish biomass of macroinvertebrates that we observed. Bisson & Bilby (1998) found that streams in the nearby coastal rainforests of Washington have low primary production. The almost complete canopy closure of our study streams undoubtedly limited primary production there. A lack of algae requires invertebrates to feed on other food resources, such as wood or detritus that falls into streams. Monoculture coniferous forests bordered most of the study streams, consequently the allochthonous inputs of detritus consisted mostly of needles, which are a very poor quality food resource (Friberg & Jacobsen 1994). In other regions, deciduous trees are

more common and the litter inputs are higher quality. Whiles & Wallace (1997) found that macroinvertebrate biomass in a pine forest stream was 54% to 74% lower than in a nearby hardwood forest stream. The hemlock forests surrounding our sites might explain the lack of macroinvertebrates that we observed.

Predation might be another reason for reduced densities of macroinvertebrates in coastal Washington streams. These habitats support high densities of predatory salamanders and crayfish. Salamander densities were as high as 10 individuals  $m^{-2}$  in some reaches, and averaged 2 individuals  $m^{-2}$  (Cross 2001). Crayfish densities averaged 1 individual  $m^{-2}$  and ranged from 0 to 4 individuals  $m^{-2}$ . Bury & Martin (1967) and Metter (1963) found that stream salamanders consume Ephemeroptera, Plecoptera, Diptera larvae, and amphipods. Crayfish can be important predators in some streams (Parkyn et al. 1997) and the *Pacifastacus* crayfish at the study sites are primarily predaceous (Hershey & Lamberti 1998, Nystrom & Abjornsson 2000). *Pacifastacus*, is noted for aggressive behavior (Tierney et al. 2000) and we observed these crayfish attacking larval salamanders and each other. The cambarid crayfish found in many other 1<sup>st</sup>- order streams are more omnivorous than *Pacifastacus* (Huryn & Wallace 1987) and differences in crayfish function may have contributed to the overall differences between macroinvertebrate assemblages in coastal Washington streams and those elsewhere.

Our experiments with wood support findings elsewhere that suggest wood to be an important resource for headwater macroinvertebrates. In-stream wood affects macroinvertebrates by creating habitat stability (Swanson & Lienkaemper 1978), retaining detritus (Benke & Wallace 1990), increasing surficial habitats (Minore & Weatherly 1994), providing refugia (Keim et al. 2000), and acting as a direct food resource (Anderson et al. 1978). It appears likely that the macroinvertebrate responses to manipulations of wood volume in this study were food related. The response of macroinvertebrates to wood was particularly pronounced for shredders (Figure 7). Larval elmid beetles, many of which are noted xylophagous invertebrates (Anderson et al. 1978), comprised most of the shredder biomass. Macroinvertebrate detritivores unable to directly consume the wood might have benefited from the epixylic biofilms. For example, *Juga* snails will consume the biofilms on wood when other resources are unavailable (Anderson et al. 1978). The fact that shredders still dominated the cobble-only baskets indicates that other sources of organic matter were available besides wood. We found that over the 6-week study, an average of 77 g basket<sup>-1</sup> of hemlock needles and other non-wood detritus accumulated in baskets. Many taxa might be facultative in their food requirements (Friberg & Jacobsen 1994). *Gammarus* amphipods in particular have plastic feeding habits, with feeding behavior changing over their life cycles and with the availability of different food sources (Friberg & Jacobsen 1994, MacNeil et al. 1997). However, whereas wood was obviously important to macroinvertebrates, the increased amount of wood in the wood-only vs. mixed baskets did not elicit a stronger response.

Aspects of wood other than as a food resource might have also influenced macroinvertebrate patterns. In terms of surface area, the baskets with wood had approximately 50% (mixed baskets) or 100% (wood-only baskets) more colonizable surface area than cobble-only baskets. Further, the complex micro-texture of decaying wood provides additional habitat for invertebrates (O'Connor 1991), and these microhabitats probably provide refugia for invertebrates. With the prominence of predators in the study streams, many invertebrates probably live in the wood to avoid predation. In fact, we observed many taxa inside wood or in crevices where they are relatively inaccessible to

predators. Macroinvertebrate response was probably not due solely to variation of surface area or food levels because the mixed and wood-only baskets supported similar assemblages even though wood-only baskets had the most food and surface area. The heterogeneous character of the mixed baskets might explain the high diversity and large biomass associated with those samples. Variation in particle size and surface texture has been directly correlated to increases macroinvertebrate diversity, abundance, and biomass.

#### *Responses of macroinvertebrates to timber harvest*

Our results indicate that logging has significant initial impacts on overall macroinvertebrate densities, collector densities, shredder abundance and biomass, and organic and inorganic matter accretion. However, 1 yr following harvest these responses were no longer detected. After 2 yrs, some differences were again evident but responses were different in each watershed. Where longer-term responses to harvest were detected, they generally involved increases of specific taxa in harvested streams compared to reference conditions.

Several possibilities exist for the initial increase in macroinvertebrate densities in clear-cut streams as compared to reference streams, including wood addition and increased amounts of organic and inorganic matter accretion in 1999. In the Coastal Mountains of Washington, wood appears to be an important resource to stream invertebrates (see above). Many stream invertebrates of the Pacific Northwest are associated with woody debris (Anderson et al. 1978, Pereira et al. 1982), and 56% of the taxa collected in the study streams have been described as being xylophagous or wood associated (Anderson et al. 1978, Pereira et al. 1982).

The most dramatic initial response in terms of the FFG assemblage occurred in collector abundance, where densities were highest in clear-cut streams immediately after logging (Fig. 3A). This response was largely due to chironomids. Chironomids may respond to both increased organic and inorganic matter accretion as the former serves as food and the latter as habitat (Newbold et al. 1980, Brown et al. 1997). Chironomids have been shown to have higher densities in sediment-laden areas, including pools (Luedtke et al. 1976, Hilderbrand et al. 1997). However, in the study streams, collector abundance was not directly correlated with either the mass of organic or inorganic matter (Haggerty 2000). Collector biomass did not parallel collector abundance results, presumably because midges did not contribute much to the overall biomass of that FFG group. The increase in shredder abundance and biomass in clear-cut and buffered streams immediately after logging probably developed in response to the large influx of woody debris to the stream channels. While the abundance response was transient (disappearing by 2000), shredder biomass remained elevated in clear-cut and buffered streams in 2000. The quantity and quality of food available to shredders after logging may help explain the presence of larger-bodied individuals.

The longer-term second year responses to harvest differed from initial responses. In Watersheds 12 and 13, where either fly larvae, EPT organisms, and/or amphipods increased post-harvest, the responses were probably food related. Wood and FPOM levels are probably higher in those clearcut streams, which could benefit macroinvertebrates that shred or collect detritus. Increased insolation from lost canopy can increase algal primary production and increase food supplies of algivores. The development of a fauna dominated by oligochaete worms in the Watershed 17 clearcut stream was the response of most concern. Of all of the watersheds, Watershed 17 was most affected by increased sedimentation, and thus a positive response by

sediment dwelling worms might be expected. The lack of sensitive EPT organisms in the Watershed 17 clearcut stream is a concern, although that watershed never supported large numbers of those invertebrates.

#### *Comparison to macroinvertebrate impacts in other stream systems*

Macroinvertebrate assemblages in headwater streams of coastal Washington were in many ways similar to headwater streams elsewhere. The study streams were dominated by shredders and collectors, presumably because of the large amounts of woody debris and associated organic material available to them (Triska and Cromack 1980). In addition, some of our results paralleled impacts of logging observed in other streams. Due to road construction, yarding, and soil compaction after logging, an increase in sediment yield can develop in affected headwater streams (Brown and Krygier 1971, Corner et al. 1996). We observed an initial increase in inorganic matter (sediment) accretion in the clear-cut streams immediately after logging. Moreover, reduction in particle size in the clear-cut streams was observed, presumably due to trapping of fine sediments by the excess organic debris.

However, some of the initial harvest impacts observed in our study streams were very different than those observed elsewhere. Brosofske et al. (1997) suggested that logging practices altered microclimates around small streams in western Washington by increasing stream temperature. After removal of the canopy cover from adjacent forests, solar insolation of the stream should increase, as should heat exchange due to the loss of the windscreen. We, however, did not observe a post-harvest increase in stream temperature. Stream temperatures were similar in clear-cut, buffered, and reference streams, presumably because groundwater discharge is the main source of flow or because slash protected the channel from solar radiation or acted as thermal insulation in the logged streams.

Another commonly reported effect of logging is a shift in the macroinvertebrate assemblage towards one that is scraper-based. Riparian vegetation can influence algal production in streams through light attenuation (Gregory et al. 1991). Murphy and Hall (1981) and Duncan and Brusven (1985) suggested that the typical shift after clear-cut logging is from a primarily allochthonous based system to an autochthonous based system, through increased periphyton growth. This periphyton growth increases scraper densities and biomass (Murphy et al. 1981, Silsbee and Larson 1983, Hachmöller et al. 1991). However, with the large inputs of woody debris after logging, we initially did not see the typical shifts in assemblage structure seen elsewhere. Scrapers did not respond to logging and were, in fact, a very small portion of the benthos. Even in baskets open to sunlight, clear-cut logging did not induce a scraper response. Instead, we found that collectors and shredders, rather than scrapers, were the groups most affected by logging. The subsequent responses in 2001 by EPT taxa and amphipods in some streams might, however, result from increased insolation and algal growth as slash cover began to diminish.

Murphy and Hall (1981) suggest that logging reduces input of large woody debris to stream systems. Rather than a decrease in wood loading, we observed a dramatic increase in large and small woody debris in logged streams. The process of tree felling and the subsequent limbing and topping of trees in or near stream channels causes the streams to be clogged with slash. Harvest debris left behind accumulated in stream channels, often burying the streams under slash. However,

these conditions will change over the coming years as logging debris decays or is mobilized during high flows.

#### *Efficacy of buffer strips in reducing impacts on macroinvertebrates*

Trees along buffered streams acted as a fence to keep large organic debris (limbs and tree trunks) out of the streams and helped maintain channel morphology. However, they did not reduce inputs of fine sediments and these sediments are of special concern of streams in western U.S.A. and Canada (Corner et al. 1996). Immediately after timber harvest, organic and inorganic matter accretion was greater in both clear-cut and buffered areas than in non-logged streams. Additionally, percentage of fine sediments increased in a few of the buffered streams, probably due to runoff from logging roads and landings. Buffers also did not effectively prevent shifts in macroinvertebrate compositions seen in clear-cut streams. These results suggest that the buffer-strips employed on study streams, which were extremely variable in width (ranging from 2.5 to 21 m), were not immediately effective in lessening some key impacts of harvest.

The effectiveness of buffer-strips at protecting water quality is sometimes dependent on their width and position in catchment, but more often on site-specific factors such as slope, soil type, and tree species (Belt and O'Laughlin 1994). Most of the trees in our buffers were lost to wind throw within 1 yr after logging. Subsequent blow-down may yet increase the amount of sediment and organic material in the form of needles, twigs, and limbs entering the streams through bank failure thereby altering stream geomorphology. Recent changes in Washington's best management practices (BMPs) have placed more emphasis on buffer-strip design and function, and better designed buffers may yet protect the biotic integrity of 1<sup>st</sup> - order streams of coastal Washington.

#### *Headwater Amphibians*

Although the four sites used for these investigations share similar management histories, canopy closure and composition, they vary greatly in geology and topography. Site 12 has very steep side slopes, 13 and 17 have steep side slopes and 21 has low gradient side slopes. The channel gradients of site 12 are also steep, while at the remainder of the sites, gradients are moderate. Site 21 is characterized by highly weathered crescent basalts, site 12 has competent basalt, site 13 has more weathered basalt, and site 17 has mixed unconsolidated alluvial and glacial outwash sediments overlying marine sediments. Since lithotopo units have been identified as a dominant factor influencing amphibian presence, abundance, and richness in the Pacific coastal region (Diller and Wallace 1996, Wilkins and Peterson 2000), lithology and topography may be driving the inter-site variability in amphibian populations that we recorded.

#### *Torrent Salamanders*

Despite this inter-site variability, torrents show several relationships to the habitat variables we considered. Torrent salamanders were absent at site 21 supporting the theories that these salamanders are very sensitive to geology and topography and are positively associated with higher stream gradients (Wilkins and Peterson 2000) and that torrents almost exclusively inhabit streams in consolidated geologies (Diller and Wallace 1996).

As indicated in the upstream-downstream investigation, torrent salamanders predominantly used small streams, as their numbers drastically decreased with increased stream size. In addition, their densities increase in streams with low flow, reduced drainage area, and low unit stream power index and unit stream power; conditions associated with smaller headwater streams.

Torrent salamanders were collected from all seven of the in-stream habitat types surveyed in the specific habitat investigation, suggesting that within suitable small streams, torrents utilize all available habitats. However, presence of torrents in these habitats was not uniform. Torrents were least likely to be present in pools and most likely to be present in organic jams and riffles.

In the reach-scale habitat investigation, high values for percent pool area predict higher densities of torrent salamanders. This is contrary to Welsh and Lind (1996) and Welsh and Ollivier (1998) who determined that torrents are negatively associated with scour pools and non-step pools respectively. However streams in these studies were larger and habitat unit designations were made on a larger scale than in our headwater streams. Because the pools in our streams are formed by obstructions in the flow, the percentage of pools in these streams is a reflection of the percentage of steps or jams in the streams. Therefore the pool association determined in the reach-scale habitat investigation is not contrary to the indication of the specific habitat investigation that torrents prefer organic debris jams and riffles to pools. The percentage of pools in the streams may simply reflect the type of habitat complexity preferred by torrents. This, in fact, corroborates a finding of Welsh and Olivier (1998) who noted that torrents are positively associated with pools that are in association with step habitat.

### *Giant Salamanders*

Because giants were found in relatively consistent numbers among sites, and because none of the variables served as good predictors of density, it appears that *Dicamptodon* as a genus are insensitive to the degree of variation in geology, topography, and physical stream characteristics represented in our study streams. However, our analysis pools *D. tenebrosus* and *D. copei* together, possibly obfuscating habitat relationships specific to each species. Similarly, in the specific habitat investigations, *Dicamptodons* were found in relatively equal proportions in six of the seven specific habitat types surveyed. Further, no strong trends for densities of *Dicamptodon* were found for stream order comparisons in the Up-Down investigation. Before generalizing about *Dicamptodon* habitat relationships, studies of the individual *Dicamptodon* species should be conducted.

The apparent lack of *Dicamptodon tenebrosus* specificity is not surprising as they have higher tolerances for variations in temperature, substrate, and elevation than torrents or tailed frogs. *D. copei*, however, may be more restricted in water temperature requirements. In fact most studies of giants similarly have found only the following habitat requirements for *Dicamptodon tenebrosus*: higher gradient streams (Corn and Bury 1989, Hawkins et al. 1983, Kelsey 1995, Murphy and Hall 1981, Murphy et al. 1981), rock/cobble substrates, with little sand and embeddedness (Corn and Bury 1989, Hawkins et al. 1983, Murphy and Hall 1981, Murphy et al. 1981, Parker 1991, Welsh and Ollivier 1998).

## *Pacific Tailed Frogs*

In the upstream-downstream investigation, tailed frog densities increased with stream order, perhaps explaining why few were detected in the reach-scale and specific habitat investigations (both of which considered only 1<sup>st</sup> order streams). The upstream-downstream investigation also suggested a positive relationship between *Ascaphus* density and stream power and unit stream power. Since these two variables help determine the presence of large gravel/small cobble habitat typically preferred by tailed frogs, we expected them to have a positive relationship to *Ascaphus* density. The low numbers of Pacific tailed-frogs in first- and second-order forested coastal streams may also result from the high canopy coverage and low light penetration over these streams. Larval *Ascaphus* are scrapers, feeding on diatom films on cobble and large gravel substrate. Food resources for larval *Ascaphus* are therefore limited by stream primary productivity and light. The macroinvertebrate investigations suggest that primary productivity in these streams is very low. This may be compounded in this study by the dominance of western hemlock as the overstory species in many of the study streams. Western Hemlock detritus breaks down slowly, and is a relatively poor food source for headwater streams.

As headwater streams are only beginning to receive attention as targets for research on riparian species, we hope our investigations will serve as models for much-needed studies concerning aquatic amphibian habitat use in very small streams of the Pacific Northwest. From the literature it is apparent that specifics concerning stream size should be described in future studies of headwater streams. Typical past studies give general descriptions of streams as small, medium, or large or even as 1<sup>st</sup>, 2<sup>nd</sup>, or 3<sup>rd</sup> order, yet these do not provide adequate detail to predict amphibian trends. Though in our study stream size could be inferred from stream order, we caution that in other geographic regions and in different topographies researchers may find very different size-order associations. Because our upstream-downstream investigation revealed correlations between amphibian densities and stream size/order, researchers should note both these metrics before predicting amphibian assemblages.

The specific habitat investigation is an effective way for field crews to characterize habitat use. Our methodology worked well for our <3m stream widths where flow velocities and water depths were low. In larger, deeper streams, researchers may need to make modifications to our protocols and habitat unit designations. Basing our habitat designations on our observations of specific habitat units, we were able to focus our surveys on very specific portions of the stream and collect precise data in this investigation without the disturbance associated with the bank-to-bank searches typical of standard aquatic amphibian sampling. We believe that when considering habitat requirements of these amphibians, similar protocols involving specific habitat searches should be used so that details of habitat preferences will not be overlooked.

## *Responses of Headwater Amphibians to Harvest*

Although the mechanisms for the apparent declines in *Ascaphus* and *Dicamptodon* densities that we observed are not known, decreases may reflect immediate emigration or direct mortality as a result of mechanical destruction (death caused during scour of streams by cable yarding or as large timber debris rolled into the streams). Although direct mortality has not been documented,

Kelsey (1995) found a higher number of injured individuals in harvested streams than in streams at unharvested sites.

Emigration of stream-dwelling amphibians has not been studied. Although studies of the dispersal capabilities of *Ascaphus*, *Dicamptodon*, and *Rhyacotriton* adults exist, the movements of larvae of these species have not been considered. As untransformed individuals, these larvae would be unable to leave harvested streams and move overland to more suitable streams because they are dependent on the aquatic environment for mobility, feeding, and desiccation prevention. The only chance for larval individuals to successfully evacuate uninhabitable harvested stream reaches would be for them to move within the stream to more acceptable reaches of water. This would require that a continuous aquatic environment exist between the areas. Based on our observations of slash accumulation and post-harvest changes to channel morphology, downstream emigration of larvae seems unlikely because post-harvest flow disappears intermittently and larval movement would therefore require travel for several meters in unwetted channels.

Overland dispersal of adult (and metamorphosed) aquatic amphibians to more suitable streams likewise would be unlikely during most summer climatic conditions, as clear-cut slopes quickly become very warm and dry. Overland emigration during cool damp weather cannot be ruled out. Adult tailed frogs are long-lived, and thus larval survival through one harvest event may not be important in maintaining *Ascaphus* populations in small headwater streams. Our sampling methods were not suited for finding adult tailed frogs, so this research sheds no light on adults ability to survive harvest events. Subsurface, within-channel emigration (movement downstream that is shaded by slash) remains a possibility. *Dicamptodon* spp. were ubiquitous in our second-growth headwater streams and in the higher-order streams below. Because the length of coastal headwater streams above harvest-protected fish-bearing streams is short, *Dicamptodon* could potentially recolonize these headwaters as habitat recovers from harvest.

Logging debris also altered the geomorphology of the channels by choking the flow with organic material which trapped the fine sediments that had been set in motion by logging activity. In our study, fine sediments increased from an average of 12% to 44% between pre- and post-treatment surveys (Jackson et al. in press). Fine sediments negatively impact stream dwelling amphibians by 1) filling interstitial spaces, thereby eliminating critical amphibian micro-habitats (Corn and Bury 1989), 2) preventing the adherence of tailed frogs to rocks, thereby preventing their feeding (Nussbaum et al. 1983), 3) impacting critical food resources (Welsh and Ollivier 1998), and 4) clogging the gills of aquatic amphibians, impairing their respiration (Kelsey 1995).

Once the slash covering the survey streams decays and insolation increases, trends similar to previous studies of harvest effects may be observed. If shading is not provided by rapid herbaceous and shrub growth, torrent densities may decrease as a result of stream temperature increase, and *Dicamptodon* densities may increase with increased primary production. However, since our findings about immediate *Dicamptodon* and torrent density changes were opposite those predicted based on previous short-term research, we believe that continued monitoring at these sites is necessary to determine the long-term effects of timber harvest on these amphibians. Since tailed frogs appear to be the species most narrowly distributed and most sensitive to short and intermediate-term timber harvest effects, management of timber surrounding headwaters should focus on protecting habitat for *Ascaphus*.

We also suggest that our qualitative observations about buffering have value for investigators and managers. We found that non-merchantable buffers are ineffective in protecting stream-dwelling amphibian habitat in headwater streams. We were supposed to have three non-merchantable buffers in the study, but because two of the streams lacked non-merchantable timber in the riparian corridor, these streams became clearcut treatments. These previously managed tracts of timber typically contain even-aged stands of trees of similar size. Therefore, virtually all timber on the harvest unit can be harvested, and a non-merchantable buffer is composed of few to no trees.

Full riparian buffers may serve a valuable role as fences preventing logging debris from sliding/rolling into streams. In addition, buffering prevented changes in particle size distributions in the streams following harvest because the "fences" filtered inputs of fine sediments to the streams (Jackson et al. in press). Also maintenance of full buffers prevents skidder trails from impacting channels. When buffers are left around streams, highlead cables can not be positioned across the channels, loggers must position towers to avoid crossing headwater streams with cabled logs.

The results of the amphibian monitoring suggest that tailed frogs should be a focus of research and management with respect to non-fish-bearing streams in Washington State. Furthermore, our study, like most PNW amphibian studies before it, sheds little light on relatively rare riparian-associated salamander species such as Van dyke's salamanders and Dunne's salamanders. The results suggest that within Washington State, *Rhyacotriton* are a hardy species that do not appear affected by timber harvest in most streams. While *Dicamptodon* populations in headwater streams were very sensitive to immediate harvest impacts, the presence of *Dicamptodon* in the nearby fish-bearing streams means there is usually a source of individuals to repopulate the headwater streams after harvest. The streams surveys done two years after harvest suggest that *Dicamptodon* were already beginning to repopulate the clearcut streams.

Pacific tailed frogs are a priority for two reasons. One, within their mapped coastal range they are present in far fewer stream segments than *Dicamptodon* or *Rhyacotriton*. Second, the monitoring data suggest that Pacific tailed frogs are very sensitive to clearcutting along headwater streams, although the rarity of tailed frogs in our study sites supports little confidence in the results with respect to tailed frogs. From physical arguments alone, however, it makes sense that tailed frogs would not do well in the post-clearcut streams. Pacific tailed frog larvae do not have legs and cannot crawl. In a sediment-buried stream segment, Pacific tailed frog larvae will have difficulty finding substrate on which to adhere and they will have difficulty moving to find suitable substrate.. Pacific tailed frog larvae use clean cobble habitat for diatom foraging, and such habitat is rare in the post-clearcut stream condition.

Van Dyke's and Dunne's salamanders should be priority species for two reasons. First, their behavior and/or their rarity have made them difficult species to study. In this and other studies, they have not appeared in high enough numbers to support any sort of statistical analysis. It is not known whether this is due to their rarity or their behavior. More descriptive biological information is needed on these species. Second, the moist, mossy, stream-adjacent habitat these species have been observed to use does not exist in the post-clearcut condition along headwater streams. Whether these species are also present along the buffered fish-bearing streams below is an important research and management question.

The results also stress the importance of defining goals for buffering headwater streams before creating buffer prescriptions. In this study, thin linear buffers on headwater streams performed some, but not all desirable functions. These buffers served as a fence to prevent the introduction of logging debris to channels, and they also maintained channel structure for at least two years. They did not maintain significant overhead canopy coverage, and they did not maintain pre-harvest stream temperatures. They are certain to contribute large amounts of woody debris to the channel system in the future as blown-down trees break up and pieces fall into the channel.

Rather than place thin linear buffers on sections of all headwater streams, the monitoring results suggest it may be worthwhile to explore other buffer strategies. One possible strategy is to only buffer streams with Pacific tailed frogs present, or to buffer one stream with the best habitat out of every five or six headwater streams. Another potentially useful buffer strategy is to extend the buffer from the fish bearing streams a short distance up each non-fish-bearing tributary. Operationally, this would probably be much easier than long linear buffers, and it would serve to protect refuge populations to repopulate each stream as favorable habitat develops over time. Ideally, an HCP or watershed analysis would be conducted to identify the best habitat for Pacific tailed frogs and Van dykes salamanders, and buffers would focus on those streams.

The high variability in amphibian abundance in headwater streams renders it very difficult to achieve statistically significant results even if a treatment effect appears pronounced. Simply increasing the number of blocks in a blocked study design is not a feasible solution because of the limited number of suitable blocks that are harvested in a year. The Principal Investigator for this study screened all of Champion's and Rayonier's planned harvest units for 1998-1999 and could only come up with four suitable units, and one of these was barely suitable. Once logging began, operational concerns and summer drying of a supposed perennial stream ruined the block design anyway. A staggered temporal design might be an effective way to increase the number of blocks. If it is desired to document a statistically significant clearcut effect on *Dicamptodons* and Tailed frogs in headwater streams, other study designs must be evaluated. One potential solution is to screen the study streams based on pre-harvest abundances. In the pre-harvest year, a large number of potential reference and treatment streams would be surveyed for amphibians, but only those within a relatively narrow range of abundances (for instance between 0.5 and 1.0 individuals per meter) would be kept in the study. The obvious drawback to this study design is that the results can only be considered applicable to streams with medium to high amphibian abundance. Another way to increase the statistical power of pre- and post-harvest monitoring is to confine the monitoring and analysis to individual litho-topo units where topographic and geologic controls on habitat quality are relatively uniform (see Wilkins and Peterson, 2000).

It is difficult to conduct repeated amphibian sampling in the clearcut streams because the sampling greatly alters the stream environment, and the perennial sections of these streams are generally fairly short (less than 100 meters). Sampling for amphibians requires clearing 10 – 15 meters of woody debris from the channel, so two years of post-harvest sampling alters the hydraulics and habitat in a significant portion of the stream. Due to the destructive nature of sampling these streams and the relatively slow pace of channel response, it would probably be best to wait at least five years before sampling this set of streams again to monitor the longer term response of habitat conditions and biota.

Additional studies of this type should probably include a botanist in the study team to document changes in the near-channel vegetation over time. This study team did not have the knowledge or expertise to describe and quantify understory and mid-story riparian vegetation. Due to the increase of light on the channel banks in the clearcut and buffered streams, a definite release of near-channel herbaceous vegetation exists in the years following harvest.

Because small headwater streams generally have low fluvial power, it is widely believed that they export large woody debris only during catastrophic debris flow events (e.g. Lancaster et al. 2001; May 2001). The corollary to this belief is that these headwater valleys experience a gradual build-up of valley sediments between debris flow events, but that habitat is relatively static from year to year. While this research did not address the question of static versus dynamic behavior of headwater channels, it is an intriguing area for future research. Jackson and Sturm (in press) have shown that relatively small woody debris is capable of creating habitat complexity in these streams. Wood less than 20 or 30 cm diameter may actually be transportable by headwater streams, in which case some variability in habitat from year to year would be expected. The habitat surveys conducted for the buffered and reference streams strongly suggest that habitat does vary from year to year in headwater streams.

## 8. PUBLICATION PLAN

As listed in the Table of Contents, this project has already produced three MS theses and three journal articles, reprints of which have been transmitted to LWAG and NCASI. Another paper on the responses of macroinvertebrates to timber harvest has been submitted. An article on amphibians has been drafted and is currently in team review. These manuscripts all focus on 1998, 1999, and 2000 data. In addition, the project team is writing a single article describing changes in channel morphology, amphibians, and macroinvertebrates two years after harvest (using 2001 data).

### *Submitted:*

Haggerty, S.M., D.P. Batzer, and C.R. Jackson. In Revision. Macroinvertebrate assemblage responses to logging in coastal headwater streams of Washington, USA. Canadian Journal of Fisheries and Aquatic Sciences. Suggested for resubmission October 2001. Resubmitted April 2003.

### *In team review:*

Cross, S.S. and C.R. Jackson. Stream-dwelling amphibian communities of headwater streams in Washington's Coastal Range: Habitat relationships.

Jackson, C.R., Batzer, D.P., Cross, S.S., and Haggerty, S.M. Headwater stream responses to Logging: morphology, macroinvertebrates, and amphibians

### *Previous Project Reports to NCASI:*

Jackson, C.R. Integrated Headwater Stream Riparian Management Study, Progress Report #4: Project Summary. Submitted to NCASI on March 27, 2000

Jackson, C.R. Integrated Headwater Stream Riparian Management Study, Progress Report #3. Submitted to NCASI on August 30, 1999

Jackson, C.R. Integrated Headwater Stream Riparian Management Study, Progress Report #2: a) Preliminary Results, b) Updates to Summer 1999 Study Plan, and c) Preliminary Stream-dwelling Amphibian Literature Review. Submitted to NCASI on December 29, 1998

Jackson, C.R. Integrated Headwater Stream Riparian Management Study, Research Plan and Progress Report. Submitted to NCASI on August 10, 1998

In addition to these publications and reports, the project team has given a number of conference presentations regarding these studies.

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