Sources, Transport and Fate of Terrestrial Organic Matter Inputs to Small Puget Lowland Streams: Effects of Urbanization, Floods and Salmon

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This is to certify that I have examined this copy of a doctoral dissertation by

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Abstract

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Urbanization can alter various structural and functional properties of aquatic systems including flow, water quality, habitat, energy regimes, and biotic interactions. This study evaluated the effects of urbanization on the delivery and processing of terrestrial organic matter from riparian vegetation along small streams. Changes in the quantity, quality, and time of delivery of organic matter and nutrients as well as organic matter transport and decomposition within small streams with altered physical, chemical, and biological processes due to urbanization were assessed. In many cases, urbanization has changed riparian vegetation from conifer to deciduous trees, altering the amount and timing of organic matter inputs. In some locations, development has led to the modification or removal of riparian trees, reducing total organic matter inputs. Urban streams transport organic matter farther, limiting availability for macroinvertebrate and microbial processing that may otherwise buffer increasing nutrient levels. Finally, red alder leaves decompose faster in urban streams, although the effects of flooding in 2003 and large salmon runs in 2002 were larger than the effect of urbanization. Urbanization alters the inputs, transport, and decomposition of organic matter in small streams, which could affect nutrient delivery to downstream water bodies like Puget Sound.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

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Chapter 1

Litterfall Rates and Nutrient Inputs to Puget Lowland Streams

Introduction

Riparian zones occupy a unique location in the landscape as the transition between terrestrial and aquatic ecosystems (Gregory et al., 1991). Although riparian zones cover relatively small areas, they represent ecological hotspots with enhanced biodiversity that results from strong physical, chemical, and biological gradients between upland conditions and lotic processes (Naiman and Décamps, 1997).

Riparian vegetation performs a variety of valuable ecological functions. Vegetation shades streams, maintaining cooler water temperatures by reducing stream heating due to solar radiation (Chen et al., 1998; Beschta et al., 1987). Large trees provide instream wood that protects banks, creates pools, and traps sediment and organic matter (Bilby and Bisson, 1998), and both live trees and instream wood provide habitat for riparian species (Naiman and Décamps, 1997). In addition, riparian forests play important roles in the cycling of nutrients and buffer the effect of upland activities on stream ecosystems.

Terrestrial allochthonous inputs represent an important nutrient source to small stream ecosystems (Wallace et al., 1997; Triska et al., 1982; Anderson and Sedell, 1979). Organic matter, in the form of leaves, needles, cones, wood, mosses, and other plant material, falls from surrounding vegetation into streams, where biological, chemical, and physical processes break it down (Webster and Benfield, 1986), providing energy to multiple trophic levels within the stream ecosystem. Riparian vegetation type determines the amount, timing, and quality of litterfall reaching streams (Bilby and Bisson, 1992; Gregory et al., 1987; Bray and Gorham, 1964).

Altering the delivery of allochthonous inputs affects food webs, nutrient regimes, and overall system productivity. A long-term study at Coweeta Hydrologic Laboratory established a distinct change in stream biota when litterfall inputs were physically blocked. Detritivore populations, which rely on allochthonous inputs for food resources, declined (Wallace et al., 1997), but higher trophic levels also were strongly affected as exclusion reduced macroinvertebrate predator populations as well. A related study demonstrated that litter exclusion reduced phosphorus and ammonium retention, producing a chemical change due to the loss of benthic organic matter reaction sites (Webster et al., 2000).

Riparian vegetation removal due to forestry or agricultural activities drastically reduces allochthonous inputs and shifts streams to autochthonous production (Oelbermann and Gordon, 2000; Bilby and Bisson, 1992). Many riparian areas in the Pacific Northwest historically supported conifer trees. Following logging many sites were colonized by native deciduous vegetation, which typically occupy these locations for up to a century before shade-tolerant conifers replace the hardwood trees (Franklin and Dyrness, 1973). The successional processes set in motion by the removal of riparian trees alter both the amount and type of allochthonous inputs to streams (Bilby and Bisson, 1992).

Urban development permanently alters the landscape with somewhat different patterns than those occurring in forestry. Development pressures are strong within riparian zones, and these areas offer high economic and aesthetic value, in addition to their ecological value. In the Puget Sound Lowland, urban development has significantly altered forest cover. Tree cover in Seattle has decreased from 40% in 1972 to 18% in 2006 (City of Seattle, 2006). While no studies have systematically quantified changes to regional riparian forests, changes are likely to be similar to or more dramatic within riparian zones.

Historically, riparian and valley bottom forests in the Puget Sound Lowland were dominated by conifer species, as reconstructed from 1873 General Land Office survey notes (Collins and Montgomery, 2002). Sites included in the 1873 surveys were re-examined in 2000, and the study found that conifers were far less prevalent in 2000 than in 1873, when previously conifers represented 61% of the basal area. By 2000, deciduous species represented over 85% of the basal area in riparian zones, a shift attributed to selective logging of conifer species.

Historical dominance by conifers was also shown by pollen analyses in sediment cores from Lake Washington, east of Seattle (Davis, 1973), which demonstrated a shift from hemlock, cedar, and fir to red alder (*Alnus rubra*) before 1920. While very little old-growth forest remains in the Puget Sound Lowland, old-growth communities occur at moderate elevations in the Cascade Mountains. Remnants maintain conifer dominance, primarily Douglas fir (*Pseudotsuga menziesii*) in less-disturbed landforms such as terraces and hillslopes, while red alder dominates active floodplains (Rot et al., 2000).

Urban and suburban land cover has increased as population has increased 40% between 1980 and 2000 in the three western Washington counties included in the study (Washington State Office of Financial Management, 2006a). While urbanization may result in the complete removal of riparian trees as historically occurred in forestry (Bilby and Bisson, 1992) or agricultural activities (Oelbermann and Gordon, 2000), often urban development maintains some riparian forest cover but alters the composition and density. Landscaping practices may favor individual specimen trees or non-native plants selected for aesthetic purposes. Municipal or residential mowing eliminates natural regeneration of native vegetation, suppressing tree growth and altering the stand structure. Highly disturbed areas with degraded growing conditions for native plants may enhance the spread of invasive species. Urban development alters riparian forest structure, which may alter the natural cycle of litterfall inputs by changing the quantity, quality, and timing of those inputs.

Several historical studies conducted in the Pacific Northwest have quantified litterfall production rates for common Pacific Northwest trees, including red alder (Gessel and Turner, 1974; Zavitovski and Newton, 1971) and conifers, such as Douglas fir and western hemlock (*Tsuga heterophylla*) (Murray et al., 2000; Abee and Lavender, 1972). Red alder forests produce 1.5 to 2 g/m^2/d of organic matter, nitrogen inputs of 28 to 39 mg/m^2/d and 0.95 mg/m^2/d of phosphorus (Gessel and Turner, 1974; Zavitovski and Newton, 1971). Conifer forests generally produce ≤1 g/m^2/d of organic matter (Murray et al., 2000; Bilby and Bisson, 1992), although one study found inputs up to 1.6 g/m^2/d (Abee and Lavender, 1972). Nitrogen loads from conifer forests are lower than those from deciduous forests and provide 5 to 8 mg/m^2/d, but phosphorus loads were similar (0.6 to 1.3 mg/m^2/d) (Murray et al., 2000; Abee and Lavender, 1972). One study in a pristine system on the Olympic Peninsula found Douglas fir contributed as much as 55 mg/m^2/d in nitrogen loads (O'Keefe and Naiman, 2006). All studies were conducted at moderate elevations, generally between 500 and 1500 m, and in stands dominated by a specific type of vegetation. Bilby and Bisson (1992) found that clear-cut sites contributed low organic matter loads of 0.05 g/m^2/d. However, no litterfall studies have been conducted at low-elevation Puget Sound Lowland sites, nor have any studies evaluated the effect of urbanization on absolute amounts or relative changes to organic matter contributions to small streams.

The purpose of this study was to determine how urban development affects allochthonous inputs to small streams. Specifically, the study quantified terrestrial organic matter and nutrient loads to small streams in the Puget Sound Lowland from riparian vegetation types representing a range of disturbance regimes.

Methods

Study site selection

Hundreds of small streams flow through the Puget Sound Lowland ecoregion of western Washington. Many of these originate in the bedrock of the Green and Gold Mountains and the glacial till uplands of the Kitsap Peninsula and greater Tacoma area. Streams generally trend north/south as a result of glacial fluting and typically exhibit a pool/riffle morphology.

Study sites (Figure 1-1) were selected using a stratified-random approach based on orthophoto interpretation of riparian vegetation characteristics. Small streams with watershed areas between 4 and 24 km^2 were identified using a 30-m digital elevation model. In each watershed, primary stream centerlines were refined using orthophotos. Polylines were segmented at 100-m intervals (Oregon DEQ, 2001). Only accessible points, defined as those within 1 km of a road or trail, were retained. Each point was coded with the initial riparian vegetation disturbance level, assigned from orthophoto interpretation supplemented by previous qualitative field observations. Four disturbance categories describe riparian vegetation patterns in the study area. Reference vegetation (REF) includes both mature conifer-dominated and mixed riparian forests. While not pristine, these areas represent minimal disturbance. Low-disturbance sites (LOW) include mature, deciduous-dominated vegetation with a few conifers. Medium-disturbance sites (MED) include a range of vegetation characteristics, from very young natural vegetation to residential landscaping to restoration sites with native vegetation plantings. High-disturbance sites (HIGH) generally lack overstory trees, and herbaceous vegetation dominates.

Ten sites were identified within each of four initial disturbance levels using a random-number generator. Property owners were contacted to request access to these locations, and qualitative observations confirmed the orthophoto-based vegetation classification. Quantitative plot characteristics, described below, were used to finalize the plot and site classification.

The final study design included four reference sites, three low-disturbance sites, four medium-disturbance sites, and two high-disturbance sites. The original study design included four in each category, but two high-disturbance sites and one low-disturbance site were discontinued during the study due to vandalism.

Field methods

Plot characteristics

For each of ten litterfall traps placed at each site, 10-m radial plots were used to characterize vegetation. Because traps were immediately adjacent to the wetted stream, the active channel area within the bankfull width was subtracted from the plot area. Similarly, bridges and roadways within the plot area were subtracted, a modification to traditional mensuration techniques necessitated by the developed study area.

Tree species were identified (Pojar and MacKinnon, 1994), and a cloth tape was used to record diameter at breast height (DBH) for all trees greater than 1-cm in diameter. Multiple stems of coppiced trees and shrubby plants were included in the total area for basal area calculations but only as a single stem for stem density calculations.

Each plant was identified as coniferous or deciduous (tree or shrub), and the basal area was determined for each type. Table 1-2 lists the species found in the plots. Plots were assigned an overall type, with conifer (CON) and deciduous (DEC) plots identified as those with >70% of the basal area occupied by either conifer or deciduous trees, and mixed (MIX) if the basal area was <70% conifer or deciduous. Plots with no trees were assigned a type of NONE.

Canopy cover was determined using a densiometer oriented over the center of each litterfall trap and at 10-m intervals along the stream centerline. Percent canopy cover was estimated as the average of four measurements taken along both axes of the rectangular traps or longitudinally and laterally along the channel.

Litterfall samples

Litterfall samples were collected in rectangular baskets (0.2 m^2) lined with fine-mesh netting to ensure the retention of finer materials and minimize leaching losses from sitting in a pool of water at the bottom of the trap. Litterfall collectors measured vertical litterfall only and did not quantify horizontal movement of litter from the adjacent streambanks to the wetted channel due to wind or overland flow. Baskets were placed on either right or left banks as close to the wetted channel as possible and at 10-m intervals measured along the stream centerline. A pebble was added to weight the netting and to indicate whether the trap had been disturbed since the previous collection. Traps with missing pebbles were indicated in field notes and observed for evidence of tampering, such as missing mesh or unusually low organic matter accumulations. Traps that had been tampered with were removed from the litterfall calculations.

Samples were collected monthly between March 2004 and February 2005, and twice monthly during October and November 2004. Dates were assigned as the end of each sampling period. Materials were placed in paper bags and dried at 60°C to constant mass for preservation.

Laboratory methods

Litterfall samples were separated into components, including red alder leaves, bigleaf maple (*Acer macrophyllum*) leaves, other leaves, catekins, cones, needles, red cedar (*Thuja plicata*), moss and lichen, twigs and wood, grass, other components >1 mm, and other components <1 mm. Other leaves were further identified and tallied. Each subsample was re-dried at 60°C to constant mass and weighed to the nearest 0.001 g.

Each component of the samples collected in March, May, and late-October was ashed at 500°C for five hours. Ash-free dry mass (AFDM) was determined by subtracting ash mass from the oven-dry mass. Percent organic matter (%OM) determined for each fraction varied little between sites and traps, except for the OTH>1 and OTH<1 fractions. For all but these fractions, the average %OM determined from the three collections was used to estimate %OM for the remaining collections (Table 1-1). All OTH>1 and OTH<1 fractions were ashed, as were any fractions with visible sediment accumulated on the materials. In addition, the other leaves were ashed by species in July, September, and late November to determine %OM for the most common leaf types. Green materials were not separated from senesced vegetation, given that Murray et al. (2000) found green needles were 3% of the total needle litterfall and the contribution was minor.

Annual organic matter rates were determined for 128 litterfall collection locations, summarized by component, including leaves (LVS), needles and red cedar (NDRC), other plant materials (OTH), and wood (WD). Mean organic matter rates were quantified for each study reach, as well as for the plot type (CON, MIX, DEC, NONE) and study reach disturbance level (REF, LOW, MED, HIGH).

Litterfall samples collected in late November were analyzed for nutrient levels. Component samples were ground using a Wiley mill (no. 40 screen), and nitrogen and carbon levels were determined using a CHN analyzer (Perkin Elmer Model AD-4). Concentrations of phosphorus and other metals were determined using inductively coupled plasma (ICP). Samples were digested in high-grade concentrated nitric acid and stored in acid-washed vials to avoid contamination. A total of 44 component samples were analyzed for carbon and nitrogen and 39 for phosphorus.

Statistical analyses

Single-factor ANOVA was used to test for differences among categories, and the Student-Newman-Keuls (SNK) test was used for multiple comparisons. For all analyses, a significance level of $\alpha \le 0.05$ was used.

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Results

Plot characteristics and riparian vegetation disturbance level

Plot composition, total basal area, and conifer basal area were used to assign plots to one of the four riparian vegetation disturbance levels, which initially had been classified based on qualitative field observations. Reference sites were those with at least five plots dominated by conifers, although some traps were surrounded by mixed forest or deciduous vegetation. Presence or absence of conifer basal area was used to distinguish between low- and medium-disturbance sites. High-disturbance sites contained few or no trees. As a result of applying plot data to these definitions, two sites changed categories, with ANDE transferred from low-disturbance to reference and BURL from reference to low disturbance.

Sites generally exhibit heterogeneous plot characteristics (Figure 1-2), and few sites have the same plot type throughout the site. For example, at LOST, six of the ten traps and plots were dominated by conifers, one was dominated by deciduous trees, and the remaining three were of mixed composition. Conifer-dominated plots were less prevalent but still present at the low-disturbance sites, which had mixed or deciduous plots predominantly. Medium-disturbance sites were defined by the dominance of deciduous vegetation, while high-disturbance sites had few or no trees.

Mean total basal area was higher among the reference (101 m^2/ha) and low-disturbance (73 m^2/ha) sites than the medium-disturbance sites (36 m^2/ha). High-disturbance sites (0.3 m^2/ha) included few or no trees (Figure 1-3). Mean conifer basal area ranged from 50 to 113 m^2/ha in the reference sites and from 25 to 39 m^2/ha in the low-disturbance sites. Differences in total basal area and conifer basal area were generally significant among disturbance levels; differences between conifer basal area for medium- and high-disturbance sites were not significant, however, likely due to the characteristic absence of conifers within the medium- and high-disturbance vegetation sites.

Stem density (Figure 1-4) was highest at the medium-disturbance sites, although plots with high stem density occurred within the reference sites as well, where a young understory was regenerating below the mature overstory. Particularly high stem density occurred at two restoration sites (CLRM and PUGE), levels that likely reflected recent planting strategies rather than natural regeneration. Low stem densities occurred where disturbance was minimal (350/ha at ANDE) or where trees had been removed from the riparian zone (101/ha at HYLE and 0/ha at MEEK). Reference and low-disturbance sites had mean conifer stem densities that ranged from 80 to 560 stems/ha, while medium- and high-disturbance sites had low conifer stem densities (0 to 65 stems/ha). PUGE was the exception, where young conifers had been planted densely at a restoration site (310 stems/ha).

Reference and low-disturbance sites generally had larger mean conifer diameters (0.2 to 0.35 m) and mean tree diameters than the medium- and high-disturbance sites (Figure 1-5), indicative of more mature stands. The two high-disturbance sites and two of the medium-disturbance sites had no conifers at all. One medium-disturbance site had higher mean conifer diameter than overall mean diameter because of one very large red cedar tree within the site.

Canopy cover values determined from densiometer measurements verified that riparian canopy cover was representative of stream canopy cover. Overall canopy cover was high, with 74% of traps with >80% canopy cover, typical for small streams with closed canopies. Riparian plot canopy cover was highly correlated with stream centerline canopy cover (R^2=0.93, p<0.0005) but was 3% higher, a difference within the error range of densiometer measurements. While the difference was statistically significant, it was not ecologically meaningful. Therefore, riparian litterfall traps were judged representative of stream inputs.

Riparian plot canopy cover was lowest where total basal area was zero, but high canopy cover occurred even where total basal area was low (Figure 1-6). Canopy cover was highest among mixed sites (93.2%) and lowest where no trees were present and shrubs produced the only measurable cover (3.2%). Deciduous plots produced a mean canopy cover of 83.2%, significantly lower than mixed sites and conifer sites (89.2%).

Annual litterfall rates by riparian vegetation disturbance level and plot type

Annual litterfall loading rates ranged from 0.0 to 2.5 g/m^2/d in 128 traps distributed over 13 small stream sites that represent the range of riparian vegetation conditions currently present in the Puget Sound Lowland. Conifer and deciduous stands produced similar organic matter inputs at 1.3 g/m^2/d.

Figure 1-7 presents mean annual litterfall rates for each of the study sites, ordered by overall site disturbance level. High within-site variability, indicated by the standard deviation between traps for each site, produced similar mean annual organic matter rates among vegetation disturbance levels, but differences in the dominant component produced differences in nutrient loads.

Mean annual organic matter loading rates for forested sites did not vary significantly by riparian vegetation disturbance level (Figure 1-8a). Means ranged from 0.75 to 1.36 g/m^2/d for each of the forested sites (reference, low-disturbance, and medium-disturbance). High-disturbance sites, with no tree cover, produced significantly lower loads. Total basal area was significantly different among riparian vegetation disturbance categories, but this did not influence litterfall rates.

Annual nutrient loads based on riparian vegetation disturbance level followed similar patterns (Figure 1-8a). Nitrogen loads were significantly lower at the high-disturbance sites and reference sites compared with the low- and medium-disturbance sites, which were not distinguishable. Higher nitrogen loads at the low- and medium-disturbance sites were due to the higher proportion of leaves in the total litterfall, whereas reference sites contributed more needles and high-disturbance sites produced very little organic matter at all. Phosphorus loads were lowest for the high-disturbance sites, and reference sites produced lower phosphorus loads than the medium-disturbance sites. Phosphorus loads from the low-disturbance sites could not be distinguished from phosphorus loads from reference or medium-disturbance sites.

Based on the individual plot type instead of riparian vegetation disturbance level (Figure 1-8b), deciduous plots produced more organic matter than mixed plots, but the contributions from conifer plots were not distinguishable from deciduous or mixed. When the higher nutrient concentrations in leaf components were factored in, however, deciduous sites produced the highest nitrogen and phosphorus loads while those from conifer and mixed plots produced less and could not be distinguished from each other. Several sites with no trees produced 0.5 to 1.0 g/m^2/d of organic matter, almost entirely from episodic clumps of grass from municipal and residential mowing activities. However, organic matter, nitrogen, and phosphorus loads were uniformly lower than for the forested plots, regardless of disturbance level.

Figure 1-9a summarizes the contributions by component (leaves, needles and cedar, wood, and other). Total contributions by leaves and needles were similar between reference vegetation and low-disturbance sites, with the decrease in conifer inputs offset by the increase in leaf litter from reference to low disturbance. At the reference locations, needle inputs contributed 0.40 g/m^2/d, while the low-disturbance sites produced 0.22 g/m^2/d from needles. The medium-disturbance sites produced the highest leaf inputs (0.85 g/m^2/d), more than the sum of needles and leaves from the reference and low-disturbance sites. Inputs of other materials (OTH), including cones, mosses, grass, and reproductive parts, were consistent across all disturbance categories (0.24 to 0.31 g/m^2/d); these represented 94% of the total organic matter inputs at high-disturbance sites.

Based on plot type, total litter inputs were highest for deciduous sites and lowest for mixed forests. Conifer-dominated plots produced the highest needle inputs (0.50 g/m^2/d) and lowest leaf inputs (0.27 g/m^2/d) of the forested riparian plots (Figure 1-9b). The lack of needle inputs in deciduous-dominated plots (0.03 g/m^2/d) was more than offset by an increase in leaf inputs (0.87 g/m^2/d).

Average wood contributions were similar among the plot types at 0.1 g/m^2/d, although wood inputs to individual traps were highly episodic.

Nitrogen loads were significantly higher from deciduous sites (19.3 mg/m^2/d) than for conifer (12.5 mg/m^2/d) or mixed (12.8 mg/m^2/d) sites, which were not significantly different from each other (Figure 1-9b). Other materials produced relatively uniform nitrogen loads of 4 mg/m^2/d, mostly from grass and reproductive parts, across plot types. Likewise, phosphorus loads from deciduous sites (1.27 mg/m^2/d) were higher than those from conifer (0.91 mg/m^2/d) or mixed (0.90 mg/m^2/d) riparian forests. Phosphorus loads from other materials were uniform among vegetation types (0.35 mg/m^2/d). Wood accounted for 0.05 to 0.07 mg/m^2/d phosphorus input at forested sites.

Needle contributions were moderately related to conifer basal area (R^2=0.55) and leaf contributions weakly to deciduous basal area (R^2=0.31); both relationships were significant (Figure 1-10). However, for both components, many plots with low basal area produced high loads and some high-basal area plots produced low inputs. Neither total nor conifer basal area were strong determinants of organic matter loads.

Peak autumn organic matter litterfall rates

Autumn peak litterfall rates followed similar patterns to annual rates, but with higher magnitudes. High-disturbance sites produced significantly lower litterfall inputs than the reference, low-disturbance, and medium-disturbance sites ($p<0.0005$), but autumn peak rates were not significantly different among the forested sites. Reference, low-disturbance, and medium-disturbance sites produced 3.3 to 6.5 g/m^2/d of OM and 2.6 to 5.4 mg/m^2/d of phosphorus (Figure 1-11). However, autumn nitrogen loads from medium-disturbance sites (52 to 84 mg/m^2/d) were significantly higher than those from reference locations (32 to 55 mg/m^2/d).

Discussion

Because most stream reaches were bordered by a mosaic of conifer, mixed, and deciduous plots, there were no statistically significant differences in litterfall rates among riparian vegetation disturbance categories except where plots with no trees produced less litterfall. However, deciduous riparian forests produced significantly higher nitrogen (+54%) and phosphorus (+40%) loads than conifer or mixed forests. While some deciduous stands are a natural part of the Puget Sound Lowland riparian forests, as indicated by historical patterns, deciduous stands currently dominate riparian forests following the selective cutting of long-lived conifer species (Collins and Montgomery, 2002). The conifer-dominated reference sites produced lower nitrogen loads than more-disturbed vegetation but phosphorus loads from reference sites were not statistically different from those of other categories.

Comparing litterfall rates to previous studies

The results reported for this study quantify the relative contributions among litterfall components, plot types, and study site riparian vegetation disturbance level. Litterfall magnitudes generally were within ranges reported in previous studies, summarized in Table 1-3, or differences could be attributed to specific stand characteristics. No other studies have evaluated the effect of urban activities.

Litterfall rates were highly variable, both within and among sites despite the large number of traps used in the study. Overall site means ranged from 0.1 to 1.4 g/m^2/d. Nitrogen loads ranged from 12.5 to 19.3 mg/m^2/d and phosphorus loads from 0.9 to 1.3 mg/m^2/d for conifer, mixed, and deciduous plots.

Organic matter loads from conifer plots in the present study were within the range found in previous studies, which varied from 0.64 g/m^2/d in Bilby and

Bisson (1992) to 1.61 g/m^2/d in Abee and Lavender (1972). Nitrogen loads in Puget Sound Lowland conifer plots (12.5 mg/m^2/d) were higher than reported previously for old-growth forest, which generally ranged from 5.3 to 7.5 mg/m^2/d. Even the conifer plots produced abundant leaf litter nitrogen loads (mean 4.3 mg/m^2/d), without which total nitrogen loads for those plots would have been closer to literature values because needle inputs were similar. Reference locations were not pristine old-growth conifer sites, which are extremely rare in low-elevation riparian zones in the Puget Sound Lowland. Mature red alder and bigleaf maple trees interspersed with Douglas fir, western hemlock, and red cedar were common within the reference sites, and the presence of deciduous trees likely was responsible for the higher values found within conifer sites in this study compared with previous studies. Phosphorus loads from conifer sites, however, were similar to those previously reported for old-growth sites (0.55 to 1.29 mg/m^2/d).

Mean organic matter loads from deciduous sites (1.3 g/m^2/d) were lower than the overall means reported previously for red alder (1.5 to 2.1 g/m^2/d) but were within reported inter-annual and site variability (Murray et al., 2000; Abee and Lavender, 1972). Puget Sound Lowland sites were not entirely red alder, which had the highest leaf nitrogen concentrations of the nine species analyzed. Red alder nitrogen concentrations (1.87%) were similar to concentrations reported by Zavitovski and Newton (1971) but the higher overall organic matter loads in the previous studies produced higher nitrogen loads (22.9 and 39.1 mg/m^2/d) than those found in the deciduous plots in the present study (19.3 mg/m^2/d). Phosphorus loads from Gessel and Turner (1974) (0.46 mg/m^2/d) were lower than in the Puget Sound Lowland sites (1.27 mg/m^2/d) because of the lower phosphorus concentration for red alder leaves (0.030% compared with 0.091%).

Plots with no trees in the current study produced more litterfall than Bilby and Bisson (1992) found in the clear-cut sites, primarily grass clippings following mowing in urban areas. Even sites with no adjacent trees produced measurable terrestrial organic matter inputs.

Both long-term studies that evaluated inter-annual variability in litterfall rates specifically cited extreme low temperatures or harsh conditions as contributing factors to unusual litterfall rates, either high needle litterfall (Dimock, 1958) or low leaf litterfall (Gessel and Turner, 1974). While some low temperatures were recorded within the overall study area, none occurred that would be expected to produce unusual litterfall rates. Therefore, the period of study was representative and not influenced by extreme events.

Two studies quantified seasonally-varying organic matter and nutrient loads. While Abee and Lavender (1972) found total inputs peaked in the winter due to very high wood inputs, needle inputs peaked in the autumn at 0.49 g/m^2/d, similar to rates found for the conifer-dominated sites (0.50 g/m^2/d). Dimock (1958) reported that autumn rates peaked at 1.2 to 2.6 g/m^2/d in young Douglas fir stands, much lower than the peak autumn inputs from conifer-dominated sites in the present study, likely due to the abundance of leaf litter in the Puget Sound Lowland sites.

No strong relationships were found between stand characteristics and litterfall inputs. Zavitovski and Newton (1971) also found litterfall inputs were not a function of stand age, height, or density. However, O'Keefe and Naiman (2006) found a strong relationship between litterfall rates and stand characteristics in coastal riparian forests along a pristine river on the Olympic Peninsula. Litterfall inputs from 15 deciduous and conifer stands ranging in age from eight to 350 years increased to a peak of 2.8 g/m^2/d after 100 years before declining to 1.4 g/m^2/d in an old Douglas fir stand. Nitrogen loads followed a similar pattern, peaking at 82 mg/m^2/d in 100-year-old stands and declining to 55 mg/m^2/d as Douglas fir dominated. Litterfall rates and nitrogen loads were much higher than in the Puget Sound Lowland, possibly due to the optimal growing conditions in the pristine coastal rainforest with no discernible human influences.

This study demonstrated that terrestrial organic matter inputs were consistent across a wide range of riparian vegetation disturbance, and these values were comparable to natural rates except where disturbance was so severe that few or no trees were present adjacent to streams. Plot characteristics demonstrated a wide range in mean tree diameter and stem density as well as species composition, and litterfall inputs were not sensitive to these variables. While canopy cover was lowest for deciduous sites and highest for mixed sites, values were high for all forested plots. The presence of a closed canopy, typical of small streams, controls organic matter input rates.

Heterogeneity in plot characteristics

Heterogeneity and diversity are typical of the riparian ecotone (Naiman and Décamps, 1997), where both upland forest conditions and lotic forces influence vegetation characteristics at varying spatial scales. For example, sinuosity in small streams creates meander lengths much smaller than the 100-m reaches used in the present study, leading to a mosaic of landforms that favor a high degree of variability in the vegetation occurring in these areas. In western Washington, Douglas fir often occupies hillslopes and terraces, while red alder quickly colonizes floodplains, and natural riparian vegetation reflects the disturbance regime of the site.

Activities associated with urban and suburban development can compound or simplify natural heterogeneity. High-disturbance areas occurred where the riparian zone was stripped of all overstory vegetation and mowing practices disrupted regeneration. In this study, the two high-disturbance sites were homogeneously disturbed. For sites within other disturbance categories, however, many of the reaches crossed multiple property lines, where landowners manipulated vegetation to varying degrees and disturbances occurred at a finer scale than the 100-m study reaches.

Similar to the present study, Murray et al. (2000), Abee and Lavender (1972), and Zavitovski and Newton (1971) found no relationship between total litterfall rates and stand characteristics. Murray et al. (2000) did find a significant relationship between annual needle litterfall and western hemlock basal area, similar to the finding in the present study that needle inputs were related to conifer basal area. Bilby and Bisson (1992), however, found a difference between a forested stand and a recently clear-cut area, and O'Keefe and Naiman (2006) established a clear successional pattern. Stand characteristics, other than the total absence of trees within urban areas of the Puget Sound Lowland, do not appear to influence organic matter loads.

Study sites exhibited a wide range of plot characteristics and generally spanned the range found in sites used in previous litterfall studies that reported this information. Basal area in conifer-dominated reference sites fell within the range from two studies. The young Douglas fir stands studied by Dimock (1958) had a mean basal area of 36.8 m^2/ha, much lower than the reference sites in the present study. The six old-growth conifer sites in western Oregon studied by Abee and Lavender (1972) ranged from 83 to 130 m^2/ha. Plots used by Murray et al. (2000) were oriented parallel to the streams but apparently did not include active stream channels, which produced much lower total basal area than the Puget Sound Lowland plots. Mean diameter was much higher in the late seral plots of Abee and Lavender (1972) than in conifer plots in the current study. Stem density in 15 young (<33 years) upland red alder stands in western Oregon ranged from 600 to 130,000 stems/ha with a median of 5150 stems/ha (Zavitovski and Newton, 1971). Only the two restoration sites had such high stem densities, with most plots <2000 stems/ha. The conifer plots of Abee and Lavender (1972) had stem densities ranging from 168 to 494 stems/ha, generally lower than in the present study.

Stand characteristics within the forested plots (reference, low disturbance, and medium disturbance) used in the present study were similar to those used in previous studies. While they represented varying levels of human disturbance, reference, low-disturbance, and medium-disturbance site characteristics were not so different from natural sites used in previous studies that they preclude comparison.

Adapting methods for urban study areas

Unlike most litterfall studies, the present study utilized areas frequented by residents and recreational users on both private and public property. The study included 1,820 sample collections from 13 study areas, ten traps per reach, and 14 sampling events. As indicated by the pebble weighting the netting, 95.4% of samples were collected without interference. While some disturbance is expected in an urban setting, the high sample recovery in this study indicates that litterfall rates can be obtained for urban areas. However, two traps (1.5% of total) were discontinued during the study due to loss or damage, and another five traps were disturbed during two or more collection events. The overall high sample recovery may have been due to signage indicating the purpose of the research study and contact information on each trap.

Traps suspended over the stream were considered in the study design but were not feasible in an urban setting. However, given the very low difference between canopy cover over the traps and over the stream centerlines, litterfall collected by the traps situated on the banks was representative of that reaching the active stream channel.

Implications of changing vegetation on long-term nutrient loads

The large-scale conversion from predominantly conifer or mixed riparian forests to deciduous forests likely has increased the terrestrial allochthonous nutrient loads to small streams in the region. Pollen analyses in sediment cores collected from Lake Washington indicate a shift in dominant vegetation from conifer species (hemlock, cedar, and fir) to red alder between 1890 and 1916 (Davis, 1973), near the time that logging had removed most of the timber from the Seattle area. Crecelius et al. (2007) also noted the appearance of red alder in sediment cores collected from Hood Canal and Puget Sound. Coupled with the higher nitrogen and phosphorus loads quantified in this study, the historical vegetation shift from conifer to deciduous vegetation has resulted in higher nutrient inputs to streams, which may have altered nutrient dynamics.

While this study did not include stream nutrient concentrations, other studies indicate that some leakage of these higher nutrient inputs occurs and results in increased stream nutrient concentrations. For example, Brett et al. (2005) suggests that the presence of red alder trees in urban Seattle watersheds may contribute to autumn peaks in stream nitrogen concentrations. Volk (2004) found higher stream nitrate concentrations from red alder-dominated watersheds as compared with old-growth conifer watersheds on the Olympic Peninsula, while Osborne (2006) found that stream nutrient concentrations increased with the proportion of deciduous vegetation in the watersheds tributary to Hood Canal.

The nitrogen subsidy from red alder and other deciduous species could increase loads not just within small streams but also as loads delivered to downstream water bodies, such as Puget Sound. Increased incidence of low dissolved oxygen in Hood Canal and other parts of Puget Sound may be due in part to increased nitrogen loads from the historical conversion from conifer to deciduous forests, and the contribution is currently being assessed (Newton et al., 2006). Large-scale watershed vegetation changes could have profound effects on freshwater and marine productivity.

Riparian vegetation stands with high canopy cover produced similar organic matter loads, although the nutrient quality varied with the plot composition. Disturbed but forested areas were dominated by pioneer species, such as red alder, willow, and some non-native species, capable of producing high inputs even though the plot basal areas were much lower than for mature stands. Because leaf litter constitutes the bulk of the inputs, nitrogen loads were much higher than from conifer plots. Therefore, urban development can produce higher nitrogen loads than less-disturbed sites specifically due to riparian forest modifications.

In contrast, high-disturbance sites with no tree cover decreased organic matter inputs compared with the historical conifer-dominated conditions or more recent deciduous-dominated conditions. Where riparian trees had been removed, grasses and other materials produced some inputs, but at lower levels than those from forested areas. In addition, delivering nutrients in the form of grasses or small reproductive parts, as opposed to leaf litter, could decrease the availability of that organic matter for microbial and macroinvertebrate processing, since coarse particulate organic matter transport is a function of the size, shape, and density of materials. Canopy removal also likely shifts stream reaches from allochthonous based to autochthonous based, further altering organic matter processing.

Urban development clearly influences the quality and quantity of terrestrial inputs to small streams. Where recovery of allochthonous material into the aquatic food web is desired, results from the forested plots suggest that restoration of natural organic matter patterns should emphasize first the establishment of a closed canopy, with a secondary goal of transitioning to conifers over time.

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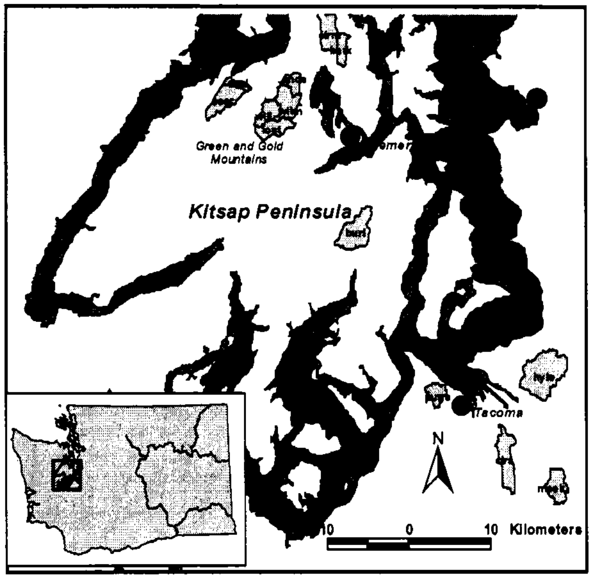

Figure 1-1. Litterfall study locations were selected using a stratified-random approach based on riparian vegetation characteristics interpreted from orthophotos. The sites represent the range of riparian vegetation disturbance found in the Puget Sound Lowland.

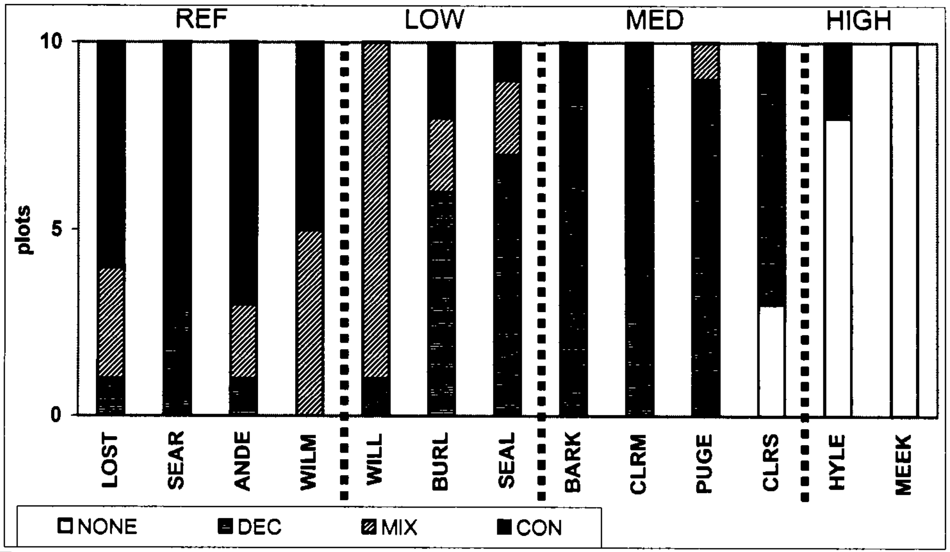

Figure 1-2. Distribution of plot type, determined from basal area within radial plots around each of ten traps, by study site. CON plots had >70% coniferous basal area, DEC had >70% deciduous basal area, and MIX plots were those where neither CON nor DEC basal area exceeded 70%. Where no trees occurred within the plots, the type was identified as NONE. Lines differentiate overall site riparian vegetation disturbance, determined from plot type, total basal area, and conifer basal area.

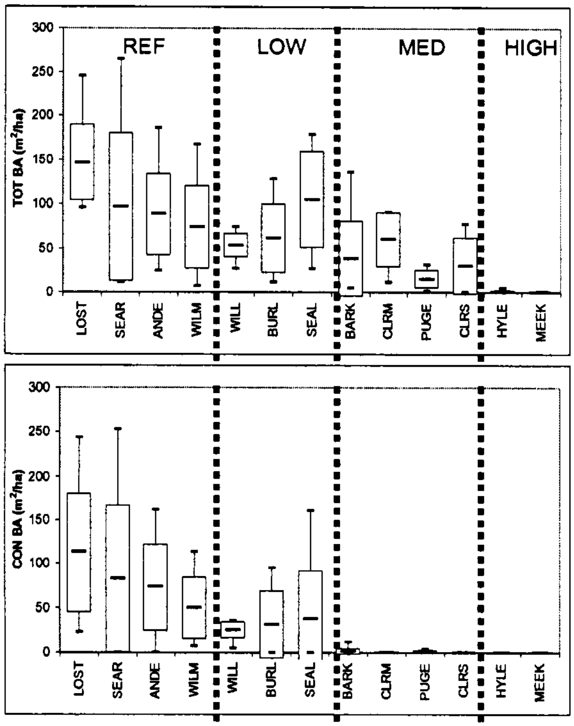

Figure 1-3. Total and conifer plot basal area (m^2/ha) by site. Bars indicate the mean and standard deviation for ten traps, and whiskers indicate minimum and maximum values. Lines distinguish riparian vegetation disturbance levels.

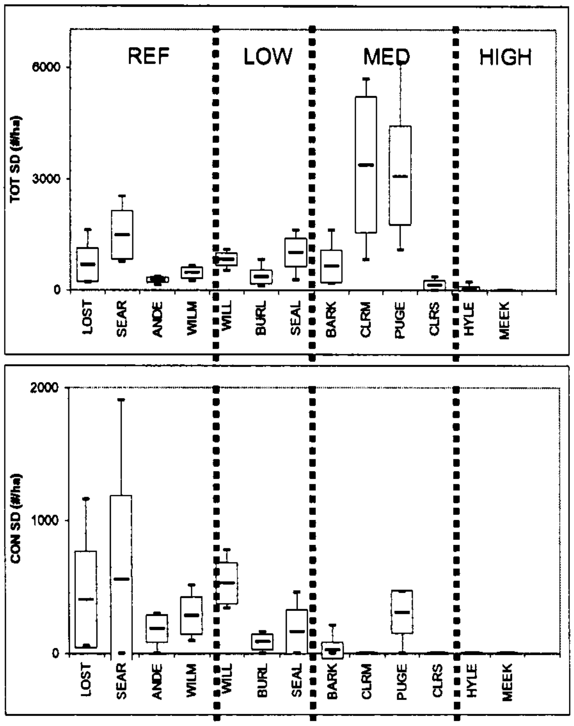

Figure 1-4. Total and conifer stem density (stem#/ha) by site. Bars indicate the mean and standard deviation for ten traps and whiskers indicate minimum and maximum values. Lines distinguish riparian vegetation disturbance levels.

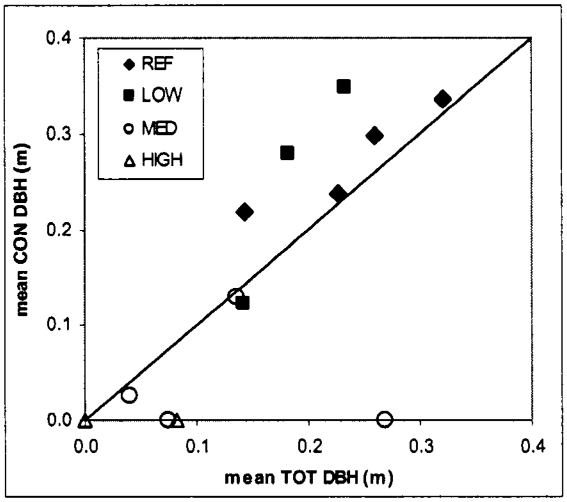

Figure 1-5. Mean conifer tree diameter (DBH) vs. total tree diameter by site. Mean plot DBH was determined from the basal area and stem density, and site values are the mean of 10 plots. Sites are coded by riparian vegetation disturbance level.

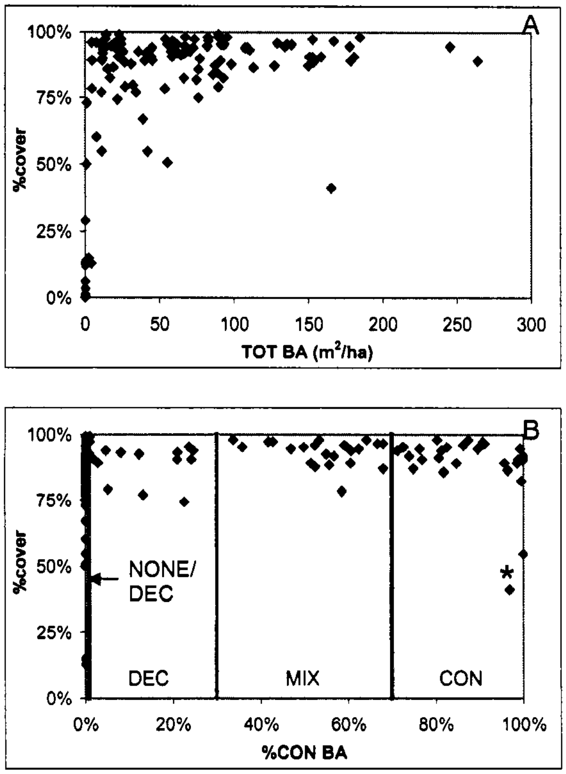

Figure 1-6. Riparian plot canopy cover (%), determined from densiometer readings, as a function of (A) total basal area and (B) percent conifer basal area. Low canopy cover occurs where basal area is near zero, but even plots with low basal area produce high canopy cover. The asterisk indicates two outliers where standing but dying red cedar trees produced high basal area but low canopy cover.

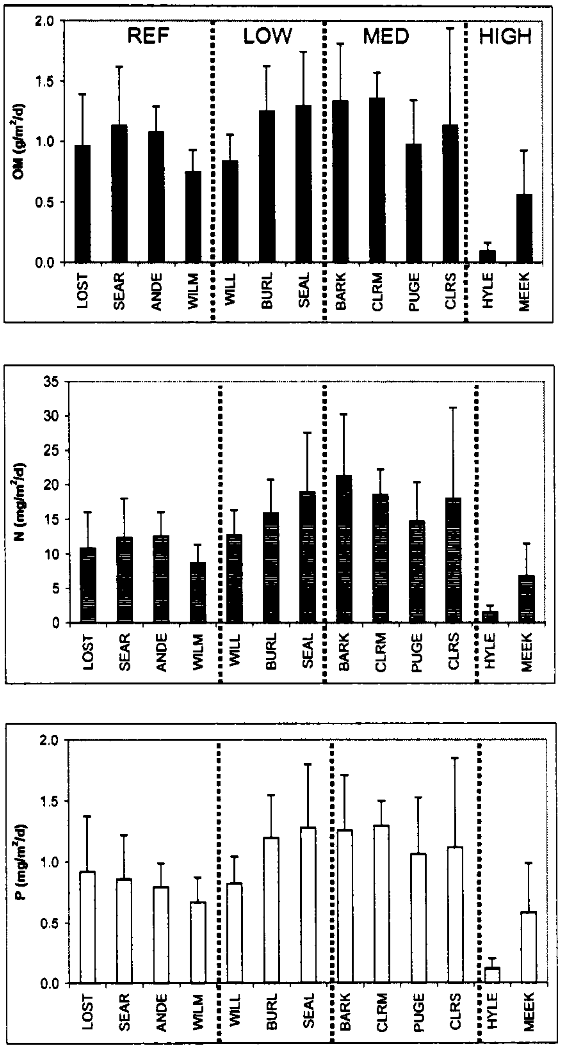

Figure 1-7. Annual organic matter, nitrogen, and phosphorus loading rates for each of the study sites. Columns indicate mean ±1 standard deviation (within-site). Lines distinguish riparian vegetation disturbance levels.

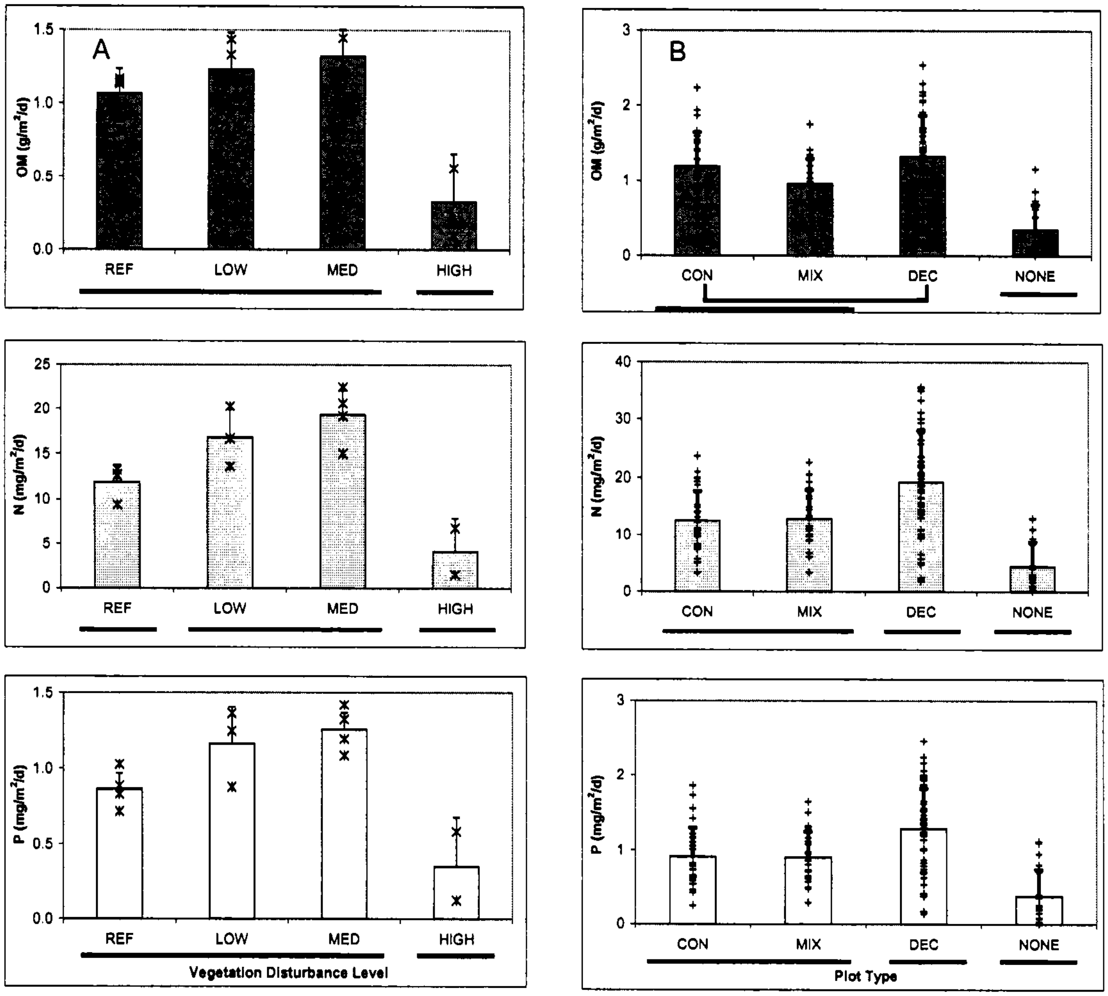

Figure 1-8. Annual organic matter, nitrogen, and phosphorus loading rates by (A) reach riparian vegetation disturbance level and (B) plot type. Columns indicate mean ±1 standard deviation, asterisks represent individual site means, and crosses indicate individual plot values. Lines identify categories that were not significantly different.

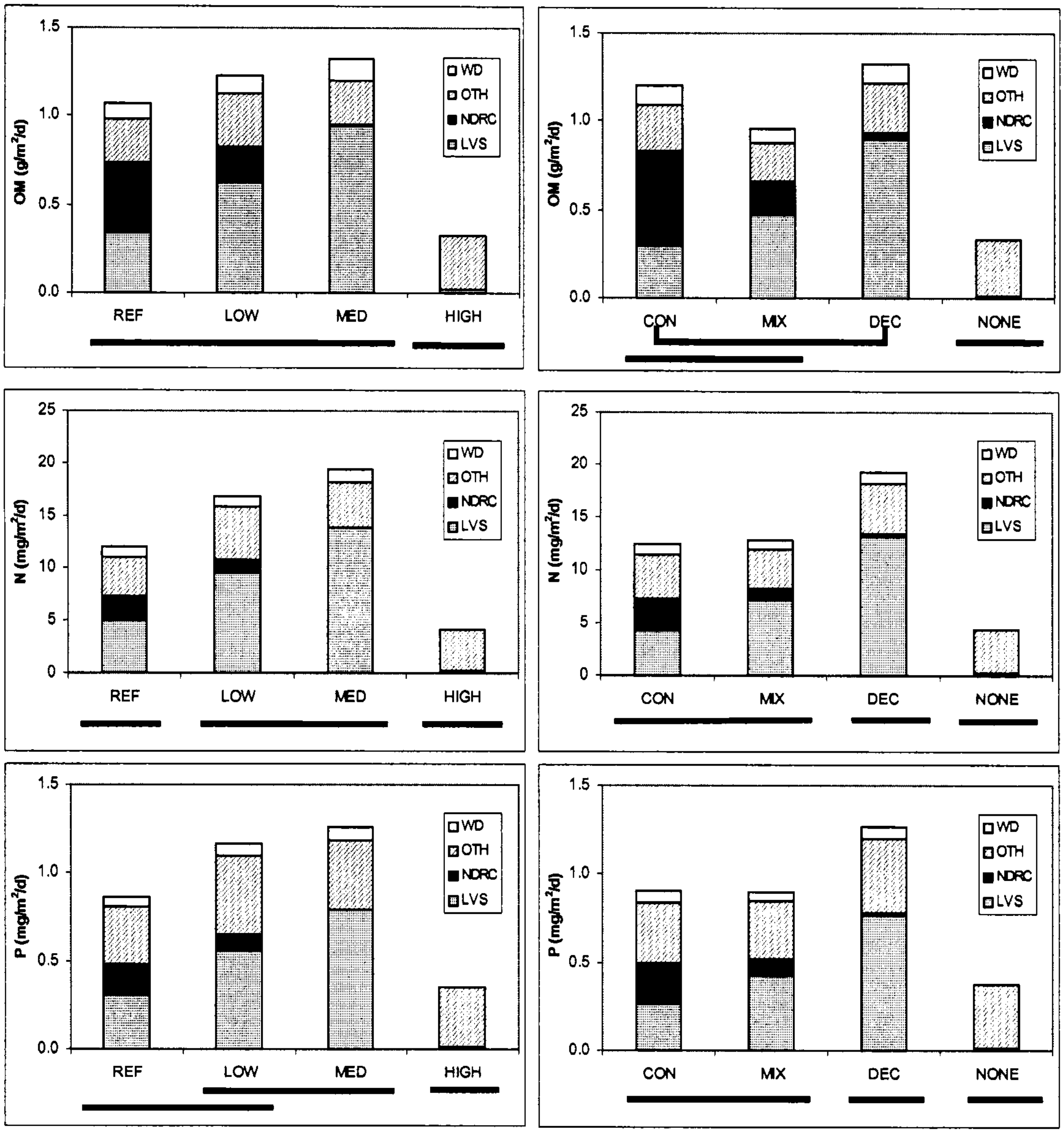

Figure 1-9. Annual organic matter, nitrogen, and phosphorus loads by component (LVS = leaves, NDRC = needles and red cedar, WD = wood, and OTH = other plant materials). Mean values are presented as a function of (A) overall site riparian vegetation disturbance level, and (B) individual plot type. Lines identify categories that are not statistically significantly different.

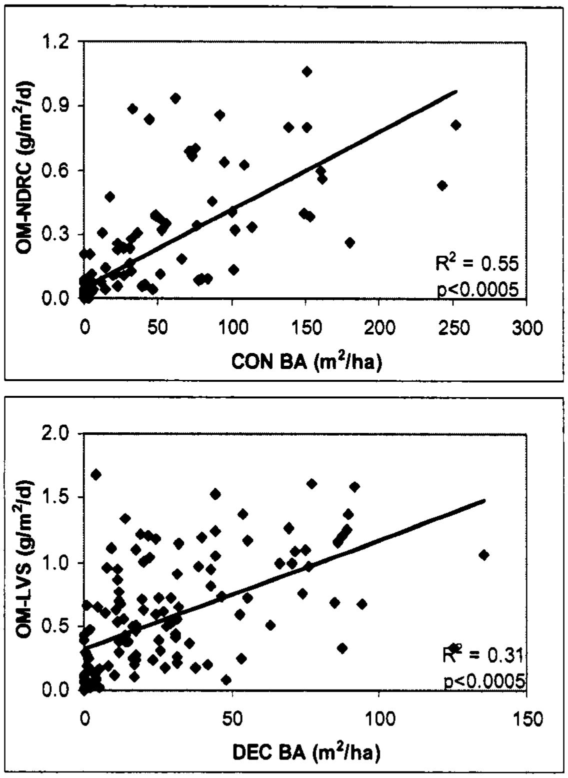

Figure 1-10. Relationships between litterfall components and plot characteristics. Needle and red cedar (NDRC) organic matter inputs are moderately related to conifer basal area, and leaf organic matter inputs (LVS) are weakly related to deciduous basal area.

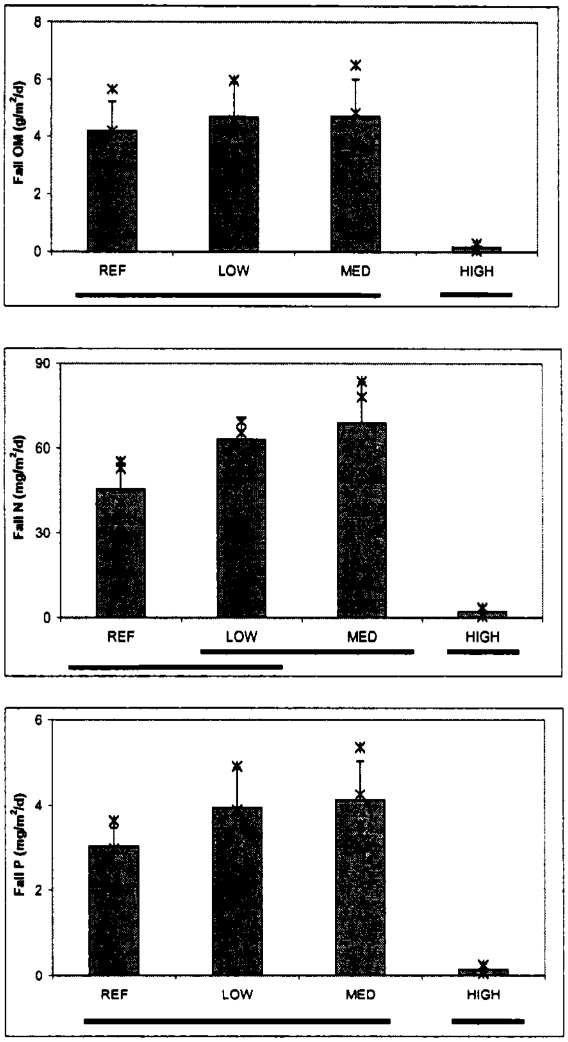

Figure 1-11. Autumn organic matter, nitrogen, and phosphorus loading rates by riparian vegetation disturbance level. Asterisks indicate site means. Lines identify categories that are not significantly different.

Table 1-1. Organic matter components and %OM determined from AFDM and oven-dry mass. Leaf materials without specific nutrient levels were assigned the mean LVOT values, as the composite mean of seven native and non-native species.

Fraction	**%OM (mean ± SD)**	**C (%)**	**N (%)**	**P (%)**
Red alder	95.1 ± 1.6%	47.8	1.87	0.091
Bigleaf maple	91.8 ± 1.6%	45.8	1.1	0.087
Catekins	96.7 ± 0.7%	48.3	0.84	0.045
Cones	97.6 ± 1.2%	48.3	0.84	0.045
Needles	95.1 ± 1.5%	50.1	0.62	0.060
Red cedar	94.8 ± 1.2%	50.5	0.59	0.042
Moss and lichen	95.0 ± 1.8%	43.7	1.74	0.171
Wood	96.5 ± 2.3%	48.6	1.02	0.062
Grass	92.5 ± 5.4%	43.1	1.51	0.129
Other>1	93.9 ± 7.5%	47.9	2.12	0.200
Other<1	86.2 ± 16.5%	47.9	2.12	0.200
Other Leaves:				
Other leaves (mean)	91.7 ± 4.6%		1.22	0.080
Black cottonwood	90.1 ± 3.8%			
Black hawthorn	93.5 ± 0.7%			
Cascara	90.4 ± 2.5%			
Dogwood	89.5 ± 3.4%			
English ivy	ND	45.0	1.98	0.113
Elderberry	91.1 ± 3.6%			
Holly	ND	47.7	0.81	0.042
Himalayan blackberry	93.0 ± 4.3%	45.2	0.88	0.069
Huckleberry	91.6 ± 6.1%			
Indian plum	87.7 ± 2.0%			
Japanese knotweed	86.9 ± 3.1%	42.7	1.25	0.095
Willow/oblong	92.7 ± 0.8%			
Salmonberry	93.6 ± 4.1%	45.5	1.77	0.069
Vine maple	92.1 ± 2.9%	44.8	1.02	0.079
Willow/rounded	91.5 ± 2.9%	48.8	0.80	0.094
Weeping willow	90.3 ± 3.7%			

Table 1-2. Tree and shrub species (>1-cm diameter) found within riparian plots along small Puget Sound Lowland streams or differentiated within litterfall samples.

Common name	**Latin name**
Native conifer trees	
Western hemlock	*Tsuga heterophylla*
Douglas fir	*Pseudotsuga menziesii*
Western red cedar	*Thuja plicata*
Sitka spruce	*Picea sitchensis*
Native deciduous trees	
Red alder	*Alnus rubra*
Bigleaf maple	*Acer macrophyllum*
Black cottonwood	*Populus balsamifera*
Willow	*Salix lucida* and *Salix sitchensis*
Vine maple	*Acer circinatum*
Cascara	*Rhamnus purshiana*
Black hawthorn	*Crataegus douglasii*
Indian plum	*Oemleria cerasiformis*
Native deciduous shrubs	
Beaked hazelnut	*Corylus cornuta*
Red osier dogwood	*Cornus stolonifera*
Red elderberry	*Sambucus racemosa*
Sitka mountain ash	*Sorbus sitchensis*
Pacific ninebark	*Physocarpus capitatus*
Oceanspray	*Holodiscus discolor*
Pacific rhododendron	*Rhododendron macrophyllum*
Yellow osier dogwood	*Cornus alba*
Crab apple	*Malus fusca*
Non-native conifer trees	
Cypress	*Taxodium* sp.
Fir species	*Abies* sp.
White pine	*Pinus* sp.
Non-native deciduous trees	
Apple	*Prunus* sp.
Mulberry	*Morus* sp.
Poplar	*Populus* sp.
Sassafras tree	*Sassafrax albidum*
Tulip tree	*Liriodendron tulipifera*
Non-native deciduous shrubs	
Apricot species	*Prunus* sp.
Azalea species	(unidentified)
Cotoneaster	*Cotoneaster* sp.
Ornamental deciduous shrub	(unidentified)
Non-tree invasive species	
Himalayan blackberry	*Rubus discolor*
Japanese knotweed	*Polygonum cuspidatum*
English holly	*Ilex aquifolium*
English ivy	*Hedera helix*

Table 1-3. Litterfall organic matter and nutrient levels from previous studies in the Pacific Northwest.

Component		**Murray et al. (1999)**		**Bilby and Bisson (1992)**		**Zavitovski and Newton (1971)**		**Gessel and Turner (1974)**	**Abee and Lavender (1972)**	
		lower WS	upper WS	clear cut	old growth	RA (yr 1)	RA (yr 2)	RA (8 yrs)	Old growth	Young DF
Organic matter ($g/m^2/d$)	Total	0.84	1.09	0.05	0.64	2.15 ± 0.38	2.02 ± 0.35	1.53 (1.26 to 1.81)	1.61 (1.24 to 1.89)	0.48
	Leaves	(none)	(none)			1.51 ± 0.24	1.35 ± 0.21		0.10 [C]	
	Needles	0.58	0.59						0.76	
	Wood	0.11	0.17			0.53 ± 0.22	0.65 ± 0.26		0.54	
	Repro	0.10	0.16						0.2	
	Misc	0.03	0.04			0.11 ± 0.09 [A]	0.03 ± 0.02 [A]			
Nitrogen levels (%)	Leaves	(none)	(none)			1.82 [B]		1.82	0.588	
	Needles	0.62 ± 0.10	0.77 ± 0.23					0.6	0.497	
	Wood	0.48 ± 0.08	0.55 ± 0.15						0.21 to 0.39	
	Repro	0.69 ± 0.33	0.84 ± 0.37						0.449 [E]	
	Misc	1.16 ± 0.22	0.93 ± 0.28						0.998 [D]	
Phosphorus levels (ug/g)	Leaves	(none)	(none)					300	940	
	Needles	614 ± 162	917 ± 208					300	1150	
	Wood	444 ± 69	583 ± 146						170 to 470	
	Repro	725 ± 540	931 ± 619						570 [E]	
	Misc	1216 ± 355	1126 ± 315						840 [D]	
Nitrogen loads ($mg/m^2/d$)	Total	5.29	7.81			39.1	36.8	27.8 (22.9 to 32.9)	7.48	
	Leaves	(none)	(none)						0.37	
	Needles	3.37	3.95						3.22	
	Wood	0.52	1.23						1.98	
	Repro	1.07	2.14						1.38	
	Misc	0.30	0.49						0.53	
Phosphorus loads ($mg/m^2/d$)	Total	0.55	0.99					0.46 (0.38 to 0.54)	1.29	
	Leaves	(none)	(none)						0.06	
	Needles	0.36	0.52						0.83	
	Wood	0.05	0.11						0.19	
	Repro	0.14	0.27						0.16	
	Misc	0.03	0.05						0.05	

A Catekins, cones, and other miscellaneous material

B Pooled across all ground components. Load calculated from value in Gessel and Turner (1974)

C All hardwood materials and mosses combined into one category

D Mosses and lichens

E Cones

Chapter 2

Dispersion and Reach-Averaged Velocity as Indicators of Enhanced Surrogate Transport in Small Streams Across an Urban Gradient

Introduction

Terrestrial organic matter represents an important source of energy and nutrients in many small streams (Triska et al., 1984; Vannote et al., 1980). Riparian vegetation type determines the amount of litterfall reaching streams (Bilby and Bisson, 1992; Gregory et al., 1987; Bray and Gorham, 1964), and also affects the physical hydraulic structure by influencing the channel complexity of small streams (reviewed in Tabacchi et al., 2000). The physical complexity, in turn, affects the retention of leaves and other organic matter (Bilby and Likens, 1980). Higher retentiveness increases storage of organic matter that may support macroinvertebrates relying on detritus as food or shelter (Laasonen et al., 1998).

The availability of organic matter in streams strongly affects overall ecosystem structure and function. In a long-term litter exclusion study at the Coweeta Hydrologic Laboratory, the decrease in leaf litter availability strongly affected aquatic communities and nutrient regimes. Litter exclusion resulted in a decline in the detritivore population, which utilized leaf litter directly, but the effects of exclusion cascaded to higher trophic levels as predator populations declined as well (Wallace et al., 1997). Webster et al. (2000) found that litter exclusion decreased phosphorus and ammonium retention compared with reaches receiving litterfall. Leaves provide a medium for biotic processes that sequester nutrients, and without those reaction sites nutrient concentrations are not mitigated by instream biotic processes. For example, Meyer et al. (1998) found that leaf litter exclusion decreased the dissolved organic carbon export.

Factors that affect relative leaf retention include stream order and discharge (Pretty and Dobson, 2004; Larrañaga et al., 2003; Snaddon et al., 1992; Jones and Smock, 1991; Young et al., 1978), channel gradient (Larrañaga et al., 2003), channel depth (Speaker et al., 1984), substrate texture (Larrañaga et al., 2003; Mathooko, 2001; Oelbermann and Gordon, 2000; Webster et al., 1994; Speaker et al., 1984), and channel complexity such as the pool/riffle structure (Brookshire and Dwire, 2003; Muotka and Laasonen, 2002; Lamberti et al., 1989). Evaluations of the influence of leaf features (e.g., size and flexibility) on transport have produced contradictory results. Pretty and Dobson (2004) and Steart et al. (2002) demonstrated that smaller materials travel farther than larger materials, but Young et al. (1978) found the opposite and Larrañaga et al. (2003) found no significant relationship.

Urbanization directly and indirectly affects a variety of processes that influence organic matter availability in small streams. Increased impervious surface area and hydraulic connectivity enhance the transport of stormwater from land surfaces to streams, increasing peak discharge and storm flow frequency (Konrad et al., 2005; Booth et al., 2002; Booth, 1991). Discharge in urban watersheds exhibits a spiky hydrologic response, with frequent high-flow pulses occurring in response to even low-intensity rain events (Konrad and Booth, 2002; Burges et al., 1998).

Urbanization also alters stream channel geomorphology. Traditional stormwater infrastructure and flood control practices enhance stormwater runoff and directly reduce channel complexity through channel straightening and removal of woody debris, practices that destabilize stream channels (Booth et al., 2002). Elevated water velocities and reduced woody debris inputs associated with urbanization decrease the replenishment of structural channel complexity elements, indirectly decreasing channel complexity. Streams with simplified channel geomorphology have fewer sites where particulate organic matter can accumulate, decreasing organic matter retention and enhancing transport.

Finally, riparian vegetation closely tracks overall watershed development levels, with more disturbed vegetation patterns found where development levels are higher (Morley and Karr, 2002). Historical riparian vegetation influences existing instream wood accumulations, while current riparian vegetation structure determines future recruitment and replenishment of woody debris. Therefore, riparian vegetation not only produces litterfall inputs but also influences the retention and transport of organic matter within streams, through woody debris accumulations, and effects on the pool/riffle morphology.

Stream restoration plans generally focus on channel characteristics and riparian vegetation planting, both of which could improve the retention of organic matter within the restored stream reach. However, organic matter retention and transport are not addressed explicitly in the plans, nor do restoration efforts include measures of organic matter retention and transport.

Because leaves and other organic matter are difficult to track and identify in situ (e.g., Muotka and Laasonen, 2002), previous studies have relied upon both natural and surrogate materials that are highly visible yet retain the relative transport characteristics of natural organic matter. Flagging tape (Bilby, 1981), plastic strips (Larrañaga et al., 2003; Díez et al., 2000; Richardson and Maxcy, 1997; Wallace et al., 1995; Webster et al., 1994; Speaker et al., 1988), Rite in the Rain™ paper (Brookshire and Dwire, 2003), painted or dyed natural leaves (Pretty and Dobson, 2004; Mathooko et al., 2001; Chergui et al., 1993; Jones and Smock, 1991), gingko leaves (Brookshire and Dwire, 2003; Ehrman and Lamberti, 1992) and native vegetation (Larrañaga et al., 2003; Oelbermann and Gordon, 2000) have been used to compare organic matter transport among study sites. Experimental design varied widely. The numbers of surrogate material pieces released to characterize a given stream reach range from 20 (Pretty and Dobson, 2004) to over 1,000 (Larrañaga et al., 2003; Muotka and Laasonen, 2002). Time between release and collection generally have ranged from 1 minute (Pretty and Dobson, 2004) to three hours (Larrañaga et al., 2003; Muotka and Laasonen, 2002; Díez et al., 2000), although a few studies have used time periods of months between release and collection (Bilby and Likens, 1980). Stream lengths characterized by these studies tend to be short.

The factors that affect relative organic matter transport also influence hydraulic characteristics, such as reach-averaged velocity and dispersion, and these may be quantified over longer stream reach lengths than organic matter transport. Dissolved solute studies may be used to determine reach-averaged velocity, based on the distance between two points along a stream divided by the travel time between them. Thus, reach-averaged velocity represents the cumulative velocity over longer distances that may encompass multiple pool/riffle units. In streams, longitudinal dispersion refers to the spread of water parcels in a downstream direction that results from velocity variations. Wilcock et al. (1999) found longitudinal dispersion was positively related to stream discharge and velocity, and the presence or absence of aquatic macrophytes affected the velocity structure, which also resulted in differences in dispersion. Koussis and Rodríguez-Mirasol (1998) found longitudinal dispersion varied with stream depth, width, wetted cross section, and channel slope.

The purpose of this study was to evaluate relative organic matter retention across an urban gradient using surrogate leaf materials. In addition, an alternative method was evaluated to characterize relative leaf litter transport using two hydraulic characteristics, reach-averaged velocity and hydraulic dispersion.

Methods

Study area description

Hundreds of small streams flow through the Puget Sound Lowland ecoregion of western Washington. In the central part of the lowland, many of these streams originate in the bedrock of the Green and Gold Mountains and the outwash glacial till of the Kitsap Peninsula and greater Tacoma area. These small streams typically exhibit a pool/riffle morphology. Channel substrates consist of coarse gravel to silt, with the latter occupying low-gradient streams or urban areas.

Human influences include widespread timber harvesting in the Puget Sound Lowland in the late 1800s and early 1900s, followed by increasing residential and commercial development in the mid- to late-1900s. King County and Pierce County, on the eastern shore of Puget Sound, developed early in this progression, due to the economic activities of Seattle and Tacoma (Figure 2-1). Between 1980 and 2005, however, the population in Kitsap County, on the western shore of Puget Sound, increased by 63% (Washington State Office of Financial Management, 2006b), although level of development remains considerably lower than in the Seattle-Tacoma area. Riparian vegetation disturbance patterns reflect these spatial and temporal human patterns, with riparian disturbance in Kitsap County less extensive and more recent compared with King and Pierce counties.

Study site selection

Study sites (Figure 2-1) were selected using a stratified-random approach. Small streams with watershed areas between 4 and 24 km^2 were identified using a 30-m digital elevation model. In each watershed, primary stream centerlines were refined using orthophotos. The polylines were segmented at 100-m intervals (Oregon DEQ, 2001). Only accessible points, defined as those within 1 km of a road or trail, were retained. Each point was coded with the initial riparian vegetation disturbance level, assigned from previous qualitative field observations supplemented by orthophoto interpretation. Ten sites were identified within each of four initial disturbance levels using a random number generator. Property owners were contacted to request initial access to the locations; if permission to monitor that reach was not given, the site was removed from consideration.

Categorical riparian vegetation disturbance levels were assigned based on vegetation species and size within plots, as described in Chapter 1. The final study design included four sites each within reference and low-disturbance sites and three sites each within medium- and high-disturbance sites (after one medium- and one high-disturbance site were discontinued due to vandalism).

Four disturbance categories based on the condition of riparian vegetation were used. Reference vegetation (REF) was defined by the presence of mature conifer-dominated and mixed riparian forests. While not pristine, these areas represented the lowest level of human disturbance existing in the study area. Low-disturbance sites (LOW) included mature, deciduous-dominated vegetation with a few conifers. Medium-disturbance sites (MED) included a range of vegetation characteristics, from very young natural vegetation to residential landscaping to restoration sites with native vegetation plantings. High-disturbance sites (HIGH) generally lacked overstory trees completely, and herbaceous vegetation dominated.

Field methods

Discharge was determined using the velocity-area method (Marsh-McBirney velocity meter) once during the late-summer low-flow conditions targeted in the study, flow conditions also typical of autumn non-storm conditions when litterfall inputs increase (Chapter 1). Riffles within study reaches were used. Station velocity (V_{sta}) was found from the discharge and the cross-sectional area at the discharge location.

The channel morphology, defined as the relative cumulative length of pools and riffles in a 100-m study reach, was determined during summer low-flow conditions. Pools were defined as water features with subcritical flow characteristics, where waves propagate upstream, and riffles as supercritical reaches where waves do not propagate upstream. For a subset of eight streams, two in each disturbance category, spring and summer pool and riffle areas were quantified for four randomly selected pools and four randomly selected riffles. Pool/riffle ratios (P/R) also were determined during spring conditions to compare results between low-flow and higher-flow conditions.

The number of wood jams per 100-m reach was tabulated as an additional indicator of relative reach complexity. Jams were defined as containing at least three pieces of wood (minimum size 10-cm diameter and 1-m length) within the wetted margin.

Reach-averaged velocity was determined using a pulse release of saturated salt solution at the top of a riffle (Hubbard et al., 1982). Reach lengths for hydraulic characterization depended on the pool/riffle structure, ranging from 14 m to 108 m. The original 100-m reaches were modified based on the hydraulics and geomorphology at each site. Release locations were shifted to the nearest upstream riffle to achieve rapid mixing of the solute. Measurement locations, also were shifted to the nearest downstream riffle. Reach lengths were decreased due to sluggish velocities or a change in geomorphology. For example, the release at WILL used a reach length of 34 m to encompass multiple pool and riffle units due to the very long travel time (4.9 hours) from sluggish velocity (0.004 m/s). Reach length at CRES was decreased because of a wood constriction near the upstream end of the reach that would have affected surrogate travel. At least two pool and riffle sequences, and generally five to nine sequences, were used.

Enough salt solution was released such that the pulse was discernible above background levels but peak conductivities did not exceed 1000 umhos/cm at the release location or 400 umhos/cm at the bottom of the reach. A Hanna Instruments (model HI-9033) conductivity meter, calibrated to a solution of 100 umhos/cm and self-compensating for temperature, recorded conductivity at a point downstream from the salt release. Conductivity was recorded until in situ values approached background levels. Reach-averaged velocity was determined from the travel time for the tracer concentration center of mass from the release point to the conductivity recorder.

Estimating reach-averaged velocity and longitudinal dispersion

Longitudinal dispersion, also called mechanical or hydraulic dispersion, refers to the spread of a dissolved constituent, such as a tracer or contaminant, that results from velocity variations within the stream. Dispersion includes the effects of turbulent mixing, although vertical velocity variations represent the dominant dispersive mechanism.

Dispersion is analogous to molecular diffusion and is represented using a form of Fick's first law. In one-dimensional systems, such as streams, the longitudinal dispersion coefficient characterizes mixing, which increases as the velocity distribution becomes more complex.

The one-dimensional mass transport equation

$$\frac{\partial C}{\partial t}+U\frac{\partial C}{\partial x}=D_L\frac{\partial^2 C}{\partial x^2}$$

can be solved for solute concentration (C) resulting from an instantaneous solute release:

$$C = \left[\frac{M}{2A\sqrt{\pi D_L t}} \right] e^{\frac{-(x-Ut)^2}{4D_L t}}$$

where M is total solute mass, A is the stream cross-sectional area, D_L is the dispersion coefficient, x is longitudinal distance, U is reach-averaged velocity, and t is time. The form of the equation is Gaussian with respect to location but not strictly with time, since time is retained in the denominator. When using conductivity, solute mass is not easily measured in the field but can be estimated based on the maximum concentration (C_{max}) observed and the distance from the release:

$$C_{\max}(t) = \frac{M}{2A\sqrt{\pi D_L t}}.$$

Therefore, the reach-averaged velocity and dispersion coefficient were found using a best fit to the concentration data by minimizing the root-mean-square error.

Surrogate leaf travel distance

Relative transport and retention of leaf litter were determined using surrogate leaf materials. A small-scale trial compared the relative transport of a variety of materials with that of red alder leaves to select the best material. Materials included yellow- and orange-colored acetate, red alder leaves tagged with permanent marker, white cotton cloth, yellow fabric, pink heavy polyethylene, orange medium polyethylene, and orange-coated nylon. Materials were cut into diamond or leaf shapes approximately the same size as red alder leaves. After preliminary testing, orange diamond-shaped acetate surrogate leaves, approximately 7.8 cm by 7.8 cm by 0.16 mm thick and weighing 1.2 g, were released at the head of a riffle for each experimental run. Distance traveled was recorded for each surrogate leaf within one hour of release. At summer low-flow conditions, steady flow induced no appreciable movement after several minutes, once the surrogate materials caught on an obstacle. Nearly all released pieces were retained within several minutes.

For the surrogate leaf materials, the median distance traveled was used as a measure of central tendency, because the distances were not normally distributed. Previous studies (Larrañaga et al., 2003; Speaker et al., 1988) estimated mean travel distance by fitting a first-order decay relationship to the travel distance then calculating mean distance as the inverse of the decay coefficient (k):

$$N(x) = N_o e^{-kx}$$

where N(x) is the number of leaves passing a distance x from the release and No is the total number of materials released. However, while much of the variability was described by these relationships ($R^2 = 0.55$ to 0.97), the few materials that traveled the farthest heavily influenced the shape of the distribution. Therefore, the median was used as a more appropriate measure of central tendency (Zar, 1999).

Skewness indicates the degree of asymmetry in the distribution. The interquartile range, or the difference between the 25th and 75th percentile distances, was used to characterize the spread in the travel distance of the surrogate materials.

Statistical analyses

Relationships among transport parameters and channel characteristics were based on linear regression, while Student's t-tests were used to compare characteristics between groups. For all tests, $\alpha \leq 0.05$ was used to determine significance.

Results

Site characteristics

Table 2-1 summarizes the field parameters recorded during late-summer low-flow conditions. Chapter 1 describes land cover characteristics, determined from a 1998 LandSat Thematic Mapper Image (Hill et al., 2003) for the sites in this study. Developed land, the sum of several classification categories (high density, low density, and bare ground), ranged from 5 to 18% of the watershed areas for sites with REF or LOW riparian vegetation disturbance, while sites with MED or HIGH riparian vegetation disturbance levels had 20% to 63% developed land. Forest cover, the sum of conifer, mixed, and deciduous covers, ranged from 30 to 45% for the less-disturbed riparian vegetation sites, but only 5 to 32% for the more disturbed sites.

One site (BURL) had a much higher discharge than the others even though its watershed area was comparable to other study sites, likely due to higher groundwater inflows within this watershed. The summer unit discharge in BURL was four times the next highest value, indicating a much different summer hydrologic regime.

The cross-sectional station velocity (V_{sta}) calculated from the discharge measurement was greater than the solute-derived reach-averaged velocity (U) at most sites, since discharge measurements generally were conducted within fast-moving riffles. The reach-averaged velocity represented the net effect of riffle and pool velocities in the study reaches.

The number of wood jams tended to be greater in the REF and LOW-disturbance sites as compared with the MED- and HIGH-disturbance sites, although the difference was not significant ($0.05<p<0.10$). Because of the lack of wood in the relatively short reaches used in this study, P/R was selected as a better indicator of reach channel complexity.

Pool/riffle ratios

Most studies quantify P/R using the relative lengths of pools and riffles in a reach during low-flow conditions. This approach assumes that low-flow conditions reflect other ecologically meaningful conditions and that relative length accurately represents pool and riffle area. These assumptions were verified by repeating the pool/riffle surveys during high spring discharges and by comparing P/R derived from lengths with those based on feature area.

For summer low-flow conditions, the P/R calculated from relative length was an appropriate proxy for total feature area (Figure 2-2a), since summer P/R based on length was not significantly different than P/R based on area (t-test, $p=0.36$). The regression relationship was significant both with station CLRH ($p<0.005$) and without station CLRH ($p<0.01$), a site with a much higher P/R than any other site and appeared as an outlier. CLRH had a different morphology than the other study streams. Glides were classified as pools due to the hydraulic definition, where waves propagate upstream. While the study reach exhibited some pool/riffle structure, the low gradient of the valley resulted in very slow-moving water, and areas that may have been riffles during other flow regimes exhibited subcritical flow more typically associated with pools. Thus, the use of subcritical flow as a determinant of the feature type resulted in a higher proportion of pools at this site than the other study reaches. Because no significant difference was found, the additional field effort necessary to determine pool and riffle areas was not warranted for the study streams.

Summer and spring P/R were related (Figure 2-2b), but only when station CLRH was included. Overall, the proportion of pools was lower during moderate spring discharges for three locations, likely due to the exposure of fine-scale bottom features at the very low flows encountered during summer surveys. However, the difference between summer P/R and spring P/R was not significant (t-test, $p=0.37$) with or without CLRH. Therefore, while P/R values determined at multiple flow rates were related in some cases, the relationship did not hold across all flow rates.

These results suggest the need to establish P/R ratios at flow rates of interest to a particular study, since extrapolation to other flow conditions may not be justified. Since the purpose of the present study was to evaluate relative leaf retention during late-summer conditions, the late-summer P/R ratio based on relative feature length was used for the remainder of the analyses.

Dissolved tracer studies

Both solute velocity and dispersion increased with discharge (Figure 2-3a), but one site dominated both regression relationships. Without the data from BURL, both relationships were still highly significant, and R2 values increase to 0.73 and 0.42 for solute velocity and dispersion, respectively. Most of the sites had very low discharge rates (<0.02 m^3/s), and at low discharge the sites showed high variability in solute velocity and dispersion rates. Solute velocity also increased with dispersion (Figure 2-3b), and dispersion coefficients generally provided a better prediction of reach velocity than discharge.

As the proportion of pools increased, the reach-averaged velocity decreased (Figure 2-4). The solute velocity was moderately well described by the P/R ratio ($R^2 = 0.62$, $p<0.0025$). However, the P/R ratio described more of the variability in dispersion ($R^2 = 0.80$, $p<0.0005$) than variability in the reach-averaged velocity, and both relationships were significant. CLRH was removed from this analysis because the stream type differed from the other sites.

Surrogate leaf transport and retention

Both dispersion and solute velocity described the median distance traveled by the surrogate leaves well (Figure 2-5). During low-flow summer conditions, when surrogate leaves were released, no appreciable movement was observed once the surrogate materials caught on an obstacle. Slower water retained more leaves than faster water. The results also exhibited an increase in median travel distance with discharge in the systems ($R^2 = 0.85$), as has been found in previous investigations (Pretty and Dobson, 2004; Larrañaga et al., 2003; Snaddon et al., 1992; Jones and Smock, 1991). Station BARK was the exception, where the travel distance was controlled by wood in the stream close to the release location rather than by dispersion or solute velocity. Station BURL was removed from the analysis because a wood-induced constriction much smaller than the reach width produced very high velocities at the point of release, local conditions that were not representative of the reach.

Surrogate material travel distances also exhibited greater spread in the results with increasing solute velocity and dispersion. The interquartile range increased with both solute velocity ($R^2=0.63$) and dispersion ($R^2=0.69$), and the relationships were highly significant ($p<0.001$ and $p<0.0005$, respectively) (Figure 2-6). As with the median travel distance, the interquartile range was significantly related to discharge ($p<0.05$), although discharge did not explain as much of the variability ($R^2=0.31$) as did the solute velocity and dispersion coefficient.

The travel distances exhibited positive skew for nearly all locations, and the median travel distance was lower than the mean. Notably, BURL, with the higher flows, approached a normal distribution, and CLRH exhibited a slight negative skew. Skewness decreased with increasing discharge, station velocity, solute velocity, and dispersion, but the relationships were not significant.

At very low flows, a few surrogate leaves traveled much longer distances than the bulk of the leaves, but 50% of the surrogates were retained within 1.4 m of the release across all study locations. Ninety percent of all surrogates released in the various stream reaches were retained within 11.3 m. The median travel distance was significantly higher in sites with MED- and HIGH-disturbance riparian vegetation disturbance levels (2.7 m) as compared with sites with REF and LOW-disturbance levels (1.1 m).

Relationships between transport metrics, watershed development, and riparian vegetation disturbance levels

Study sites were selected on the basis of riparian vegetation disturbance, originally from orthophoto interpretation but refined using plot characteristics. However, the study sites also reflected a range of watershed development levels, which indirectly affect hydraulic characteristics and geomorphology within the streams. Therefore, both watershed development and local riparian vegetation disturbance level were compared with transport metrics.

Stream channel and hydraulic properties reflected the level of watershed development (Figure 2-7). As the proportion of pools increased among sites, reach-averaged velocity decreased, with the exception of BURL and CLRH, described previously. Watersheds with lower levels of development (DEV<10%) had higher P/R than watersheds with higher development levels, which resulted in lower water velocities. Low-development watersheds also produced lower median travel distances for surrogate leaves. Therefore, the hydraulic and channel complexity effects of watershed development affect leaf litter retention, and stream reaches with low-development watersheds retain more leaf litter than those with higher-development watersheds.

The P/R, solute velocity, surrogate transport, and dispersion relationships also reflected the riparian vegetation disturbance level. Less-disturbed vegetation sites (REF and LOW in Figure 2-8) exhibited higher P/R and lower solute velocity, surrogate transport distance, and dispersion, while more-disturbed sites (MED and HIGH) exhibited lower P/R and greater solute velocity, surrogate transport distance, and dispersion. Therefore, the more mature trees associated with less-disturbed riparian vegetation likely increased channel complexity, which enhances leaf litter retention.

Discussion

Small streams generally are very effective at retaining organic matter. The streams in this study were highly retentive, but level of development, channel complexity and hydraulic characteristics clearly influenced surrogate leaf transport. Surrogate material median transport distance decreased as pools increased and reach-averaged velocity decreased. Streams with a higher proportion of riffles had lower P/R ratios and higher velocities, characteristics that increased hydraulic dispersion and also produced greater travel distance.

Leaf transport reflects both watershed development and local riparian vegetation

Watershed development potentially affects organic matter transport characteristics through a variety of pathways. Increased watershed impervious cover and hydraulic connectivity that result from stormwater management infrastructure directly affect hydraulics by increasing stormwater peak flows and frequency. In addition, higher stream reach velocities likely decrease channel complexity by eliminating complexity elements, such as wood jams and sinuosity, or by hindering establishment of these elements. Previous studies demonstrated that removing complexity elements reduces organic mater retention (Bilby, 1981; Bilby and Likens 1980). These stream structural changes impact stream ecological functions, as leaf transport distance increased with watershed development.

Local riparian vegetation disturbance level also reflected watershed land cover characteristics. In this study, the eight stream reaches with less-disturbed riparian vegetation also had lower watershed development (5 to 18%) and higher forest cover (30 to 48%) than the six stream reaches with more-disturbed riparian vegetation (20 to 60% watershed development and 5 to 32% forest cover). As for watershed-scale development, local-scale riparian vegetation disturbance influenced leaf litter retention, and median surrogate travel distances were lower for reference and low-disturbance vegetation sites (1.1 m) than for medium- and high-disturbance vegetation sites (2.7 m).

The effects of urbanization on hydrology and geomorphology, coupled with riparian vegetation alterations such as removal of mature native vegetation, destabilize stream channels, which lose components such as wood jams as development increases (Booth et al., 2002). These changes in turn affect the transport and retention of organic matter. Stream reaches with more disturbed vegetation and more developed watersheds produce greater water velocities, which reduces organic matter retention and transports organic matter farther from the point of immersion than stream reaches with less disturbed vegetation and less developed watersheds. Higher velocities and lower pool/riffle ratios associated with more disturbed riparian vegetation and higher watershed development increase hydraulic dispersion as well.

Hydraulic characteristics describe ecological processes

Hydraulic parameters, such as reach-averaged velocity and dispersion, characterized leaf litter retention on a reach scale. Neither parameter has been used extensively in stream ecology studies, but both relate to a fundamental ecological process that determines organic matter availability for biotic functions.

Based on relationships between hydraulic characteristics and surrogate leaf transport in this study, hydraulic dispersion is an ecologically meaningful parameter because the velocity variations that control dispersion also reflect channel complexity, and channel complexity influences organic matter transport and retention. Organic matter transport increased with increasing discharge, but discharge explained less of the variability in median travel distance than did reach-averaged velocity and dispersion. Surrogate transport increased as reach-averaged velocity and longitudinal dispersion increased. Both solute-derived parameters are easily determined from field experiments that may be conducted at any flow conditions of interest to ecological studies.

One advantage of using hydraulic characteristics like reach-averaged velocity and dispersion is that simple field experiments can characterize longer reach lengths than can traditional organic matter surrogate release experiments. In this study, the solute study reach length was selected to encompass multiple pool/riffle units. Even longer reaches may be evaluated using these methods as long as the solute concentration is discernible above background levels. Evaluating longer reaches reduces the effect of unusual local conditions, as occurred at BURL, where a wood constriction enhanced water velocity that then transported surrogate materials farther than if the leaves had been released within a more typical riffle. These field-based programs provide an objective measure of channel complexity, whereas other channel survey field methods often are observer dependent, making comparisons among study results difficult.

This study was conducted during low-flow conditions, which occur in the Pacific Northwest in early autumn, when peak litterfall inputs occur (Chapter 1). Leaf retention at higher discharge would reflect the channel complexity and hydraulic characteristics of these higher flows. Complexity elements that retain organic matter during low flows may be submerged and ineffective at high flows. In this study, for example, the proportion of pools was lower at the higher spring discharge as compared with summer low-flow conditions, although the difference was not statistically significant. This study was not designed to evaluate the effects of higher discharge or stormwater flows on leaf litter transport directly. However, future studies should evaluate whether organic matter retention during higher discharge also reflects stream channel characteristics that result from watershed development and riparian vegetation disturbance levels.

Restoration plans should include organic matter retention and related variables

Increased organic matter transport has not been considered widely as one of the potential effects of urbanization. As suggested by Muotka and Laasonen (2002) and others, organic matter retention should be considered in stream restoration plans because organic matter availability strongly affects allochthonous ecosystem function. Many restoration efforts likely enhance organic matter availability indirectly through plantings of native understory and overstory vegetation and by increasing the pool/riffle ratio using engineered complexity units like root wads and log weirs. Monitoring plans could include hydraulic parameters determined before and after restoration as a measure of the relative success of restoration projects in enhancing channel complexity and related ecological processes.

Leaf litter represents a critical element to the support of trophic processes in stream ecosystems (Wallace et al., 1997; Cummins, 1974). Leaf litter availability may decline through a number of pathways, including decreased inputs (Chapter 1), enhanced decomposition rates (Chapter 3), and enhanced transport. Urbanization potentially affects each of these pathways such that leaf litter availability is limited in urban streams. Therefore, organic matter retention should be considered in stream restoration plans because it supports allochthonous ecosystem function. This study demonstrated experimentally that reduced retention and enhanced transport in small streams result from the combined effects of urbanization, including watershed development level, riparian vegetation disturbance level, channel complexity, and hydraulic characteristics. Hydraulic characteristics such as reach velocity and longitudinal dispersion offer relatively simple, field-based indications of organic matter retention over longer reach lengths than can be characterized by traditional surrogate release experiments.

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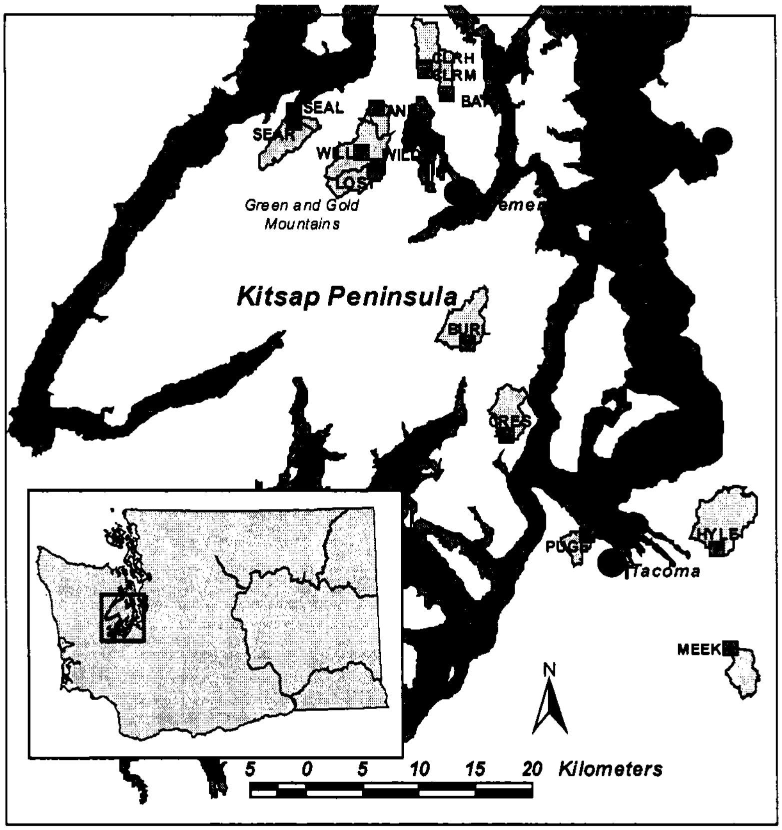

Figure 2-1. Site locations for surrogate material transport studies.

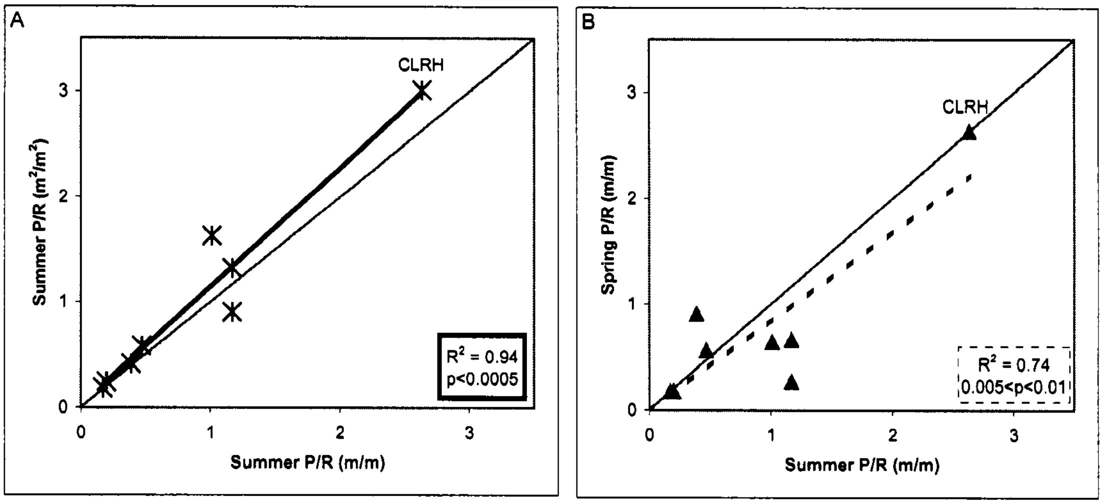

Figure 2-2. P/R ratio (m^2/m^2) for late-summer low discharge (A) and P/R ratio (m/m) for moderate spring discharge (B) as a function of the summer P/R ratio (m/m).

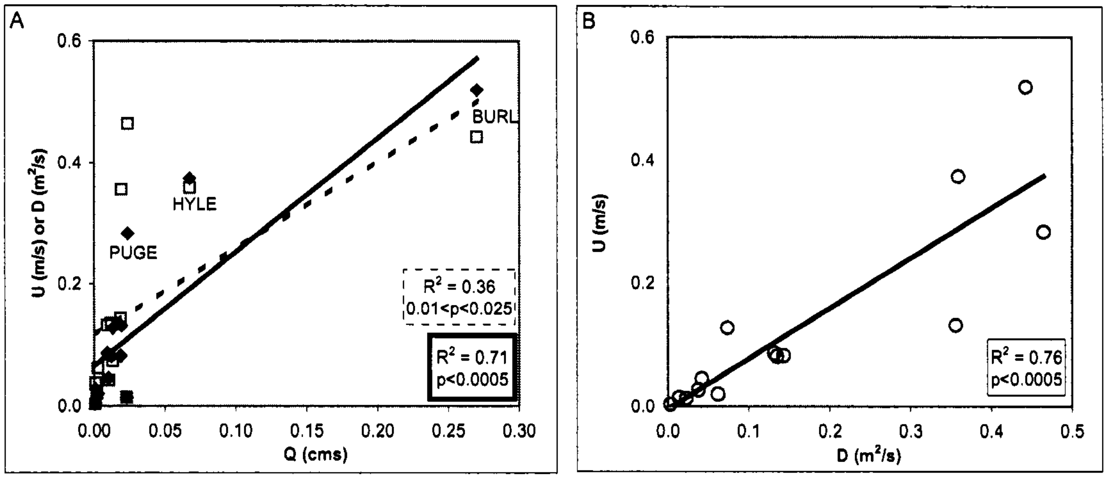

Figure 2-3. (A) Solute velocity (diamonds and solid line) and longitudinal dispersion coefficients (open squares and dashed line) as a function of stream discharge and (B) relationship between solute velocity and dispersion.

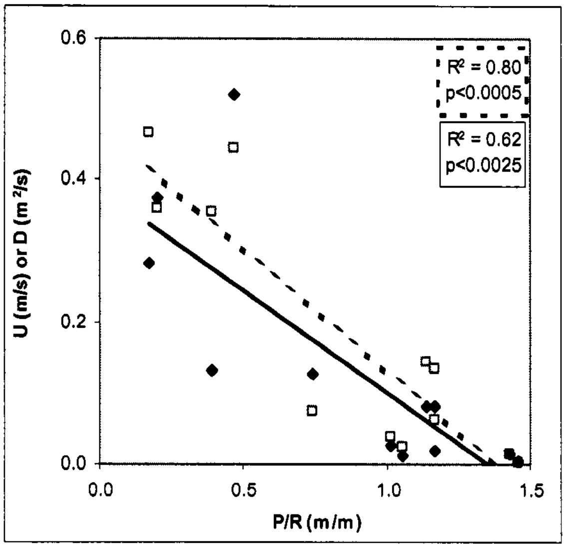

Figure 2-4. Solute velocity (diamonds and solid line) and dispersion (open squares and dashed line) as a function of the P/R ratio.

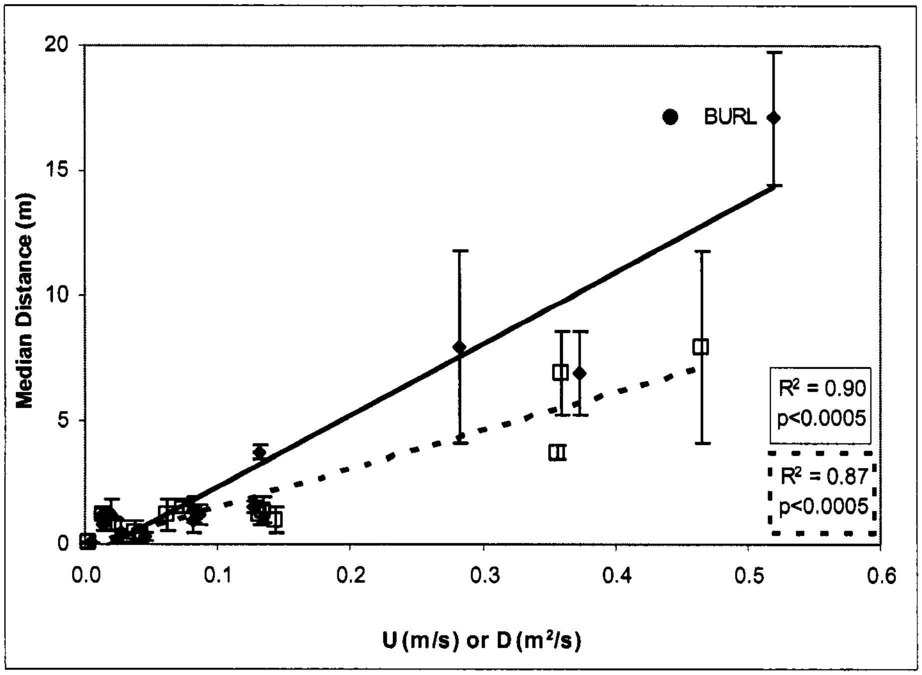

Figure 2-5. Median travel distance for surrogate leaves as a function of solute velocity (diamonds and solid lines) and dispersion (open squares and dashed lines). Bars represent the interquartile range.

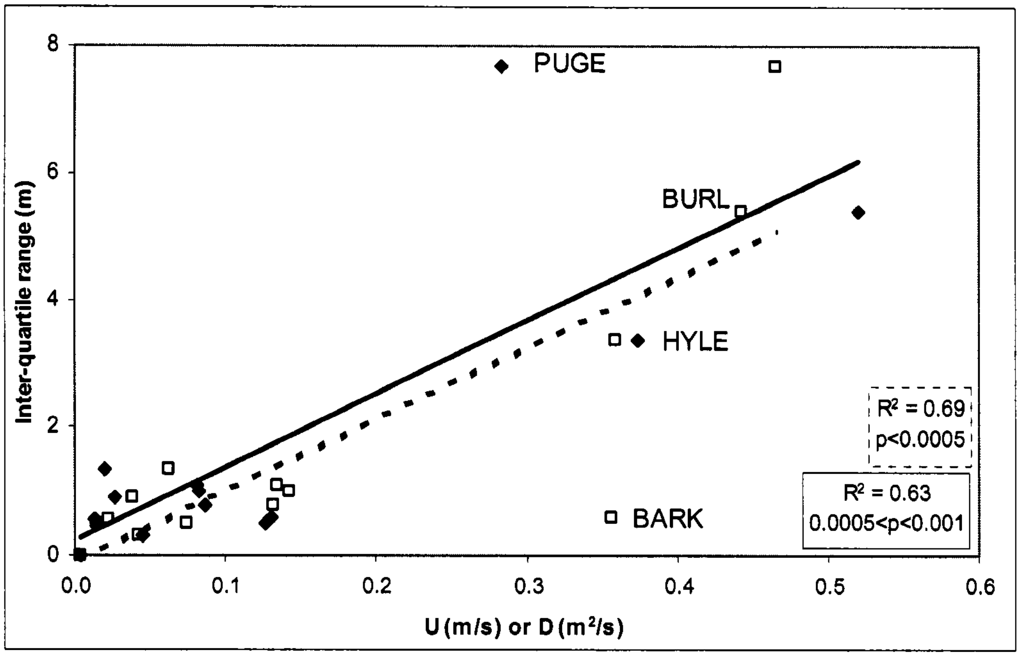

Figure 2-6. Interquartile ranges of distance traveled by surrogate leaves as a function of solute velocity (diamonds and solid lines) and dispersion (open squares and dashed lines).

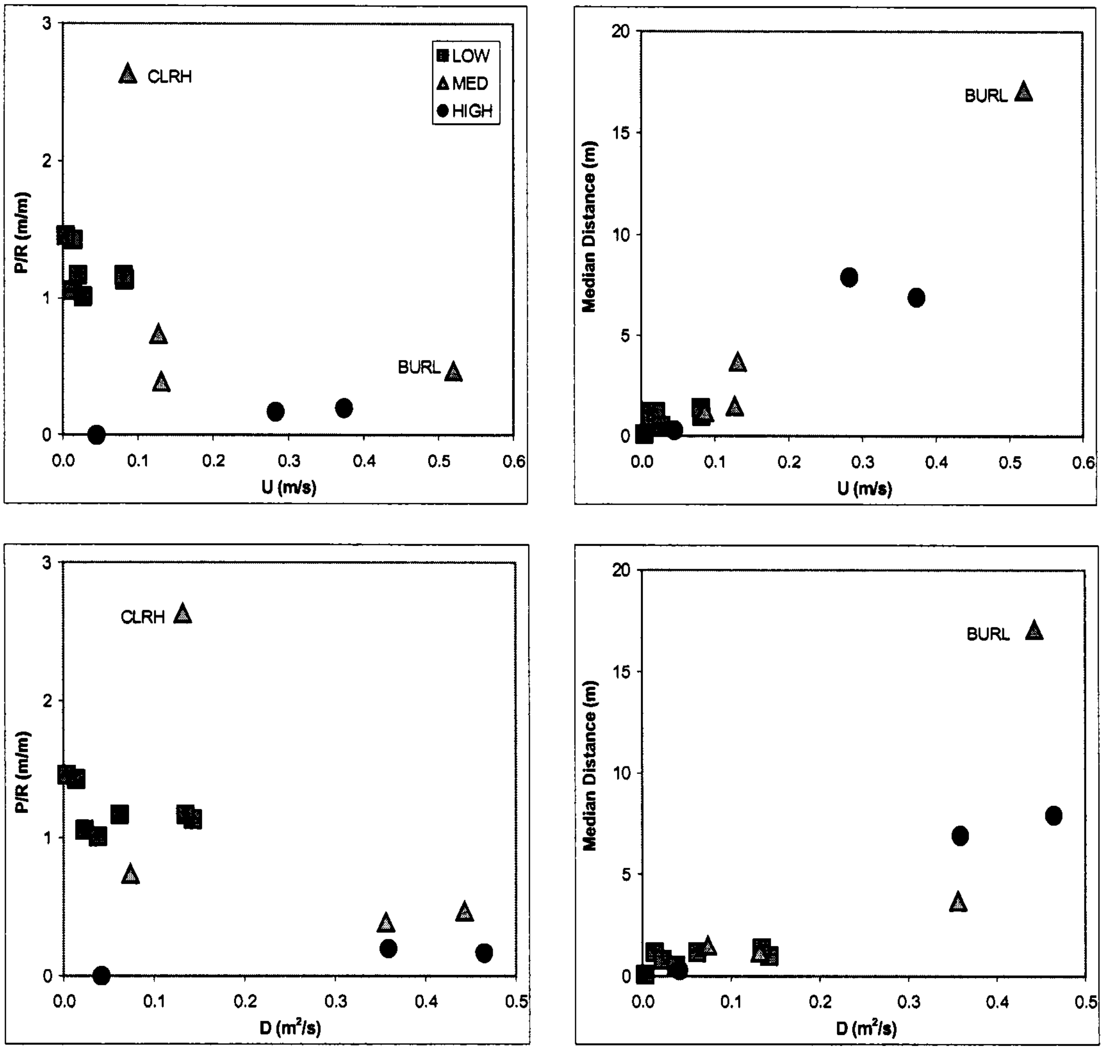

Figure 2-7. Pool/riffle ratio and surrogate leaf transport distance as a function of solute velocity and dispersion by watershed development level.

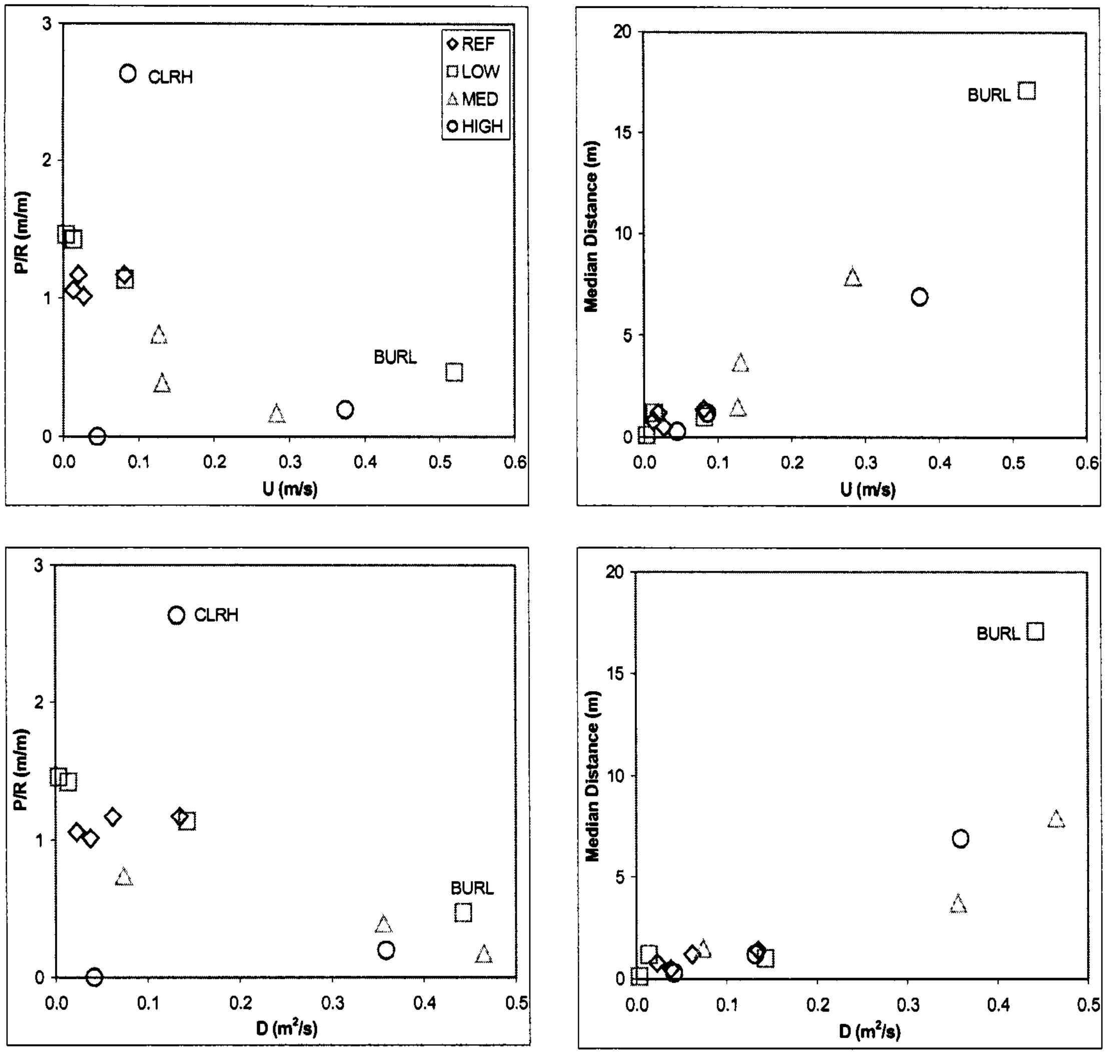

Figure 2-8. Pool/riffle ratio and surrogate leaf transport distance as a function of solute velocity and dispersion by riparian disturbance category.

Table 2-1. Organic matter transport study site characteristics. Riparian vegetation disturbance refers to levels documented in Chapter 1. Land cover is presented as percent developed land (%DEV) and percent forest cover (%FOR), with the remainder consisting of shrubby vegetation, as described in Chapter 1. A is watershed area, Q is discharge, and Q/A is the unit discharge normalized by watershed area. V_{sta} refers to the instantaneous station velocity derived from discharge measurements. P/R is the pool/riffle ratio, presented as both feature length ratios (m/m) and feature area ratios (m^2/m^2). U is the reach-averaged velocity and D is the hydraulic dispersion determined from the tracer study. Surrogate release distances are presented as the median and the interquartile range, or spread between the 25th and 75th percentile travel distances. Positive skewness indicates that the average travel distance exceeds the median.

Station	Riparian Vegetation Disturbance	Watershed			Discharge			Summer Pool/Riffle			Tracer Study		Surrogate Release		
		A (km^2)	%DEV	%FOR	Q (m^3/s)	Q/A ($m^3/s/ha$)	V_{sta} (m/s)	P/R (m/m)	P/R (m^2/m^2)	LWD Jams	U (m/s)	D (m^2/s)	Median (m)	Interquartile Range (m)	Skewness
ANDE	REF	4.2	18	44	0.001	0.02	0.09	1.0	1.6	2	0.03	0.04	0.5	0.9	0.3
LOST	REF	7.3	2	37	0.012	0.17	0.08	1.2	1.3	0	0.08	0.14	1.4	1.1	2.1
SEAR	REF	10.3	5	45	0.002	0.02	0.02	1.1		0	0.01	0.02	0.8	0.6	1.5
WILM	REF	15.8	9	45	0.003	0.02	0.06	1.2	0.9	3	0.02	0.06	1.2	1.4	2.3
BURL	LOW	14.6	18	30	0.270	1.85	0.41	0.5	0.6	6	0.52	0.44	17.1	5.4	0.0
CRES	LOW	11.2	9	48	0.023	0.21	0.15	1.4		5	0.01	0.01	1.2	0.5	0.8
SEAL	LOW	12.7	5	44	0.019	0.15	0.30	1.1		0	0.08	0.14	1.0	1.0	2.3
WILL	LOW	6.4	8	39	0.001	0.01	0.11	1.5		4	0.004	0.003	0.1	0.0	4.7
BARK	MED	4.9	20	32	0.019	0.39	0.15	0.4	0.4	2	0.13	0.36	3.7	0.6	0.3
CLRM	MED	9.7	23	30	0.013	0.14	0.07	0.7		0	0.13	0.07	1.5	0.5	1.4
PUGE	MED	5	60	5	0.024	0.47	0.21	0.2	0.2	0	0.28	0.47	7.9	7.7	1.2
CLRH	HIGH	8.7	23	32	0.009	0.11	0.11	2.6	3.0	0	0.09	0.13	1.2	0.8	-0.3
HYLE	HIGH	18.8	40	20	0.067	0.36	0.27	0.2	0.2	0	0.37	0.36	6.9	3.4	1.9
MEEK	HIGH	14.8	44	8	0.010	0.07	0.08	*		0	0.05	0.04	0.3	0.3	1.3

* Study reach had no riffles.

Chapter 3

Physical Abrasion and the Relative Influence of Urban Development, Floods, and Salmon on Leaf Litter Decomposition in Puget Sound Lowland Streams

Introduction

Humans impact stream ecosystems by altering one or more of five factors: flow regime, water quality, habitat, energy regimes, and biotic interactions (Karr, 1998). Urbanization increases stormwater peak flows and frequency from increased surface runoff and decreased infiltration due to impervious surfaces (Konrad et al., 2005; Booth et al., 2002; Booth, 1991; Schueler, 1987). Activities associated with development may increase concentrations of nutrients (Brett et al., 2005; Pascoal et al., 2005; USGS, 1999; Bryant, 1995), suspended solids (Olthof, 1994; Charbonneau and Kondolf, 1993; Byron and Goldman, 1989), as well as metals and other toxic materials (Beasley and Kneale, 2002; Callender and Rice, 2000; Bryant, 1995; Welch, 1992). Urbanization also degrades physical habitat through stream channel widening, incision, and loss of complexity (Short et al., 2005; McBride, 2001; Booth and Jackson, 1997).

By comparison, fewer studies have evaluated the effect of urbanization on energy regimes and biotic interactions. Urbanization results in biotic shifts generally described as degrading the macroinvertebrate community (May et al., 2000; Morley, 2000). Land cover changes, such as urbanization, forestry, agriculture, residential development, and road construction, affect the quantity and quality of allochthonous inputs to small streams (Chapter 1; Oelbermann and Gordon, 2000; Sponseller and Benfield, 2001; Brown et al., 1983; Stout, 1982; Triska et al., 1982; Webster and Waide, 1982). Urbanization also results in a variety of stressors that directly affect the availability (Chapter 2) and utilization of terrestrial organic matter.

Allochthonous leaf litter inputs represent an important nutrient source to small stream ecosystems (Wallace et al., 1997; Anderson and Sedell, 1979), and thus factors that affect organic matter cycling will influence food webs, nutrient regimes, and ecosystem productivity. Gessner and Chauvet (2002) proposed leaf litter decomposition as an indicator of stream ecosystem function to complement structural and biological measures widely used to assess stream health (Karr and Chu, 1999; Schultheis et al., 1997). Together these functional and structural measures can indicate the relative effects of anthropogenic disturbance (Pascoal et al., 2003).

Leaf breakdown results from a variety of biotic and abiotic processes, including chemical leaching, microbial conditioning, macroinvertebrate feeding, and physical fragmentation (Webster and Benfield, 1986), and processes are not necessarily sequential (Gessner et al., 1999). Comparing relative breakdown rates, and the relative contributions of the underlying processes to those rates, provides a means of comparing ecosystem function in or between disturbed and reference streams (Gessner and Chauvet, 2002).

Urban stream ecology has received attention recently (Meyer et al., 2005) as the extent of human development continues to increase (Walsh et al., 2005; Staubitz et al., 1997). Recently, Chadwick et al. (2006) found red maple breakdown rates peaked when impervious surfaces reached 30% of the watershed area before declining with greater urbanization in small Florida streams. Discharge, insect community, and water column nutrients and metals had the greatest influence on leaf breakdown rates. Meyer et al. (2005) found decomposition increased and nutrient uptake decreased with urbanization, while overall stream metabolism was not related to urban development. Activities associated with anthropogenic disturbance may enhance or slow leaf litter breakdown, and the net effect likely varies by system.

The purpose of this study was to evaluate the effect of urbanization on red alder (*Alnus rubra*) breakdown in small Puget Sound Lowland streams and to quantify the relative influence of physical, chemical, and biological processes. Red alder was selected as the most common riparian deciduous species for which decomposition rates have been developed in other regions of the Pacific Northwest. Three years of experiments provided the additional opportunity to compare stressors over time, and in particular the effects of a large salmon run and two flood events on relative breakdown rates.

Methods

Study area description

The Puget Sound Lowland occupies a low-elevation trough formed by repeated glacial advances, the most recent of which ended about 15,000 years before present. The region is bounded to the east and west by the Cascade and Olympic mountain ranges. The bedrock outcrops of the Green and Gold Mountains rise to 540 m on the Kitsap Peninsula and are surrounded by a vast plain of heterogeneous glacial deposits that include till and glacial outwash. The resulting landscape includes hundreds of low-elevation small streams. Channel substrates generally consist of coarse gravel with some silt-dominated reaches, particularly in low-gradient and urban streams, and streams typically exhibit pool/riffle morphologies.

Whereas Douglas fir (*Pseudotsuga menziesii*) dominates upland areas, a mosaic of conifer and deciduous vegetation occupies riparian zones. Historically, conifers were more prevalent than currently in many riparian areas (Collins and Montgomery, 2002) but were selectively removed during widespread timber harvesting in the late 1800s and early 1900s. More recently, urban residential and commercial development has altered upland and riparian vegetation communities. Specifically, urban development has modified tree species composition or has removed vegetation entirely.

Site selection and watershed characteristics

Four experiments conducted over three years measured breakdown rates in a variety of watersheds. A 2002 pilot study included five stream reaches within the Chico Creek watershed that were selected to represent hydrologic regimes modulated by large lakes, a tributary with extensive bedrock and little natural hydrologic retention, and two mainstem stations with varying riparian cover. Eighteen stations monitored in October 2003 and 2004 were selected to represent a range of forested and developed land cover within small (4 to 24 km^2) Puget Sound lowland watersheds (Figure 3-1). For the November 2003 experiment, eleven stations were selected as a subset of the October 2003 sites, avoiding streams with large salmon runs. Site elevations ranged from 3 to 120 m, while wetted widths during fall low-flow conditions ranged from 1.6 to 4.5 m.

A 10-m DEM (University of Washington, Department of Earth and Space Sciences) was used to delineate watershed areas. All study sites were within 1 km of road or trail access but were not located directly downstream of the access point. Land cover codes from a 1998 LandSat Thematic Mapper image (Hill et al., 2003) were used to determine percent forest cover (conifer and mixed) and percent developed land (high density, low density, and bare ground) for each station. Percent total impervious area from all land cover types within each watershed was estimated using values typical for the Puget Sound Lowland (Dinicola, 1990).

Stream data

Water temperature at each monitoring location was recorded at 30-minute intervals using either Onset StowAway TidBits (±0.2°C at 0 to 30°C) or Dallas Semiconductor Maxim iButtons DS1921L (±1.0°C at -10 to 60°C). Average daily temperature was used to develop heating-degree days. Dissolved oxygen was recorded in 2003 and 2004 using a YSI Model 55 handheld dissolved oxygen meter, while field pH was determined using an Orion Model 230A handheld pH meter. Low ionic strength pH buffers were used, and both meters were calibrated according to manufacturer's recommendations.

Long-term, continuous discharge data are not available at most locations used in this study. Figure 3-2 presents continuous discharge data for the study period for one urban study site (Miller Creek, King County station 42a) and one nearby low-development site (Huge Creek, USGS station 12073500), the only two nearby stations with records >10 years. Instantaneous discharge was measured in other study streams periodically over a three-year period using a Marsh-McBirney (Model 2000) velocity meter and wading rod. Peak discharges tend to occur in the fall and winter, and nearly all precipitation falls as rain. The timing of initial fall storms varies annually, but storm events generally begin in October. In 2004, fall rains did not occur at all, and discharge patterns reflect the lack of large storms. The fall 2002 experiment, conducted only on the Kitsap Peninsula, coincided with a relatively low-flow fall condition. Two storms in October and November 2003 produced the largest and second-largest 24-hour rainfall totals ever recorded at the National Weather Service site at SeaTac Airport (METAR station KSEA), resulting in extensive flooding of the entire study area during two leaf breakdown experiments. The Kitsap Peninsula sites received even greater rainfall amounts based on precipitation at Bremerton, due to the orographic effects of the Green and Gold Mountains.

During those events, discharges at Miller and Huge creeks were among the highest ever recorded at the two gages.

Because flood discharge data are limited, and because the effect of high flows varied with local channel characteristics, the response of each monitoring station to the two floods of 2003 was recorded as the maximum flood stage relative to the bankfull depth. Field observations noted whether peak water elevation was substantially above bankfull depth (>0.3 m, extensive flooding), close to bankfull depth (within 0.3 m, moderate flooding), or substantially below the bankfull depth (minimal flooding), using scour patterns and channel side slope geometry. Sites were visited immediately after both the October 20-21 and November 18-19 (2003) flood events.

Stream benthic health was evaluated using the benthic index of biological integrity (BIBI) for Puget Sound Lowland streams. The BIBI includes ten metrics that describe the diversity, abundance, and tolerance to stressors of the benthic community (Morley, 2000). Benthic invertebrate samples were collected by several academic and governmental organizations in the study streams using standardized protocols, although the methods may have varied somewhat among organizations.

The relative influence of autumn-returning adult chum salmon (*Oncorhynchus keta*) at each monitoring location was recorded in 2003 and 2004. During each leaf pack collection event, the number of spawning salmon or carcasses present within each 100-m study reach was noted as none, few (<10), moderate (10 to 20), or many (>20).

Leaf pack construction and installation

Leaf packs were constructed of freshly senesced red alder leaves collected in September of 2003 and 2004 and air dried to reduce variability in moisture content among individual leaves. Red alder leaf packs consisted of 6 to 9 individual leaves selected to produce an air-dried mass of 3.0 ± 0.05 g in 2003 and 2004. In 2002, red alder leaves collected from the Chico Creek watershed were oven dried at 60°C to constant mass. Smaller leaves were used, and leaf packs were constructed of 13 to 21 leaves with an oven-dried mass of 3.0 to 3.2 g.

Leaf packs were either tethered or enclosed in mesh bags, with mesh sizes summarized by experiment in Table 3-1. Tethered leaf packs were constructed by softening the leaves for no more than 30 minutes in deionized water and securing the petioles with colored electrical tape, on which sample numbers were recorded. Mesh bags, approximately 12 cm by 17 cm, were constructed in an envelope style by stapling or securing rolled edges with nylon ties. Envelopes of fine mesh (0.5 mm) were labeled directly with sample numbers, while coarse-mesh bags (3 to 15 mm, varying by experiment) required samples to be numbered using flagging tape secured to the leaf packs. Monofilament line was tied around the taped petioles or through the mesh envelope to attach the leaf packs to 0.6-m lengths of rebar driven into the streambed.

Fifteen leaf packs were placed at each location within riffles such that packs rested in areas of natural leaf accumulations. In 2002, three leaf packs from both the coarse- and fine-mesh bags were randomly selected and removed on days 2, 7, 14, and 28. Because of the greater variability in organic matter remaining later in the experiment, the sampling protocol was modified in 2003 and 2004 such that two leaf packs were removed on days 2 and 7, three leaf packs on day 14, and four leaf packs on days 28 and 56 or 66 for the tethered and bagged samples.

Construction and installation losses were determined for each experiment. An additional three leaf packs each for tethered, coarse-mesh bags, and fine-mesh bags were constructed and installed at one site then randomly removed immediately following installation. These were analyzed using the same procedures as the remaining leaf packs and construction losses were quantified separately for tethered and bagged samples. In addition, initial leaf pack mass was corrected if leaf pieces >1 mm detached from samples prior to installation.

Laboratory analyses

Leaf packs were randomly selected and carefully lifted from the streambed to avoid rinsing macroinvertebrates from the packs. Leaves were placed in pre-labeled plastic bags and transported to the laboratory in hard-sided containers to prevent further breakage. Packs were stored at 4°C until samples could be processed, generally within 24 hours of sample collection.

Water in the plastic bags was strained through a 0.25-mm screen to catch any leaf particles or macroinvertebrates that were detached from the leaf pack. Leaf packs were disassembled by unwrapping the petioles or by cutting open the envelopes. For tethered samples, the total number of leaves was compared with the original number to verify that no leaves had detached from the pack. Individual leaves were rinsed of silt or other non-leaf material and samples were dried to constant mass at 60°C. Oven-dry mass was recorded for each leaf pack, and then the sample was ashed at 500°C for three to five hours and ash-free dry mass recorded.

Macroinvertebrate counts on the leaf packs were recorded in 2002 and 2004. In 2002, total counts were determined for one coarse-mesh bag (3-mm) from each of the five stations at each collection event. In 2004, total counts were determined for one tethered sample from ten stations and for one coarse-mesh bag (15-mm) at six stations for each collection event. Coarse mesh size was increased in 2004 due to the observed lack of large-bodied macroinvertebrates in 2002. Free water from the plastic sample bags, rinse water from the individual leaves, and both sides of the individual leaves were viewed under an Olympus SZ40 (10X) microscope to ensure that all macroinvertebrates were counted. The insects were not identified to functional feeding group, given that functional feeding groups are not absolute and insects feed opportunistically (Graça, 2001; Anderson and Sedell, 1979; Anderson and Grafius, 1975; Coffman et al., 1971).

Statistical analyses

A first-order exponential decay rate was used to describe organic matter loss over time. Decomposition rates (k) were developed using a best fit to the individual leaf pack results, including estimates of the initial construction losses. Rates were calculated using both time (days, d) and heating-degree days (HDD) as the independent variable. To enable comparison across study sites and for different time periods, decomposition rates were calculated using several exposure time periods ranging from 14 days to 56 days.

Results were compared using a variety of statistical analyses. T-tests or paired t-tests were used to compare two sets of results, while single-factor and two-factor ANOVAs were used for multiple groups. Where differences were significant among groups, Tukey's post-comparison test was used to determine significance ($\alpha \leq 0.05$ for all analyses).

Results

Site characteristics

Land cover data were used to categorize sites as low, medium, and high development. Level of urban development (%DEV) in the study watersheds ranged from <5% to 69% and forest cover (%FOR) ranged from <5% to 56%, with grass and shrub constituting the remaining area (26 to 53%). Watershed forest cover and developed land were inversely related (Figure 3-3; R^2=0.84). Estimates of watershed percent total impervious area (%TIA) ranged from 16% to 36% for the high-development sites, to 7 to 12% for the medium-development sites, and <5% for the low-development sites.

Tethered samples vs. coarse-mesh bags

In the 2004 experiment, both tethered leaves and leaves in coarse-mesh bags were placed at a subset of sites to determine whether the exposure technique affected decomposition rates or macroinvertebrate densities. Decomposition rates decreased between 14 days and 28 days, and then became more variable after 56 days. However, there were no significant differences in decomposition rates between bagged and tethered samples for rates derived from the 56-day datasets (paired t-test, p=0.80 for $k(d^{-1})$ and p=0.87 for $k(HDD^{-1})$). Therefore, decomposition rates using the two exposure techniques were not different.

Decomposition rates varied over time, although differences were not always statistically significant. From Figure 3-4a, when only bagged samples were considered, there was no significant difference in decomposition rates calculated from 14-, 28-, or 56-day datasets (ANOVA, p=0.25). However, tethered samples did exhibit a difference between rates calculated based on the 28- and 56-day datasets (ANOVA, p=0.032). In general, there was little variation in organic matter content within a site, although variability increased over time. Because of the variation in decomposition rates with the length of the dataset, the remaining calculations explicitly indicate the number of days used to develop the decomposition rates.

In situ breakdown rates (2002 salmon and 2003 floods)

The 2002 monitoring stations were located in the Chico Creek watershed, which supports one of the largest chum salmon runs in central Puget Sound (over 70,000 spawners in 2002; T. Ostrom, Suquamish Tribe, personal communication). Leaf packs had been submerged for 15 days when the spawning run began, and by day 28 of the experiment, 50% of the remaining leaf packs (33 over five stations, all mesh sizes) had been destroyed or displaced. The extensive substrate movement induced by salmon digging the streambed to build redds enhanced physical abrasion. In many cases, salmon snapped the monofilament line (50-lb test) securing the leaf pack to the streambed, and the leaf pack was transported away from the site. In other cases, the leaf pack burst open and the contents were lost, or the entire anchoring rebar, buried up to 0.5-m deep, was undermined and transported downstream.

Physical leaching in the first two days of the 2002 experiment resulted in mass losses of 15 to 19%, in addition to a construction mass loss of 6% (Figure 3-5). Microbial and macroinvertebrate processing contributed an additional 10% mass loss through day 14. In the fine-mesh bags, 10 to 20% mass losses occurred between day 14 and day 28. However, in the coarse-mesh bags, 25 to 35% mass losses occurred over the same period. Because both fine- and coarse-mesh bags were randomly placed in the same riffle and in the same zone of natural leaf accumulation, the difference in mass loss between coarse-mesh bags and fine-mesh bags was due primarily to physical abrasion by salmon. Decomposition rates did not include leaf packs that were destroyed by salmon; therefore, the calculated rates underestimated the effect of abrasion caused by spawning.

Breakdown rates did not vary significantly with time in the 2002 experiment. While the decomposition rates decreased slightly for the fine-mesh bags and increased for the coarse-mesh bags between days 14 and 28, the low number of stations and limited sample pairs remaining at day 28 were insufficient to statistically distinguish the rates (Figure 3-4d; two-factor ANOVA with no replication, p=0.21 and 0.11, respectively for $k(d^{-1})$).

Three of the 2002 stations were monitored in subsequent years using varying exposure techniques, enabling the comparison of decomposition rates during different years. While October 2003 decomposition rates appeared higher than in 2002 for the 14-day decomposition rates (Figure 3-4e), the differences were not significant (single-factor ANOVA, p=0.052). The limited number of monitoring sites included in the 2002 program precluded a statistical comparison of the relative influence of salmon on decomposition rates in 2002 compared with other years not affected by the large salmon run. Nonetheless, the dramatic impact salmon had on the leaf packs deployed in 2002 indicates that they can play a major role in leaf breakdown in spawning habitats.

The initial 2003 experiment, conducted in October using tethered leaf packs placed at 18 stations, terminated at most locations following the flooding. Prior to the storm, construction losses (shown in Figure 3-6 as day 0 results) totaled 4% of the original organic matter of the leaf packs. Short-term leaching produced an additional 10 to 17% mass loss. The October 2003 flood resulted in tremendous organic matter losses in the tethered red alder leaf packs. Incremental losses varied by site from 19 to 63% of the original organic matter at those sites where leaf packs survived the flood. Sufficient organic matter survived the flood to continue the experiment at only four of the 18 locations, of which three were in high-development watersheds.

Because leaf packs at three high-development sites and one medium-development site survived the flood, the decomposition rate appeared inversely related to development. However, the low-development watersheds have headwaters in the Green and Gold Mountains, which received higher rainfall than the watersheds of the developed sites. Therefore, the low-development sites likely experienced far greater flood flows than the other locations. In addition, hydrology at two of the high-development sites was groundwater dominated. Most stormwater in one watershed (PUGE) bypassed the stream and discharged directly to Puget Sound through stormwater pipes, and the other watershed (PONC) is surrounded by extensive sand and gravel deposits with very high permeability. The apparent pattern is thus an artifact of the geographic structure of the experimental design.

The hydrologic response to the October 2003 flood event is illustrated in Figure 3-7. Decomposition rates (Figure 3-4f) were lower at the two sites where minimal hydrologic response occurred. The difference by hydrologic response was significant for k(HDD^{-1}) but not for k(d^{-1}) ($p=0.02$ and 0.08, respectively). To estimate the actual mass loss during the storm for the 11 of 18 sites with non-zero organic matter remaining, percent organic matter at day 18 was estimated as the mean of the leaf packs at day 14 minus the estimated mass loss using k(d^{-1}) from day 14 to day 18. Using the decay coefficients calculated from the first 14 days of exposure, mass loss between day 14 and day 18 was minor (6 to 9% of the original organic matter). Therefore, the mass loss between day 14 and day 28 is nearly entirely due to physical abrasion from the flood. The storm alone produced a mean organic matter mass loss of 40%, and individual site losses ranged from 15% to 56%.

In the November 2003 experiment (Figure 3-7), construction losses were higher at 10% and the day 2 results exhibited greater variability among sites than the October 2003 experiment, likely because the materials had been handled more.

The subsequent November storm produced floods that were not as intense as the October floods but resulted in high organic matter loss at several sites. Because the flood event occurred on a sample collection day and because some flows peaked earlier than others, some sites were sampled prior to the flood and others following the flood. Therefore, conditions represented by the day 14 samples are not homogeneous.

The November 2003 decomposition patterns reflected the hydrologic response of each station to the storm event, and leaf pack masses remaining after the flood were not related to watershed development level ($R^2<0.2$). Sites that experienced minimal response to the flood event had the highest organic matter retention at day 28, while those with extensive flooding experienced the greatest organic matter loss, presumably due to abrasion (Figure 3-7). However, decomposition rates calculated from the 28-day datasets did not vary significantly by hydrologic response (Figure 3-4g; single-factor ANOVA, $p=0.13$ for $k(d^{-1})$ and $p=0.054$ for $k(HDD^{-1})$). Sites exhibited greater variability in organic matter response than in the October experiment.

Five of the stations were accessible before or at the beginning of the storm, and the average mass loss between days 2 and 14 was 6%, attributed to the combined effect of microbial processes and macroinvertebrate feeding. The remaining six stations were inaccessible because of high flows and could not be sampled until after the stage receded. Average mass loss across these six stations was 35%. Accounting for the microbial and macroinvertebrate losses from the other sites, physical abrasion during the November 2003 flood caused a 29% mass loss in the sites with the most extensive flooding. Total mass loss between days 2 and 28 averaged 44% across all eleven stations, which included the effects of microbial and macroinvertebrate processes as well as physical abrasion through day 14 and afterwards.

The actual loss during the November 2003 storm was estimated based on the average organic matter remaining at day 28 minus the estimated pre-storm organic matter content. The latter was estimated using k(d^{-1}) calculated from the 7-day datasets, since some of the day 14 samples were collected before the storm, some after, and some during. The November 2003 storm and subsequent flood caused mass losses similar to those in the October 2003 storm, with an average of 29% and specific sites ranged from 3% to 63% mass loss. The mass loss at sites with minimal hydrologic response was significantly less than at sites with extensive flooding (single-factor ANOVA, p=0.033).

While salmon were present in some of the streams during the November experiment, the effect of salmon was likely secondary to the physical abrasion induced by the flood event. Differences in decomposition rates among salmon abundance classes were not significant (Figure 3-4h; single-factor ANOVA, p=0.10 and 0.13 for k(d^{-1}) and k(HDD^{-1}), respectively). In general, fewer salmon spawned in the study streams in 2004 compared with 2002, when salmon-induced abrasion produced extensive mass loss.

Decomposition rates for the eight stations monitored in each of the October 2003, November 2003, and 2004 experiments were compared using a two-factor ANOVA without replication, with the years representing blocks. There was a statistically significant difference among experiments (Figure 3-4i; p=0.0005 for k(d^{-1}) and p=0.002 for k(HDD^{-1})). Decomposition rates were significantly higher in the October experiment than in the November experiment because the storm occurred later in the study period when less organic matter was available. While the more intense October 2003 storm resulted in similar overall mass loss, decomposition rates were greater than in the November 2003 storm, partly due to the timing of the storm in the experiment. Therefore, the effect of storms on decomposition rates due to physical abrasion varied not only with intensity but also with timing.

In situ breakdown rates (2004 development)

In 2004, when no floods or salmon influenced decomposition, rates varied significantly with watershed development level after 28 days of exposure. The experiment took place during a particularly dry fall period. Three small storm pulses affected the eastern study area, but peak discharge was well below peak storm discharge in 2002 and 2003. Variability among sites was low after day 2 and day 7, but beginning at day 14, sites exhibited greater variability as mass loss rates increased in high-development watersheds (Figure 3-8). At day 14 and day 28, low-development sites retained more organic matter as compared with medium- and high-development sites (single-factor ANOVA, p=0.014 and 0.012 for day 14 and day 28, respectively).

After 28 days of exposure, decomposition rates were lower at the low-development sites than the high-development sites. These initial rates varied significantly (Figure 3-4b; p=0.002 for $k(d^{-1})$ and p=0.03 for $k(HDD^{-1})$). However, when the decomposition rates were calculated using the 56-day datasets, there was no statistically significant difference in decomposition rates among watershed development levels (Figure 3-4c; p=0.99 for $k(d^{-1})$ and p=0.82 for $k(HDD^{-1})$). The magnitude and variability of decomposition rates in the medium- and low-development watersheds increased when calculated with the longer exposure time, possibly due to the increased likelihood of disturbances over the longer record or because biotic processes accelerated. After 56 days, 6% to 36% organic matter remained with no apparent relationship to development level.

In general, using HDD instead of time in days did not change the findings. HDD after 28 days increased with increasing watershed %TIA (R^2=0.86) and decreased with increasing %FOR (R^2=0.71), omitting one site strongly influenced by a warm, shallow lake (KITS). Although warmer streams occurred in more developed watersheds, temperature effects alone did not explain differences in decomposition rates with watershed development level.

Other stream characteristics did not vary from site to site. The pH in study steams remained circumneutral (6.2 to 7.9) with little variability among sites. Dissolved oxygen (DO) levels also did not vary among sites, and no streams exhibited low levels. Therefore, neither extreme pH nor DO levels occurred that would have influenced breakdown rates.

Macroinvertebrates

BIBI scores (Figure 3-9) followed patterns typical of Puget Sound Lowland streams, with some degraded sites even at low %TIA but no good-condition sites in areas with high %TIA (Booth et al., 2004). Good conditions are defined as those with BIBI scores ≥38 while poor conditions have BIBI values ≤26 (Morley, 2000). BIBI scores averaged 41 ("good") for low-development sites, 33 ("fair") for medium-development sites, and 19 ("poor") for high-development sites, with better conditions generally found in less-developed watersheds.

During the 2004 decomposition experiment, macroinvertebrate counts and densities were greater in the mesh bags than in the tethered samples (paired t-test, p=0.0007 for counts, p=0.0024 for densities), possibly due to enhanced sheltering from the bags, although the differences were not significant until day 28. Peak counts and densities occurred after 28 days (Figure 3-10a-d), coincident with the peak in leaf pack nutrient levels (Appendix A).

Macroinvertebrates in bagged samples were less abundant in more-developed watersheds. Macroinvertebrate counts varied significantly with time and development level (two-factor ANOVA, p=0.0004 and 0.037, respectively), and the factors were independent (interaction p=0.64). However, when counts were normalized by the organic matter mass remaining in the leaf packs, macroinvertebrate density varied significantly with time (p=0.002) but not with development level (p=0.077), and the factors were independent (p=0.87) in bagged samples.

For tethered samples, counts and densities were lower in samples from the most highly developed watersheds compared with those from low- and medium-development watersheds. Counts did not vary significantly with time (p=0.067) or development level (p=0.25), although densities varied with time (p=0.05) but not development level (p=0.29).

In the 2002 experiment, macroinvertebrate counts peaked at day 14 and decreased by day 28, following the large salmon run (Figure 3-10e-f). After 2, 7, and 14 days of exposure, macroinvertebrate counts on the leaf packs were higher in 2002 than in the 2004 experiment, indicating high interannual variability.

Discussion

Physical abrasion from floods and salmon

Physical abrasion has not been quantified previously as an important mass loss mechanism for organic matter processing in small streams. However, in the Puget Sound Lowland, disturbances such as floods, salmon, and urban development produced high mass loss in red alder leaves by enhancing physical abrasion, and microbial and macroinvertebrate processing were secondary loss mechanisms. Further, while changes in hydrology and geomorphology associated with urban development enhanced decomposition, intense floods produced the highest abrasion losses as episodic mass losses.

Experiments over a three-year period provided an opportunity to compare breakdown among sites subject to different environmental disturbances. The two unanticipated disturbance mechanisms, floods and salmon, fortuitously occurred before the effects of watershed development could be assessed. The combined results of the four experiments offer the unique opportunity to compare the relative contributions of three distinct physical abrasion processes (floods, salmon, and urbanization). Study results also offered the unique opportunity to compare physical abrasion losses, not extensively quantified prior to this study, to mass losses from leaching, microbial processing, and macroinvertebrate feeding. Physical abrasion from the October 2003 flood produced the highest overall mass loss within 28 days, while the absence of floods or salmon in 2004 produced the lowest overall mass loss.

Most comparisons to previous studies rely on first-order decomposition rates. However, absolute rates may be influenced by method differences (Webster and Benfield, 1986) that confound comparisons. In the present study, first-order decay models do not necessarily represent the episodic mass losses induced by physical abrasion well. Therefore, ass loss by primary process provides a means of summarizing and comparing studies that complements traditional approaches using decay coefficients. Table 3-2 summarizes red alder mass losses by process for each of the four experiments.

Leaching from red alder leaves occurs primarily within the first 24 hours, and leaching was the dominant process responsible for in situ mass loss on day 2. Mass loss due to leaching did not vary among sites or among years and averaged 11 to 18%.

The 2002 experiment utilized both fine- and coarse-mesh bags to distinguish the contributions of microbial processes from macroinvertebrate feeding and physical abrasion. No storms occurred prior to the day 14 sample collection in 2002 and mass loss in the coarse-mesh bags represented the combined effects of macroinvertebrate feeding and microbial processes. Macroinvertebrate contributions to leaf breakdown were estimated from the difference in mass loss between the fine- and coarse-mesh bags. Incremental mass loss from microbial breakdown measured by the fine-mesh bags increased between days 14 and 28 from 6 to 13%, possibly enhanced by increased nutrient levels caused by salmon carcasses (Yanai and Kochi, 2005; Ito, 2003). Mass loss was greater in the coarse-mesh bags over the same period. The extensive reworking of the substrate as the salmon built redds after day 14 coincided with low macroinvertebrate densities on the leaf packs by day 28 (Figure 3-10f), and macroinvertebrates likely contributed very little to breakdown between days 14 and 28. Physical abrasion, therefore, dominated decomposition in the coarse-mesh bags between days 14 and 28.

In both 2003 experiments, only tethered leaves were used and it was not possible to distinguish between microbial and macroinvertebrate processes. No storms occurred prior to the day 14 sample collection in the October 2003 experiment, and mass loss was attributed to the combined effects of microbial processes and macroinvertebrate feeding. However, the October storm reduced organic matter content to 3 to 8% of the original mass at seven stations, completely destroyed the installations at another seven stations, and left only four stations with >10% of the original organic matter intact. Macroinvertebrates were not counted; however, field observations noted a distinct lack of macroinvertebrates following the flood event either on the leaf packs or on the channel substrate. Therefore, abiotic physical abrasion was almost certainly the dominant loss mechanism. The four sites that retained significant organic matter after the storm lost additional mass by day 56, but specific process contributions could not be quantified and the mass loss mechanism likely included the combined effects of microbial processes, macroinvertebrate feeding, and physical abrasion from the subsequent November 2003 storm.

The timing of the flood complicated the method of assigning incremental mass losses to a dominant process for the November 2003 experiment since the flood coincided with day 14 sample collection. The sites sampled prior to the November storm retained more organic matter at day 28 than those that could not be accessed during the storm, but after 66 days all sites retained an average of 9%, ranging from 3 to 28%, of the original organic matter mass. Thus, those sites with less physical abrasion experienced greater biological losses due to the combined effects of microbial processes and macroinvertebrate feeding, because more organic matter was available compared to the sites where the floods abraded much of the leaf mass.

Low storm intensity during the 2004 experiment produced low physical abrasion. Therefore, all non-leaching losses were attributed to the combined effect of microbial and macroinvertebrate processes. The combined loss was higher in 2004 (32%) than in previous experiments (15 to 22%) likely because the macroinvertebrate community was not affected by the disturbances that had enhanced physical abrasion in previous years.

The physical patterns in the leaf surfaces as mass was lost from the packs provided additional evidence for the role of physical abrasion in the various experiments (Figure 3-11). Prior to the flood events or salmon spawning, red alder leaves initially thinned but remained intact as soluble components leached from the leaves. Darkened areas grew as leaching and microbial colonization, presumably by fungi, continued. Macroinvertebrate feeding resulted in a lacy pattern on the leaf surface, with small holes in the lamina growing to larger areas where only the veins remained. Completely skeletonized leaves occurred in streams with low velocity. In contrast, abiotic physical abrasion was indicated by blunting, initially at the leaf tips and margins. Following the October 2003 flood event, very little apart from the petioles remained of the leaf structure.

In summary, physical abrasion during flood events induced tremendous organic matter loss that exceeded mass loss from microbial and macroinvertebrate processes. Spawning salmon also enhanced physical abrasion via substrate disturbance such that mass losses were similar in magnitude to the combined effect of microbial and macroinvertebrate processes. When neither floods nor salmon affected the study area, the relative influence of physical abrasion decreased and that of microbial and macroinvertebrate processes increased. However, both physical abrasion processes, particularly floods and salmon, occur frequently in autumn in the Puget Sound Lowland, and physical abrasion likely is a regionally important organic matter loss mechanism not reported in other regions. While the various processes lead to similar mass loss rates, the form of nutrients delivered varies. For example, physical abrasion produces particulate organic matter, while leaching produces dissolved constituents. Therefore, it is important to distinguish which of the biotic and abiotic mass loss processes dominates.

Decomposition rates and watershed development levels

In the absence of floods and salmon, red alder decomposition rate increased as development increased. In the 2004 experiment, leaf breakdown rates calculated from 28 days of exposure varied significantly with the watershed development level, even accounting for differences in water temperature. Decomposition rates were slower in the low-development watersheds ($k = 0.014$ to $0.020\ d^{-1}$) as compared with the medium- ($k = 0.024$ to $0.035\ d^{-1}$) and high-development watersheds ($k = 0.023$ to $0.053\ d^{-1}$), which were not significantly different from each other.

Within the high-development watersheds, the highest breakdown rate occurred where riparian vegetation consists of grasses with few sapling willows and allochthonous inputs were nearly absent (Chapter 1). Skeletonized leaves indicated macroinvertebrate feeding occurred, and the counts confirmed the presence of macroinvertebrates. Macrophytes along the channel margins at these sites likely maintained a high shredder population that utilized the experimental leaf packs when they were available. Benfield et al. (2001) and Webster and Waide (1982) proposed that experimental leaf packs represent food islands in streams where allochthonous inputs were drastically reduced, and breakdown rates were faster in these systems compared to others where detrital food resources are not limited. Removal of riparian vegetation during development may produce shifts to autochthonous-based food webs where detrital food resources are scarce, paralleling the effects of logging without a riparian buffer (Bilby and Bisson, 1992).

The lowest decomposition rate for the high-development watersheds occurred where the experimental and natural leaf packs were routinely covered with fine sediments even at the low flows that occurred in fall 2004. Buried leaf packs decompose more slowly than surface-exposed leaf packs (Chaffin et al., 2005; Pascoal et al., 2005; Tillman et al., 2003; Rulík et al., 2001; Webster and Waide, 1982), since the sediment physically blocks access by macroinvertebrates and affects microbial processes by creating anoxic microenvironments. Burial also shelters leaves from physical abrasion. Increased sedimentation or shifts to fine-grained substrate have been noted as potential effects of urbanization (Short et al., 2005), and differences in sedimentation rates among streams somewhat confounded the results.

Despite the wide range of watershed and site characteristics, there was no statistically significant difference after 56 days in decomposition rates with watershed development level. The magnitude and variability of the rates in the medium- and low-development watersheds increased to the range found in the high-development watersheds (Figure 3-4b-c), while rates within the high-development watersheds remained consistent between 28 days and 56 days of exposure.

At some sites, the breakdown rate increased over time from 0.02 d^{-1} to 0.04 to 0.055 d^{-1}, possibly stimulated by the physical and biological processes accompanying the spawning and subsequent decomposition of salmon. First, the extensive substrate disturbance by salmon during spawning increased physical abrasion of organic matter. Decomposing salmon carcasses increase water nutrient levels (Drake et al., 2005; Yanai and Kochi, 2005; Minakawa and Gara, 1999) potentially enhancing algal and microbial production in streams with nutrient limitation. In addition, the presence of salmon enhances decomposition by stimulating both microbial and macroinvertebrate processes (Yanai and Kochi, 2005; Ito, 2003). Higher primary and microbial production can contribute to higher macroinvertebrate populations. Therefore, salmon potentially increase decomposition rates due to all three major processes: physical abrasion, microbial processing, and macroinvertebrate feeding. However, two sites with elevated salmon numbers did not have high decomposition rates, so the presence or absence of salmon was not the dominant factor controlling decomposition across all sites in this study. The net effect of salmon and development on decomposition rates varies with location and over time.

Traditionally, macroinvertebrate counts and densities are used to indicate macroinvertebrate processing rate (Petersen and Cummins, 1974). Macroinvertebrates use natural leaf packs for both food and shelter (Pascoal et al., 2003; Richardson, 1992). In 2004, when no large-scale disturbance events affected decomposition processes, macroinvertebrate abundance and density peaked at 28 days after which both measures decreased. The macroinvertebrate peak coincided with peak leaf pack nitrogen concentration (Appendix A), consistent with the findings of Anderson and Sedell (1979). In 2002, counts and densities were higher during the first 14 days than in the 2004 experiment, possibly because the 2002 pilot study was limited to low-development sites with healthy macroinvertebrate communities (Figure 3-9).

However, the substrate disturbance from the salmon likely affected macroinvertebrate communities (Walter et al., 2006; Minakawa, 1997) following extensive spawning in 2002, and counts decreased from day 14 values.

Macroinvertebrate counts were higher, but mass loss attributed to macroinvertebrates was lower, in 2002 than 2004. In addition, macroinvertebrate counts and densities were higher in the bagged samples compared with the tethered samples, although there was no significant difference in decomposition rates between tethered and bagged samples. The bagged samples may have provided a more sheltered environment than the tethered samples. Comparison between the two years suggests that macroinvertebrate counts may not adequately represent macroinvertebrate processing either because counts do not characterize processing differences due to body size or because insects may use the leaf packs for shelter as well as food resources. For example, higher macroinvertebrate counts in 2002 were not associated with larger organic matter mass loss due to macroinvertebrate processing.

In summary, watershed development increased red alder decomposition rates after 28 days, with higher decomposition rates found in medium- and high-development watersheds than in low-development watersheds. However, after 56 days, decomposition rates in the low-development watersheds increased to rates in the medium- and high-development watersheds such that there were no statistically significant differences. Urbanization influences a variety of factors that in turn affect decomposition, including discharge, temperature, limited food availability, sedimentation, macroinvertebrate community, and salmon spawning. Individually, factors may enhance or decrease decomposition rates, but the combined effect of urbanization in small Puget Sound Lowland streams enhanced the short-term decomposition of red alder leaves.

Comments on using tethered leaves vs. mesh bags

Most leaf breakdown studies use mesh bags due to ease of handling. However, some contend that leaves enclosed in mesh bags create an artificial environment where water velocity decreases and dissolved oxygen levels may affect nutrient processes (Triska and Sedell, 1976). Because no large-bodied macroinvertebrates were found in coarse-mesh bag (3-mm) leaf packs during the 2002 pilot study, an alternative exposure technique was developed to avoid the use of mesh bags altogether, which could reduce physical abrasion, restrict macroinvertebrate access, and alter microbial processes.

One criticism of an alternative approach, tethered samples, is the potential loss of whole leaves or large segments from the packs. However, counting leaves before and after exposure illustrated that fewer than 0.2% of leaf packs were affected by this confounding loss mechanism. Photographs of each leaf pack in 2004 indicated site-to-site differences in dominant loss mechanisms. Macroinvertebrate feeding produced a lacy structure, while physical abrasion blunted the edges. Sharp-edged breakage patterns occurred infrequently.

While no statistically significant differences were found in breakdown rates determined using tethered samples and coarse-mesh bags (15-mm), using tethered leaves avoided potentially confounding factors from artificial microenvironments. Red alder has a particularly strong petiole that can be secured using waterproof tape. While highly successful for red alder, the method may not be sufficient for all leaf species.

Comparison to other studies

Breakdown rates in low-development watersheds without floods or extensive salmon runs (0.014 to 0.020 d^{-1}) were similar to those found in other studies in the Pacific Northwest. Sedell et al. (1975) found a breakdown rate of 0.017 d^{-1}

in Mack Creek, a 6.5-km^2 watershed in the Cascade Mountains in Oregon. Breakdown rates calculated from data presented in Triska and Sedell (1976) for a 0.002-km^2 watershed were slower (k~0.007 d^{-1}). Both of these studies used much longer exposure time (150 to 250 days) than in the present study. Triska et al. (1982) report a range of 0.015 to 0.018 d^{-1} for red alder in Mack Creek. Breakdown rates were higher in medium- and high-development watersheds (0.023 to 0.053 d^{-1}) or where floods (0.10 to 0.14 d^{-1}) or salmon (0.04 to 0.055 d^{-1}) increased rates compared with these three studies, conducted in more natural settings.

The relative contribution of the various physical, chemical, and biological processes represents another useful comparison with other studies. However, results cannot be compared in an absolute sense to the values in the literature, given the differences among leaf species, leaf preparation, and study areas. Leaching losses were similar to the 19 to 20% mass losses of red alder in small Pacific Northwest streams (Triska et al., 1982; Triska and Sedell, 1976). The combined effects of microbial processing and macroinvertebrate feeding produced higher mass losses in this study than estimated for red alder in southwest Washington (Triska and Sedell, 1976). Microbial and macroinvertebrate processing in the 2002 and 2003 experiments were consistent with mass losses after 56 days in Triska and Sedell (1976). In previous studies from other regions where microbial and macroinvertebrate contributions have been estimated, biological processes account for 40 to 50% mass losses (Cummins et al., 1973; Petersen and Cummins, 1974) in addition to 10 to 15% mass losses due to leaching, although those studies did not include red alder. Microbial and macroinvertebrate processes produced far less mass loss in Puget Sound Lowland streams as compared with other studies reporting this information. Physical abrasion appears to be a regionally important process and reduces decomposition due to microbial and macroinvertebrate contributions.

Figure 3-12 summarizes the range of red alder breakdown rates in Puget Sound Lowland streams affected by flood events, salmon, or watershed development. Except for the rates found for low-development streams in 2004 when no floods or extensive salmon runs occurred, all other decomposition rates from this study were higher than previously reported for red alder, which has some of the highest documented leaf breakdown rates reported. Only one other study quantified the effect of a flood event. Paul and Meyer (1996) inadvertently captured the effect of an annual flood event on rhododendron breakdown, although the organic matter mass loss was not as high as found in the present study for red alder. This is the first study to quantify decomposition rates that resulted from salmon-induced physical abrasion, and also the first to quantify the effect of extreme floods on organic matter processing.

Factors influencing breakdown as urbanization increases

Table 3-3 presents the relative influence of various factors on leaf breakdown rates and whether the factors would enhance or slow breakdown. Relative magnitudes are based on the four experiments conducted over three years in small Puget Sound Lowland streams subject to a variety of disturbances, including floods, salmon, and urbanization. The net effect of the factors will vary regionally but were relatively consistent within study streams for a given condition.

Physical abrasion represents an important leaf breakdown process in Puget Sound Lowland streams, given the coincident timing of fall storms, salmon runs, and peak litterfall inputs. The connectivity and extent of impervious surfaces increases with overall urbanization, producing more frequent and intense high-flow events (Konrad et al., 2005; Booth et al., 2002). This study indicated that increasing hydrologic response to storms accelerated breakdown due to the combined effect of water turbulence and substrate mobilization. Exceptions occurred, however, where stormwater infrastructure bypassed high discharges to downstream waters or where highly permeable soils minimized hydrologic response. In addition, high-flow events mobilize sediment, which can build up on physical obstructions where organic matter accumulates. Decomposition rates decrease when leaf packs are buried relative to decomposition rates for leaf material exposed to surface water processes.

The study streams did not exhibit large differences in pH and DO, and this effect on breakdown was not discernible. The lack of complete nutrient data precluded evaluation of breakdown rate influences. In 2004, HDD after 28 days increased with watershed %TIA and decreased with watershed %FOR. Therefore, instream temperatures reflected land cover characteristics. In general, however, accounting for temperature differences using HDD in decomposition rates did not change the overall findings of the influence of land cover, indicating that factors in addition to stream temperatures were important.

Urbanization alters the benthic community, reducing abundance of intolerant or long-lived taxa that may be responsible for leaf processing (Morse et al., 2003). Macroinvertebrates were found in the most urban sites, although these sites exhibited lower abundance (Figure 3-10) than in the low-development watersheds. Abundance may not indicate the relative level of macroinvertebrate processing, however. In general, microbial processing and macroinvertebrate feeding were less important here than found in previous studies, and the benthic community impact on organic matter processing may be secondary to the abiotic impacts of urbanization.

While the breakdown rate is commonly used to compare results among studies, the relative contributions of the various processes, or percent mass loss, provides an additional means to interpret results. Both breakdown rate and process contributions determine the magnitude and form of nutrients delivered to small streams. Leaching produces dissolved constituents delivered over time scales of hours to days. Microbial processes soften leaf tissue and form a nutrient-rich biofilm, which could slough as the leaf tissue breaks down or provide food for macroinvertebrates. Macroinvertebrates feeding on leaves release dissolved waste products as well as produce particulate organic matter in the form of fecal pellets and leaf fragments through inefficient feeding strategies. Both microbial and macroinvertebrate processes tend to occur steadily over time scales of weeks. Physical abrasion, however, represents an episodic process that may not occur regularly and that may not be described well by traditional exponential decay models. Therefore, the relative process contributions indicate how organic matter nutrients are delivered and in what form, while breakdown rates describe the timing.

Physical abrasion breaks up the leaf/biofilm complex and delivers nutrients in the form of particulate organic matter. During storms, high flows and transported sediment pulverize leaf litter quickly and transport it from the system. As a result, less leaf litter remains within the active stream channel. Benthic organic matter provides habitat, food and long-term nutrient sources to the stream biota. Increased physical abrasion, then, could decrease organic matter storage such that food resources are limited during late winter or early spring, when high flows occur. The loss of organic matter storage due to floods would be especially acute in developed watersheds where high flows and floods are more frequent and severe, channel simplification to improve water conveyance has reduced in-channel storage sites, and isolation of the channel from the floodplain has eliminated the opportunity for storage of organic matter in off-channel sites. Thus, urbanization alters a variety of processes, including leaf breakdown, which could change the availability of organic matter to support stream functions.

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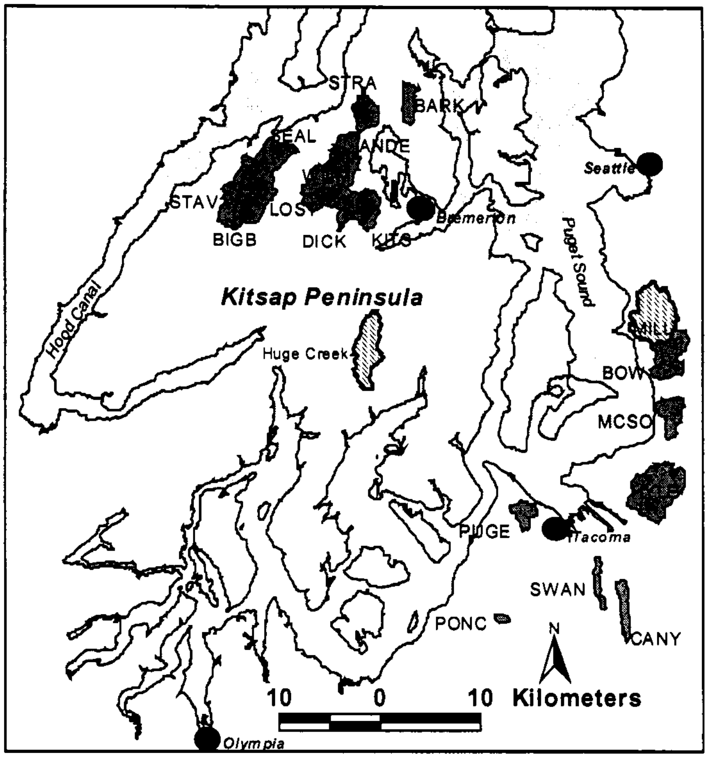

Figure 3-1. Study locations for 2002 through 2004 red alder breakdown experiments. Hatched watersheds indicate long-term discharge monitoring sites.

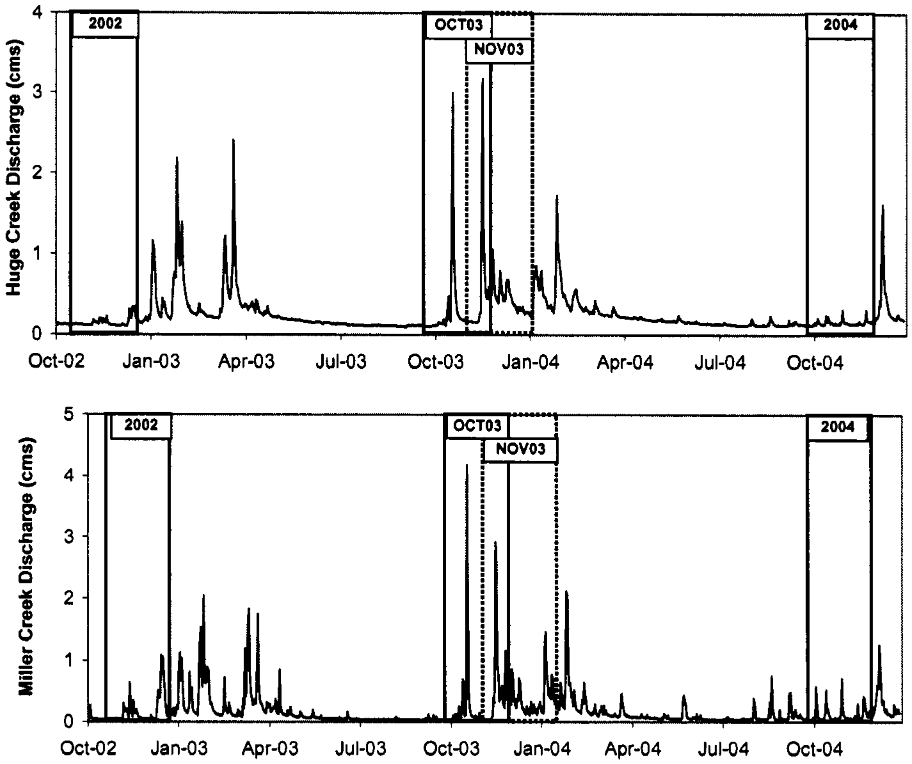

Figure 3-2. Discharge at two long-term gaging stations during the study period. Source: USGS station 12073500 (Huge Creek, 16.8 km^2, 0.17 to 0.57 m^3/s average annual discharge, rural watershed) and King County Hydrologic Information Center station 42a (Miller Creek, 22.7 km^2, 0.11 to 0.40 m^3/s average annual discharge, urban watershed).

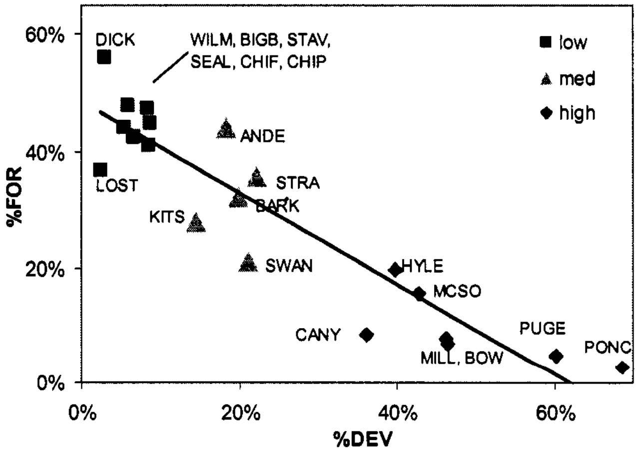

Figure 3-3. Study site watershed development level (high density, low density, and bare ground) and forest cover (conifer forest, mixed forest) relationships.

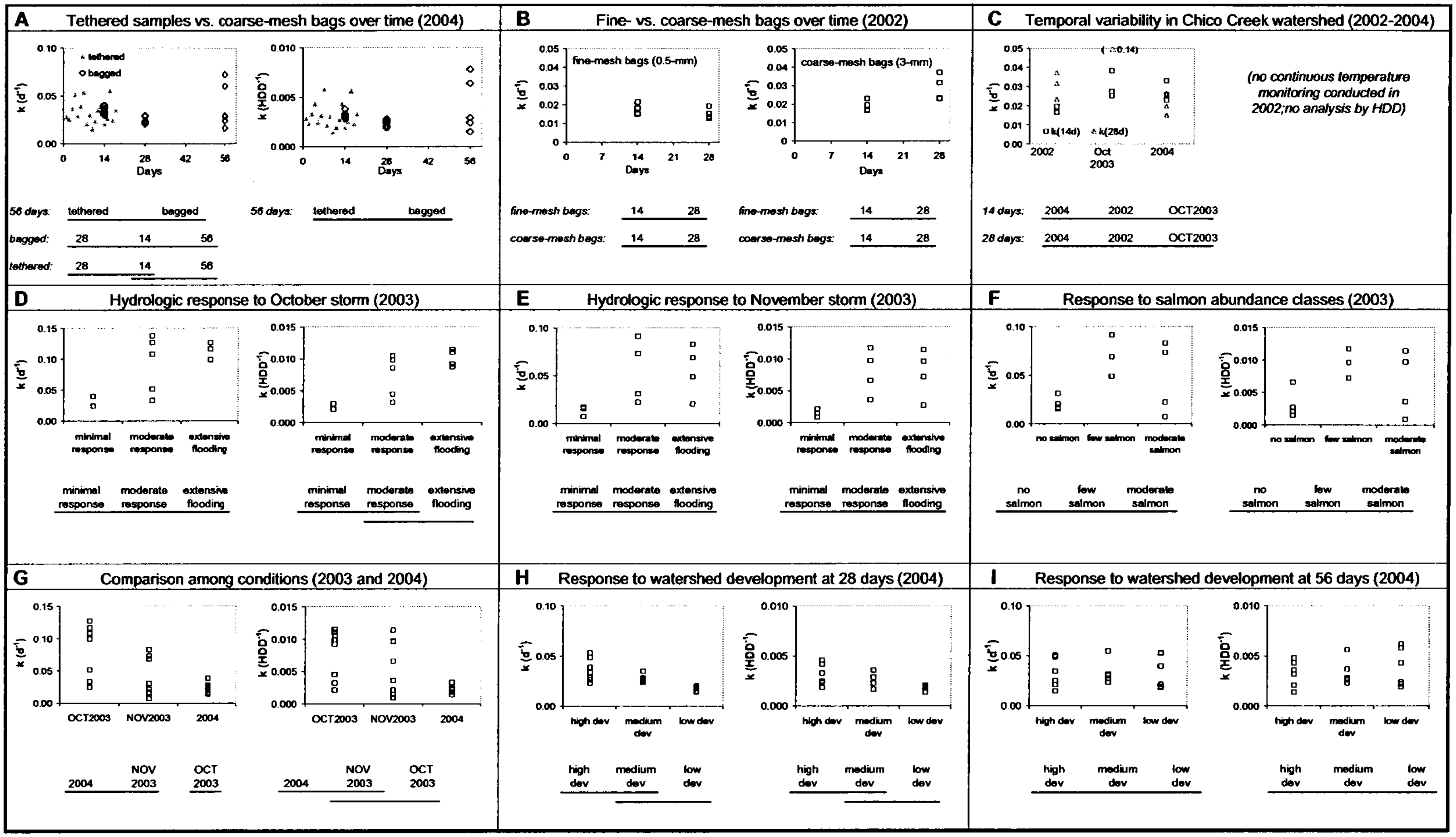

Figure 3-4. Decomposition rates for both $k(d^{-1})$ and $k(HDD^{-1})$ for four experiments. Lines indicate no statistically significant difference.

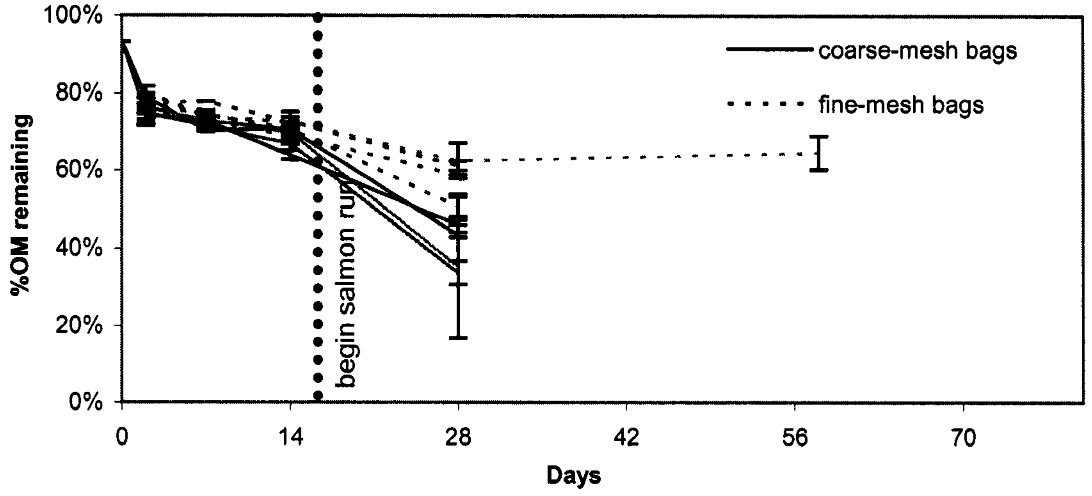

Figure 3-5. Red alder mass loss over time in 2002. Spawning chum salmon entered the system on day 15. Bars represent standard deviation of multiple leaf packs that survived (50% of the remaining leaf packs were destroyed or displaced).

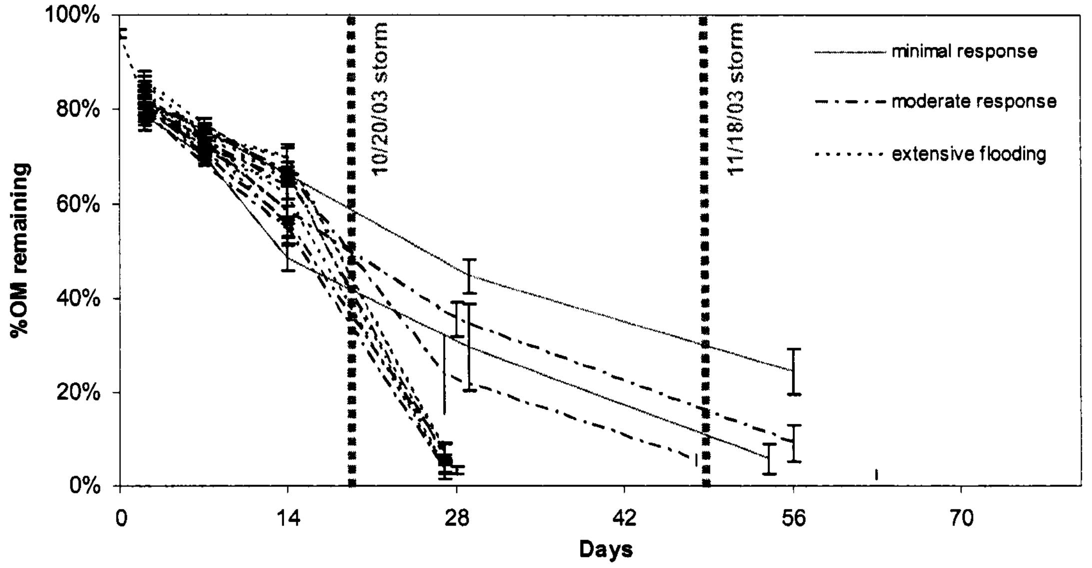

Figure 3-6. Red alder mass loss over time dominated by the October 2003 flood event. Line types distinguish hydrologic response (minimal response, moderate flooding, extensive flooding) to the October flood event. Bars represent standard deviation among leaf packs.

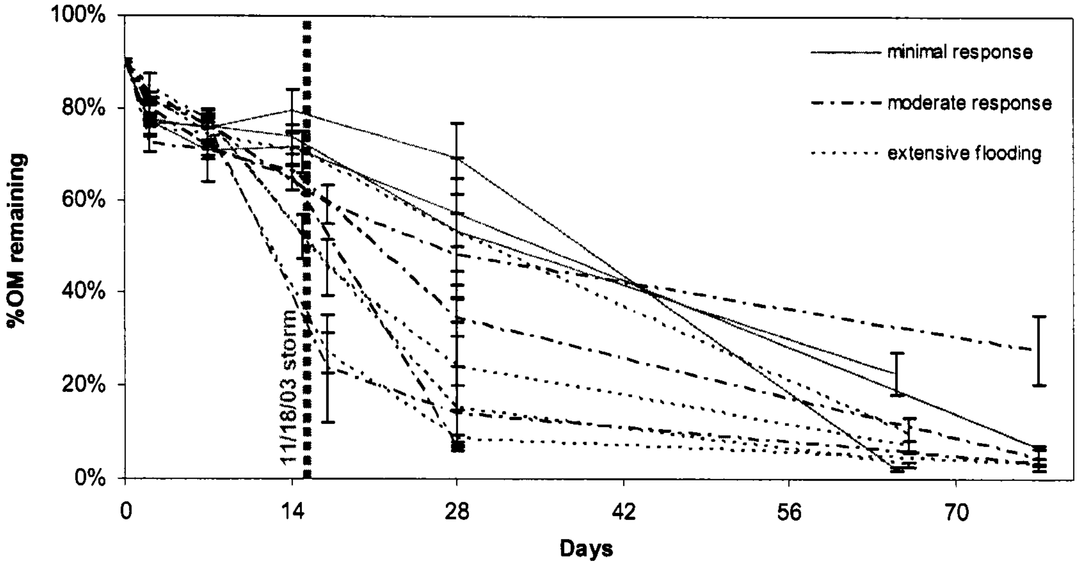

Figure 3-7. Red alder mass loss over time and the effect of the November 2003 flood event. Line types distinguish hydrologic response (minimal response, moderate flooding, extensive flooding) to the November flood event. Bars represent standard deviation among leaf packs.

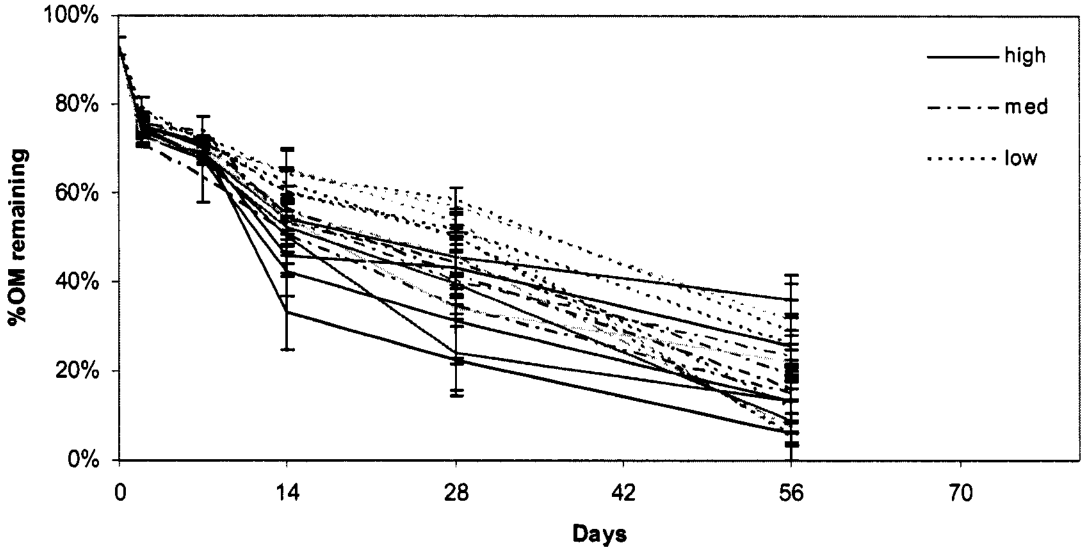

Figure 3-8. Tethered red alder mass loss over time in 2004. Line types distinguish watershed development level (high-, medium- and low-development watersheds). Bars represent standard deviation among leaf packs.

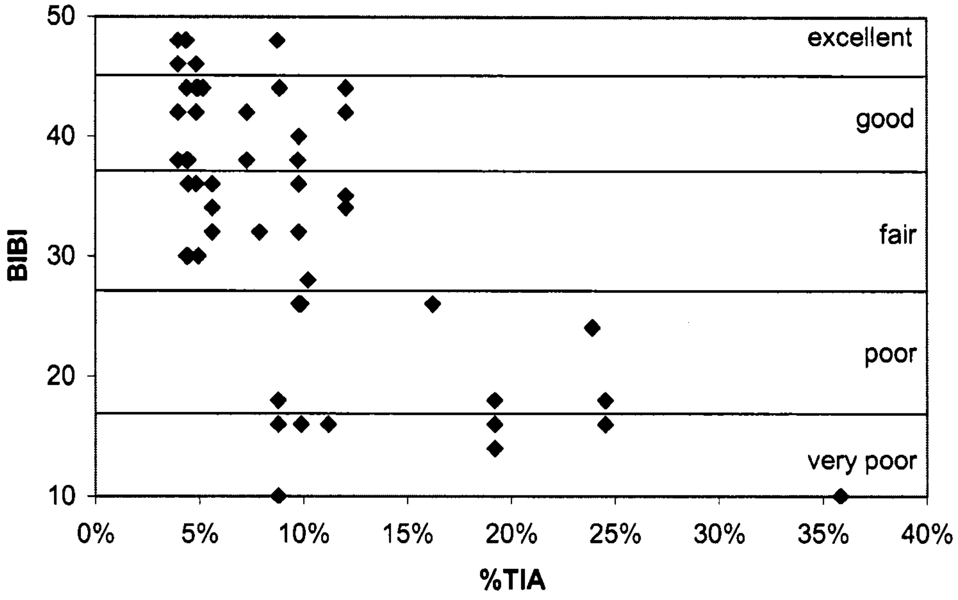

Figure 3-9. BIBI scores for study sites with general condition classifications, as described in Morley (2000). Source: May et al., 1997; unpublished data from Kitsap County Stream Team and Pierce County Water Programs.

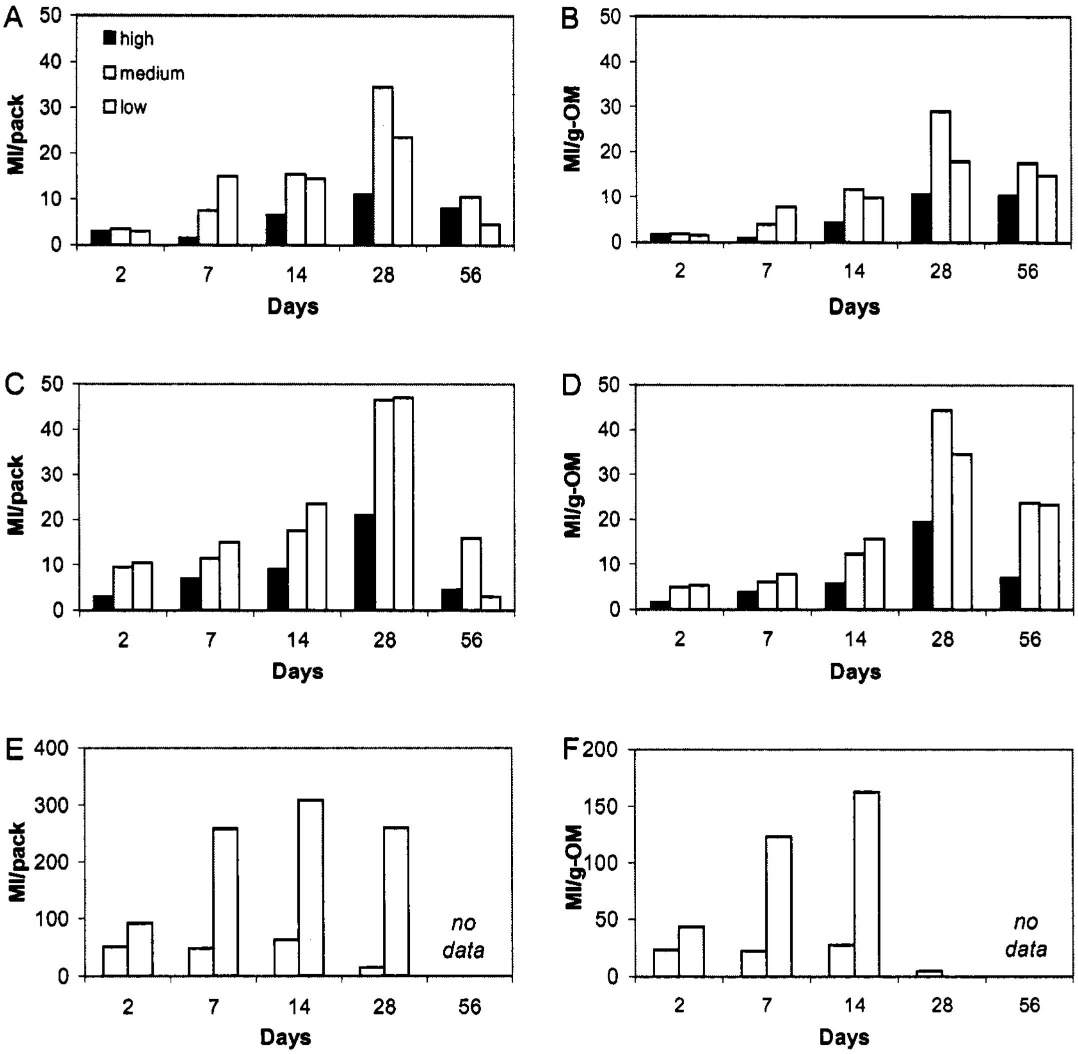

Figure 3-10. Average macroinvertebrate counts and densities per pack by watershed development level for tethered (A and B) and bagged (C and D) samples for the 2004 experiment and for bagged samples (E and F) for the 2002 experiment.

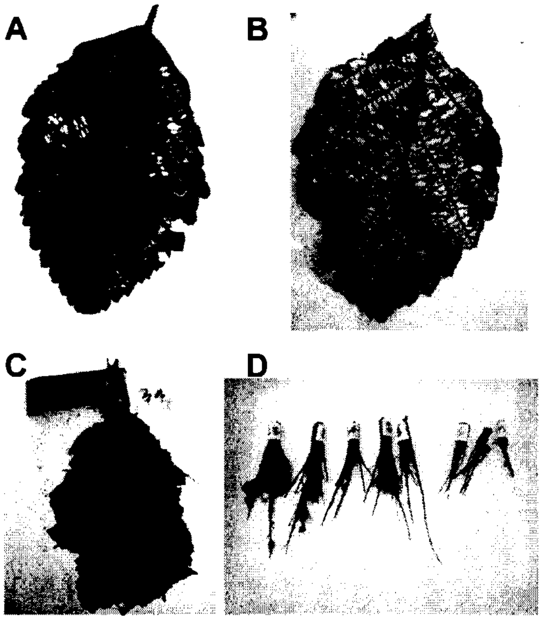

Figure 3-11. Red alder mass-loss patterns dominated by macroinvertebrate feeding (A and B) and physical abrasion (C and D) at early stages (A and C) and late stages (B and D) of leaf breakdown.

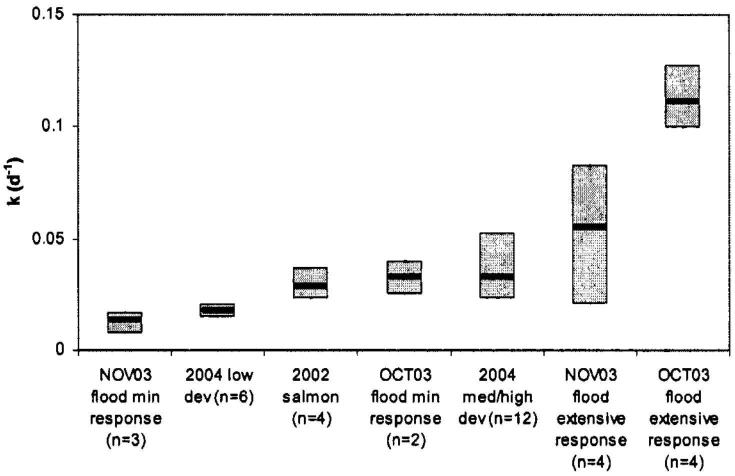

Figure 3-12. Range of red alder breakdown rates compared across disturbance regimes. Lines represent mean rates and bars present the minimum and maximum values.

Table 3-1. Exposure techniques, species, and stations for four decomposition experiments.

Year	**Number of stations**	**Exposure method**
2002	5	Coarse-mesh bags (3-mm)
2002	5	Fine-mesh bags (0.5-mm)
OCT2003	18	Tethered samples
NOV2003	11	Tethered samples
2004	18	Tethered samples
2004	6	Coarse-mesh bags (15-mm)

Table 3-2. Incremental red alder mass loss (percent original organic matter mass) by dominant process. Means are across all stations except where noted, and ranges from all stations are in parentheses. Blanks indicate process did not produce quantifiable mass loss.

Process	**Total mass loss^{A}**	**Day 0**	**Day 0 to 2**	**Day 2 to 14**	**Day 14 to 28**	**Day 28 to 56**
			2002 (salmon)			
Construction	6.4	6.4				
Leaching	17.4		17.4 (14.5-19.0)			
Microbial processes^{B}	19.5			6.4 (4.9-7.8)	13.1 (10.2-20.2)	(experiment terminated)
Macroinvertebrate feeding^{B}	3.2			3.2^{C} (1.5-6.9)		(experiment terminated)
Physical abrasion	18.6				18.6^{C} (4.6-27.1)	(experiment terminated)
			OCT2003 (flood)			
Construction	4.0	4.0				
Leaching	14.7		14.7 (10.2-17.4)			
Microbial and macroinvertebrate	19.0			19.0 (10.3-30.1)		22.3 (18.1-26.7)
Physical abrasion	46.7				46.7 (18.8-62.6)	
			NOV2003 (flood)			
Construction	9.8	9.8				
Leaching	10.8		10.8 (5.6-17.7)			
Microbial and macroinvertebrate	14.9			5.7^{D} (-1.6-15.1)	9.2^{F}	18.2/44.5^{G} (5.3-66.9)
Physical abrasion	29.4			29.4^{E} (5.4-42.5)		
			2004			
Construction	6.8	6.8%				
Leaching	18.3		18.3 (13.9-22.3)			
Microbial and macroinvertebrate	32.3			20.4 (9.8-41.9)	11.9 (2.5-26.2)	25.2 (9.4-48.8)
Physical abrasion	low					

[A] Values represent total mean mass loss produced by each process during the first 28 days of exposure. For example, in 2002 microbial processes produced 6.4% mass loss between day

2 and day 14, and another 13.1% mass loss between day 14 and day 218. The total mass loss was the sum, or 19.5%, over the first 28 days of exposure.

[B] Microbial processes were determined from fine-mesh bags that excluded macroinvertebrates and prevented physical abrasion. Mass loss in coarse-mesh bags represents the combined effect of microbial and macroinvertebrate processing and physical abrasion. In 2002, the microbial contribution between day 2 and day 14 was found by subtracting the fine-mesh bag mass loss (microbial) from the coarse-mesh bag mass loss. Physical abrasion was presumed minimal at the low flows that occurred during the experiment. In 2003 and 2004, only coarse-mesh bags or tethered samples were used, and microbial processing was not distinguishable from macroinvertebrate processing.

[C] Calculated from the difference in average organic matter mass loss in the coarse-mesh bag minus the average organic matter loss in the fine-mesh bag for each of five stations. Value is average difference across five stations.

[D] Average of five stations sampled prior to the November storm on day 14.

[E] Average of six stations sampled after the November storm on days 15-17.

[F] Calculated as the difference in average organic matter mass loss between day 2 and day 28 minus the microbial/macroinvertebrate and physical abrasion mass loss between day 2 and days 14 and 17.

[G] The sites sampled prior to the November storm retained more organic matter at day 28 than those sites that were inaccessible during the storm. These sites lost on average 44.5% between day 28 and day 66. The sites with less organic matter after the November storm lost 18.2% of the original mass loss between day 28 and day 66. The average mass loss for all sites was 28.7% between days 28 and 66. Samples were collected at day 66 rather than day 56 due to access limitations.

Table 3-3. Factors that result from urbanization and expected influence on leaf breakdown rate. The relative importance of each in controlling breakdown in urban Puget Sound Lowland streams is indicated.

Relative magnitude	**Increase breakdown rate**	**Decrease breakdown rate**	**Relative magnitude**
	Hydrology		
Very high	Increased prevalence of floods and other high-flow events	Drought or low summer baseflow	High
	Geomorphology		
Very high	Mobile substrates associated with high flows	Sedimentation and burial	Very high
Moderate	Lack of channel complexity that enhances storm routing		
	Water Nutrient Concentrations		
Moderate	Elevated water column nitrogen and/or phosphorus		
	Vegetation Changes and Loss of Canopy Cover		
High	Shift in vegetation to species with higher leaf nitrogen or phosphorus concentration	Shift in vegetation to species with lower leaf nitrogen or phosphorus concentration	Moderate
Low	Temperature increases	Shift in vegetation to species with high leaf toughness, waxy cuticle, or natural chemical inhibitors	Moderate
Low	Solar radiation increases	Decrease in allochthonous inputs	Moderate
		Presence of algal mats due to enhanced primary productivity	Low
	Metals in Water or Sediment		
		Increase in water column or sediment metals levels	Moderate
		Accumulation of heavy metals in submerged leaves	Moderate
		Presence of flocs, including metal oxide precipitates	Low
	Microbial Processes		
		Decreases in fungal activity	Moderate
	Macroinvertebrate Processes		
		Decreases in macroinvertebrate density, shredder density, or shifts from shredders	Moderate
	DO and pH		
		Low pH	Low
		Anoxic conditions	Low
	Salmon		
		Fewer salmon spawners due to blockages to fish passage, habitat modifications, or poor water quality	High

References

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Appendix A

Leaf Pack Nutrient Levels

Methods

Red alder leaf pack nutrient levels were determined for a subset of samples in the 2003 and 2004 experiments. One tethered sample from each collection date and from each site was randomly selected and oven dried. The sample was ground using a Wiley mill (no. 40 screen) and placed in an acid-washed glass vial. Nitrogen, carbon, and hydrogen levels were determined using a Perkin Elmer Model AD-4 CHN analyzer at the University of Washington, College of Forest Resources Laboratory. Concentrations of phosphorus and other metals were determined using ICP. Samples were digested in high-grade nitric acid and stored in acid-washed vials to avoid contamination. Replicate samples were analyzed at a rate of 10% and the matrix spikes and blanks were analyzed during each laboratory run.

Both nutrient concentration and total nutrient mass represented by the leaf packs were evaluated. Nutrient mass was determined by multiplying the nutrient concentration by the organic mass remaining at each time and site, averaging over all leaf packs.

Results

Quality assurance samples were analyzed to verify the accuracy of the nutrient concentrations. Seven samples were re-submitted to the laboratory as replicates for CHN analyses. Replicate carbon analyses averaged 0.3% relative percent difference (RPD), ranging from 0.01% to 0.6%. Nitrogen replicate analyses averaged 6.6% and ranged from 2.4% to 11.4%. In addition, nitrogen standards (1.20% and 2.86% nitrogen) were analyzed blindly by the laboratory. Results averaged 2.6% RPD ad ranged from 0.8% to 5.3%

compared with the actual concentration. All quality assurance samples were processed using the same protocols as field samples.

Carbon concentrations increase from 46% initially to 49 to 52%, depending on the site, after two days (Figure A-1), steadily declining through 56 days. Because the leaf pack mass decreases over time, carbon mass represented by the leaf pack decreases over time. Red alder leaf packs deliver 0.4 to 1.1 g-C/pack, or approximately 0.2 to 0.4 g-C/g-OM in the first 28 days of submersion.

Nitrogen follows a more complicated pattern. The nitrogen concentration in the red alder leaf packs increased from 1.65% initially to 1.8 to 2.4% after two days (Figure A-2). However, the organic matter content of the leaf pack decreases over that time, leading to no change or a small decrease in nitrogen bound up in the leaf packs after two days. The nitrogen response through day 28 is site specific, with some leaf packs accumulating nitrogen mass up to 2.6 mg-N/g-OM and others losing mass up to 13.6 mg-N/g-OM. Macroinvertebrate counts and densities (Figure A-3) generally peak after 28 days. Nitrogen concentration peaks at most sites after 28 days then concentration and mass decrease. After 56 days, the packs have lost 60 to 95% of their overall mass, but the microbial biofilm development maintains the nitrogen concentration at 2.0 to 3.5%. At one station where the leaf packs were often partially buried by fine sediment, the nitrogen mass remained constant through 56 days of submersion.

Phosphorus concentrations also increase over time (Figure A-4). The trend could be due to the development of the microbial biofilm and differential leaching as other soluble elements are lost from the leaf matrix. The mass of phosphorus represented by the leaf pack initially decreases from 1.1 mg-P/pack to 0.23 to 0.9 mg-P/pack in the first two days. After 28 days, two of the nine sites lost phosphorus mass (0.3 to 0.7 mg-P/pack for a loss of 0.4 to 0.8 mg-

P/pack), five of the sites remained constant (1.1 to 1.3 mg-P/pack) and two sites gained phosphorus mass at 0.4 to 0.6 mg-P/pack, to a mass of 1.5 to 2.0 mg-P/pack).

Notes to Appendix A

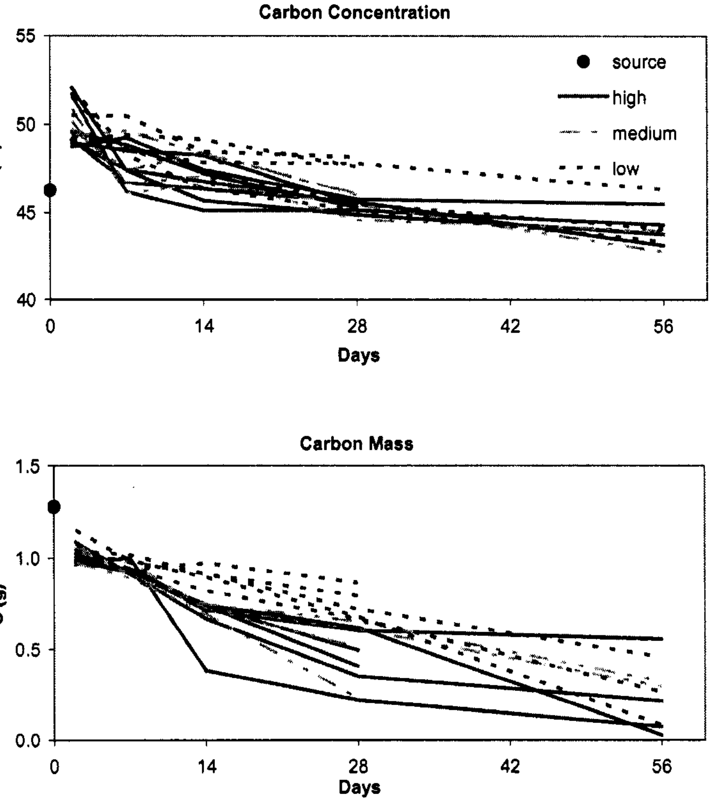
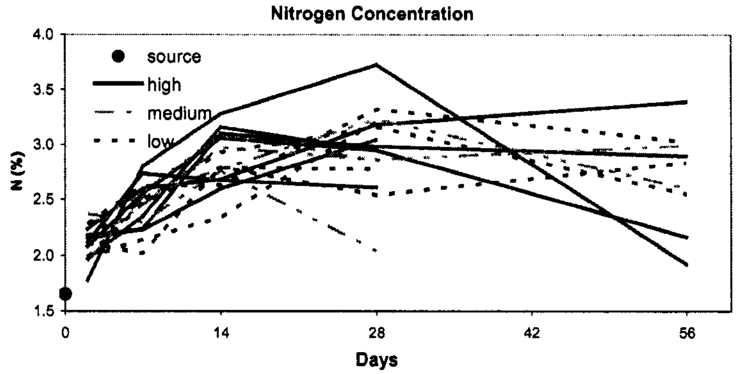


Figure A-1. Carbon concentration and mass represented by red alder leaf packs over time (2004 experiment by watershed development level).



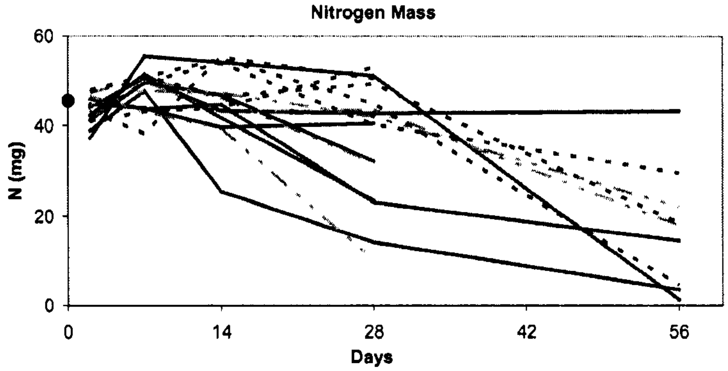


Figure A-2. Nitrogen concentration and mass represented by red alder leaf packs over time (2004 experiment by watershed development level).

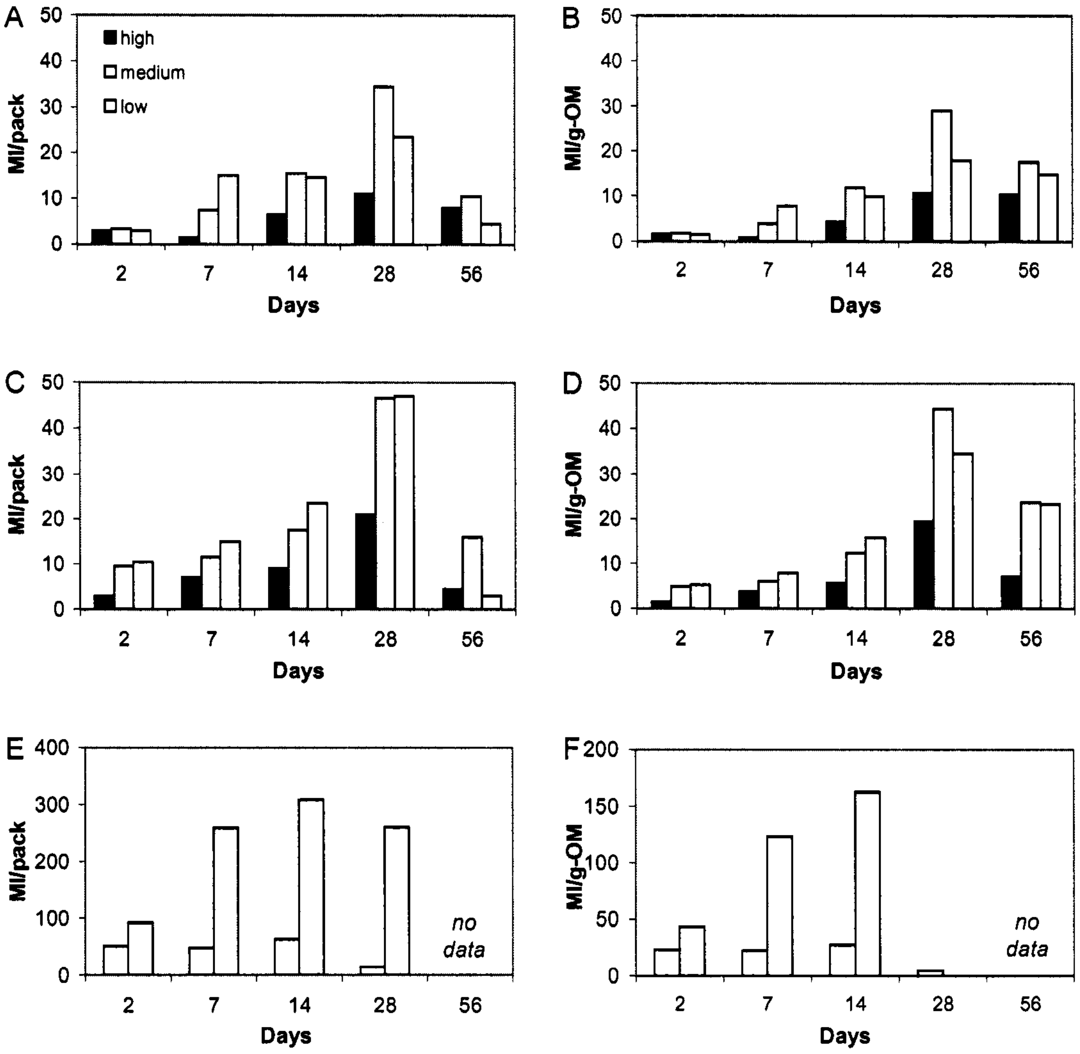


Figure A-3. Average macroinvertebrate counts and densities per pack by watershed development level for tethered (A and B) and bagged (C and D) samples for the 2004 experiment and for bagged samples (E and F) for the 2002 experiment.

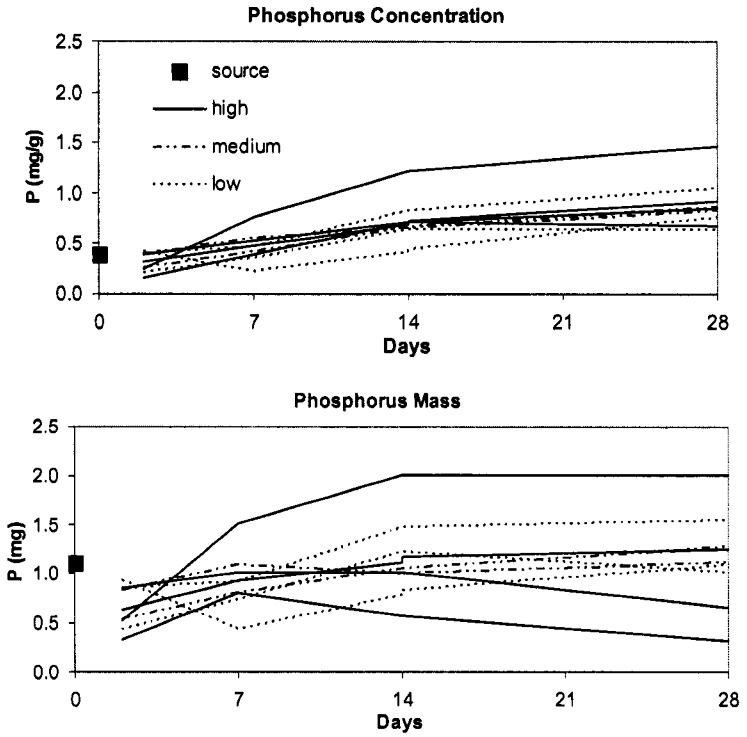


Figure A-4. Phosphorus concentration and mass represented by red alder leaf packs over time (2004 experiment by watershed development level).

VITA

Mindy Lynn Roberts was born in Middletown, Ohio, to Jim and Linda Roberts, who encouraged her to pursue math, science, and engineering. She moved ten times before college, inspiring a love of travel and exploration. In 1989, she earned a Bachelor of Science in Civil Engineering from the University of California, Berkeley, where she was president of the student chapter of the American Society of Civil Engineers and a national concrete canoe champion. She earned a Master of Science in Civil and Oceanographic Engineering in 1992 from the Massachusetts Institute of Technology and Woods Hole Oceanographic Institution, during which time she met her husband, Jim Gawel. Mindy earned her Ph.D. in Civil and Environmental Engineering from the University of Washington in 2007.

Mindy has worked in academia; for federal, state, and local governments; consulting; and nonprofit organizations. She chuckles when each of the entities describes its individual role as needing to educate the others. The key to success is mutual respect.

When not tromping through urban streams around Puget Sound, Mindy usually can be found in a helmet on a mountaintop somewhere in Washington, British Columbia, South America, or Europe with Jim and other climbing partners. She enjoys the focus that technical rock and ice climbing demand and applies mountaineering knowledge to a variety of professional and personal situations: approach seemingly insurmountable problems one pitch at a time.