University of Washington
Department of Civil and Environmental Engineering

QUALITY INDICES FOR URBANIZATION EFFECTS IN PUGET SOUND LOWLAND STREAMS

C.W. May, E.B. Welch, R.R. Horner, J.R. Kar and B.W. Mar

Final Report for Washington
Department of Ecology

Water Resources Series
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Abstract

The Puget Sound lowland (PSL) ecoregion contains an abundance of complex and historically productive salmonid habitat in the form of small streams as well as their riparian forests and wetlands. These watersheds are under intense pressure due primarily to the cumulative effects of urban development. Instream habitat characteristics, riparian conditions, physio-chemical water-quality, and biological attributes of 22 PSL streams (120 survey reaches) were studied over a gradient of development levels to determine relationships between urbanization and stream quality and suggest target conditions for management/protection. Urbanization of PSL watersheds has resulted in an increase in the fraction of total impervious area (% TIA) and a decrease in forested area, including a significant loss of natural riparian forests and wetlands. The cumulative effects of a modified hydrologic (disturbance) regime, the loss of instream structural complexity, and the alteration of channel morphological characteristics accompanying urbanization have resulted in substantial degradation of instream habitat during the initial phases of the development process. As the level of basin development increased above 5% total impervious area (% TIA), results indicated a precipitous initial decline in biological integrity as well as the physical habitat conditions (quantity and quality) necessary to support natural biological diversity and complexity. The frequency, volume, and quality of large woody debris (LWD) decreased significantly as basin development and riparian encroachment increased. Loss of LWD due to washout and removal, as well as a reduction in LWD recruitment due to loss of mature riparian forest areas, were significant factors. As a result of the reduction in the quantity and quality of LWD, along with the effects of a modified hydrologic regime, coho rearing habitat was significantly reduced. Salmonid spawning habitat was also degraded by the cumulative effects of urbanization. Fine sediment in spawning gravels generally increased as urbanization increased, while intragravel dissolved oxygen (IGDO) also decreased during the period of salmonid embryo development. Chemical constituents (primarily metals) of water quality during baseflow conditions, as well as storm events, were insufficient to have produced adverse effects in streams with low to moderate % TIA, but increased markedly in highly urbanized basins (TIA>45%).

Results suggest that resource management should place a high priority on preservation and protection of high quality stream ecosystems (TIA <5%) that currently support natural salmonid populations (coho and cutthroat). Mature, riparian forests dominated by coniferous trees should be the long-term management goal. A wide (>30 m) and near-continuous (<2 breaks/km) riparian zone appears to be a necessary, although not a wholly sufficient condition for a natural level of stream quality and biotic integrity. Restoring the natural hydrologic regime should be a primary goal for rehabilitation and enhancement efforts. A set of stream quality indices and instream habitat target conditions are proposed for monitoring and managing PSL streams.

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ACKNOWLEDGMENTS

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INTRODUCTION

The ultimate goal of this project was to develop an understanding that would help to improve and/or preserve streams in urban and urbanizing watersheds of the Puget Sound Lowland (PSL) in support of native salmonid populations. To that end, the specific objectives were to develop a set of recommended stream quality indices and target conditions that relate to urbanization intensity. This should assist resource managers in predicting the effect of proposed development on a stream's hydrologic regime, salmonid habitat, channel morphologic characteristics, physio-chemical water quality, and the overall ability to support aquatic biota. That was approached by testing hypotheses that directly or indirectly link urbanization to various measures of watershed and riparian corridor condition, instream habitat characteristics, and biological integrity.

Watershed and Riparian Conditions	=>	**Instream Habitat Characteristics**	=>	**Biological Integrity**

Understanding the linkage between landscape (watershed) level processes, physical habitat, and the organisms that inhabit aquatic ecosystems is a key to successfully managing these resources. The basic assumption is that there is a high probability of maintaining self-sustaining wild salmonid populations in PSL streams by maintaining a specified set of instream habitat characteristics, which in-turn depend on an established set of watershed and riparian corridor conditions.

With few exceptions, the investment of effort and funds to control the effects of urbanization on streams has gone to characterizing the magnitude of stormwater runoff and its chemical/physical (water quality) constituents. Relatively little has been invested to study the actual receiving stream and its habitat and biological resources, which are allegedly the main purposes for controlling urban runoff, while the literature is replete with data on stormwater chemical quality, models to predict runoff and pollutant loads and the design of devices to control the quantity and quality of stormflow. Apparently, there has been little interest in knowing what stream levels of chemical-physical quality actually result from stormwater input and even less on the relative effects of that input. While degradation of streams and loss of salmonid resources have been a generally accepted (and deplored) consequence of urbanization, there has been little actual effort in documenting the actual biological effects and their causes in the Pacific Northwest (Pedersen and Perkins, 1986; Scott and Steward, 1986; Richey and Perkins, (1982) or elsewhere (Whiting and Clifford, 1983; Klein, 1979; Garie and McIntosh, 1996). The Pacific Northwest citations were from one EPA-supported study on Big Bear and Kelsey Creeks, two streams in this study-set.

Which has a more important effect on stream biota and salmon resources when stream basins are urbanized—the loss of physical habitat required by those fishes and their support biota or the chemical water quality constituents so routinely measured and for which enforceable standards exist for some? Total metals (Cu, Zn and Pb) concentrations in stormwater are consistently higher than in untreated sewage effluent, solids are roughly equal to effluent, and BOD is high enough to exert a significant demand (Welch, 1992). Are these concentrations high enough to produce an effect even under the worst conditions of high storm runoff? Nutrients are routinely monitored, but why? They are clearly important to lake quality, but do they actually cause nuisance algal problems in these streams, or does canopy cover, when present, restrict light that limits periphyton production instead? And are other variables/constituents not routinely measured that are more important to the target biological resources? If adverse constituent levels/inputs occur, are they seasonally timed to produce a significant effect? Without answers to these questions, the effort and expenditure for control may not be cost effective.

Ecologically-meaningful and convenient measures of watershed and riparian conditions are needed to assess stormwater effects on stream resources, instream habitat characteristics, and biotic integrity. A primary objective of this project was to develop a set of stream quality indices to better monitor and manage aquatic resources in PSL streams. They would describe basin- and stream-level conditions that could trigger a range of specific management actions and/or rehabilitation efforts. Watershed conditions will be expressed using accepted measures such as basin imperviousness, as well as supplemental and alternative measures, such as road-density, riparian integrity, and drainage-density. Possible "threshold levels" of development were an expected outcome of this work. But more importantly, a project result was to be a set of regionally valid physical habitat variables or indices that provide an accurate assessment of instream conditions with suggested "targets" to maintain acceptable stream quality. A set of low-impact, "reference" streams was chosen as a basis for establishing acceptable or optimal conditions for PSL streams. Although this reference condition may not represent the absolute "ideal" or pristine condition, it is considered the "best-possible," reasonable and attainable basis for comparison.

While a useful set of quality indices for urban streams as a tool for stream protection/management has value, they should be applied with caution. Reliance on in-channel habitat and/or biologically based indicators alone may not prevent the cumulative effects of urbanization. If the burden of proof rests with documentation of fish and other aquatic resource degradation before taking corrective action, complete recovery may not be feasible or possible. Watershed modification and stream degradation will likely continue beyond the point of most cost-effective damage, due to the inherent time-lag between watershed activities and recognizable response in the stream. To manage streams most cost-effectively, instream indices must be constantly related to watershed-level variables to anticipate the early appearance of adverse effects.

The cumulative effects of urbanization on stream hydrologic regime, channel morphologic characteristics, instream salmonid habitat, and physio-chemical water quality were examined in detail. Cumulative effects of urbanization on riparian forest/vegetation were also examined.

Specific objectives were as follows:

- Document the existing conditions of representative urban streams in the PSL with respect to instream habitat quantity, quality, and diversity. To relate the temporal (high flow vs. low flow) and spatial changes in these urban streams, as a result of development activity, to the specific requirements of salmonid fishes.
- Establish a set of "target" conditions for in-stream habitat to be used as guidelines for rehabilitation and enhancement efforts.
- Establish sampling protocols for monitoring and assessing PSL streams, using developed stream quality indices. Stream quality should be evaluated from an integrated biological, chemical, and physical perspective in order to accurately reflect the existing level of degradation and evaluate restoration potential.

METHODS

Study Streams

A group of 22 lowland streams were selected as a subset of all PSL streams to examine the effects of urbanization on stream ecosystem integrity, water quality, and salmonid habitat. Figure 1 shows the general location of the study streams. The streams were chosen to represent a range of low to high land-use development levels. Land-use patterns, characteristic of urbanized and/or urbanizing basins, included residential, commercial, and industrial categories. Watersheds with the primary land-use patterns as agricultural or resource extraction (timber harvest or mining) were excluded. Basin size and land-use characteristics of the watersheds are shown in Table 1. Criteria used for stream selection were as follows:

- Has existing or historic salmonid fish habitation; if no longer existing or if reduced from historic levels, sufficient information must be available for a basic description of the fish use that once existed (Table 2).

- If fish production is degraded from historic level, the principal cause is urban stormwater drainage; other human impacts such as agricultural, logging, and channelization (or other instream engineering works) are relatively insignificant.

- Altitude of the stream's catchment ranges from sea level to 150 meters (500 ft) and the average stream gradient does not exceed 5 percent.

- There are significant areas of relatively rapid flow velocities with stony substrata; or if such areas do not now exist as a result of urbanization, there is evidence that they did in the pre-urban condition.

- The surficial geology and soils of the catchment represent one of the typical fundamental types for the Puget Sound lowland region (glacial till or outwash), and no highly unusual geologic or soil conditions exist.

- Drainage upstream of potential study sites is not highly influenced by a lake or extensive wetlands that would buffer the effects of both storm runoff water quantity and quality at the study location.

- Stormwater management practices prevailing in the catchment are in the typical range for the Puget Sound region (i.e., not atypically primitive).

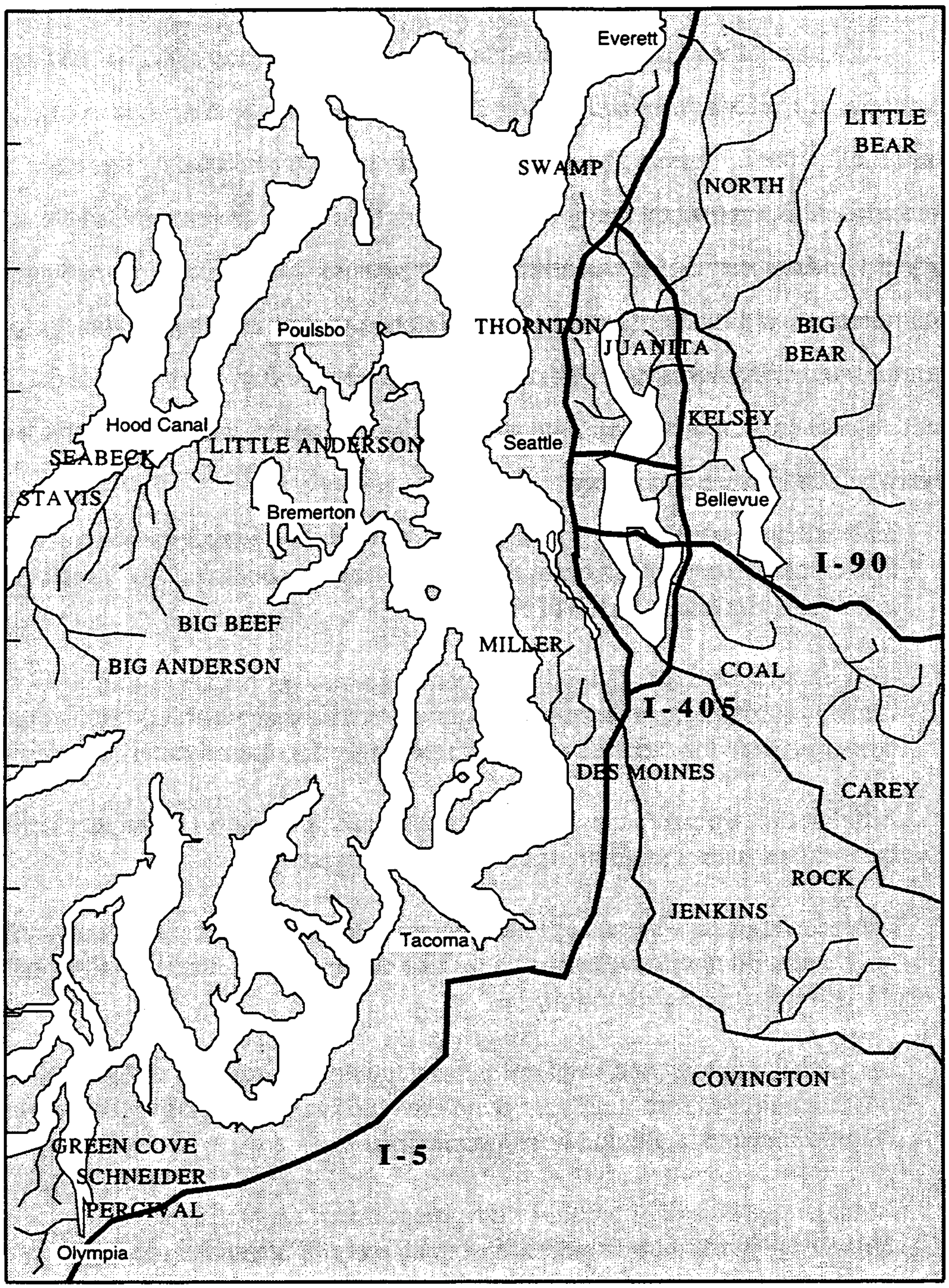

Figure 1: Map of Puget Sound lowland (PSL) region showing approximate location and distribution of study streams.

Table 1: Watershed characteristics for Puget Sound lowland streams.

PSL STREAM CATCHMENT	WATERSHED BASIN AREA (km^2)	WATERSHED BASIN AREA (Acres)	STREAM BASIN LOCATION	AERIAL PHOTO YEAR	LAND-USE DATA TYPE	TOTAL IMPERVIOUS (%TIA)	EFFECTIVE IMPERVIOUS (%EIA)	BASIN LAND-USE CLASSIFICATION (% in each category) UNDEVELOPED FORESTED	DEVELOPED OPEN	RURAL (LD) RESIDENTIAL	SUBURBAN (MD) RESIDENTIAL	URBAN (HD) RESIDENTIAL	COMMERCIAL INDUSTRIAL	TOTAL DEVELOPED	ROAD DENSITY (km/km^2)	HIGHWAY RUNWAY (km)
Big Anderson	15.5	3875	Kitsap Cty	1993	GIS	1.2	0.6	87	10	3	0	0	0	13	2.6	0.0
Carey	20.2	5058	King Cty	1991	LANDSAT	1.3	0.4	83	13	3	1	0	0	17	3.5	0.0
Slavis	18.0	4500	Kitsap Cty	1993	GIS	1.5	0.4	78	14	8	0	0	0	22	2.9	0.0
Seabeck	14.3	3575	Kitsap Cty	1993	GIS	2.7	1.6	79	10	10	1	0	0	21	2.5	0.0
Big Beef	34.3	8575	Kitsap Cty	1993	GIS	3.1	2.0	80	8	8	3	0	1	20	2.3	0.0
Rock	33.6	8397	King Cty	1991	LANDSAT	3.2	2.4	84	6	7	1	0	2	16	1.6	0.0
Little Anderson	15.1	3775	Kitsap Cty	1993	GIS	3.4	1.8	71	17	9	2	0	1	29	3.0	0.0
Covington	57.0	14253	King Cty	1991	LANDSAT	3.9	2.3	73	5	17	4	0	1	27	1.4	0.0
Green Cove	10.2	2540	Olympia	1989	BASIN PLAN	8.1	5.5	64	12	8	14	0	2	36	2.3	0.0
Big Bear	89.3	22315	King Cty	1995	BASIN PLAN	10.9	7.7	52	9	22	13	1	3	48	2.5	0.0
Jenkins	41.3	10203	King Cty	1995	LANDSAT	13.1	9.7	53	3	19	18	4	2	47	2.5	5.5
Little Bear	37.7	9417	Snohomish Cty	1989	BASIN PLAN	13.8	10.0	43	12	32	6	2	7	57	4.9	3.8
Coal	16.8	4194	Bellevue	1991	LANDSAT	20.8	15.8	46	6	2	37	4	5	54	2.9	1.2
Percival	21.2	2120	Olympia	1989	BASIN PLAN	21.8	18.4	40	22	8	9	5	16	60	4.5	2.1
North	72.4	18103	Snohomish Cty	1989	BASIN PLAN	26.4	20.9	28	8	18	26	9	11	72	4.9	27.3
Swamp	64.2	16052	Snohomish Cty	1989	BASIN PLAN	31.1	25.3	22	11	13	29	13	12	78	8.5	31.2
Schneider	2.7	265	Olympia	1989	BASIN PLAN	42.2	35.5	12	6	7	26	40	9	88	9.4	0.0
Juanita	18.4	4600	King Cty	1991	LANDSAT	45.4	38.6	12	8	1	35	25	19	88	8.2	6.0
Kelsey	21.2	5292	Bellevue	1991	LANDSAT	49.2	42.2	5	11	1	34	25	24	95	10.1	7.8
Miller	20.8	5200	King Cty	1991	LANDSAT	49.4	43.1	6	15	0	26	39	9	94	9.8	13.8
DesMoines	15.2	3808	King Cty	1991	LANDSAT	49.1	44.6	12	22	0	14	19	33	88	10.4	35.4
Thornton	29.7	7424	Seattle	1991	LANDSAT	55.4	49.0	6	5	0	11	63	15	94	8.9	14.3

- Stream dimensions and flow velocities during storm runoff of moderate quantity are conducive to sampling and taking measurements within the channel.

Stream selection was also influenced by geographic and logistical considerations, as well as the recommendations of project cooperators. Streams were selected to provide a gradient of urbanization levels in order to demonstrate a link between urbanization and stream quality.

Table 2: Historic salmonid utilization in Puget Sound lowland streams.

Stream	Coho	Chinook	Sockeye	Chum	Pink	Steelhead
North	X	X	X			X
Swamp	X	X	X			X
Little Bear	X	X	X			X
Big Bear	X	X	X			X
Juanita	X		X			X
Thornton	X					X
Kelsey	X		X			X
Coal	X					X
Carey	X	X	X			X
Rock	X	X	X			X
Covington	X	X				X
Percival	X	X		X		X
Green Cove	X			X		
Schneider	X			X		
Little Anderson	X			X		
Big Beef	X	X		X	X	X
Big Anderson	X	X		X	X	X
Seabeck	X			X		
Stavis	X			X		X
DesMoines	X			X		X
Miller	X			X		X
Jenkins	X					X

Note: All study streams also have resident and/or sea-run cutthroat populations.

The history of human activity is also very similar for most catchments including repeated cycles of timber harvest. Remaining areas of old-growth forest are scarce. Undeveloped zones are composed of mature second-growth, young alder-dominated stands and (open) shrub/grass areas. Low-intensity, livestock related agriculture has been a part of most sub-basins within the past century, as well. "Hobby-farms" are currently common in some rural areas. A detailed description of each study sub-basin is contained in May (1996; Appendix A). See **Results** for physical characteristics of the streams.

Puget Sound Lowland Ecoregion

The PSL ecoregion encompasses the entire Puget Sound basin from sea-level up to the Cascade and Olympic mountains. Ecoregions are characterized by similar geologic history, soils, land-form, vegetative succession, and land-use patterns (Omernik and Gallant, 1986). Streams of the PSL ecoregion are primarily fed by precipitation, are low-gradient, meandering, and are dominated by a pool-riffle morphology. The majority of soils in the PSL are glacial in origin and are strongly influenced by the dominant coniferous forest vegetation (Douglas Fir, Western Hemlock, Western Red Cedar). Glacial activity in the recent geologic past is responsible for the current geomorphologic characteristics common to stream systems of the region (Detenbeck et al, 1992).

Precipitation patterns within the region can vary considerably, but the overall characteristics are similar. The bulk of the precipitation (>75%) occurs between October and March, with a dry summer period from July through September. The historic average precipitation pattern for the PSL is 40 cm in the fall, 35 cm in the winter, 15 cm in the spring, and less than 10 cm during the summer (100 cm total). The National Weather Service (NWS) rain-gage at SEATAC airport is representative of the region's precipitation pattern (Figure 2). Snowfall is infrequent and rain-on-snow events are rare.

Stream Classification

Detecting and predicting the effects of land-use activity on stream habitat and aquatic biota is complicated because the responses to disturbance can occur over a variety

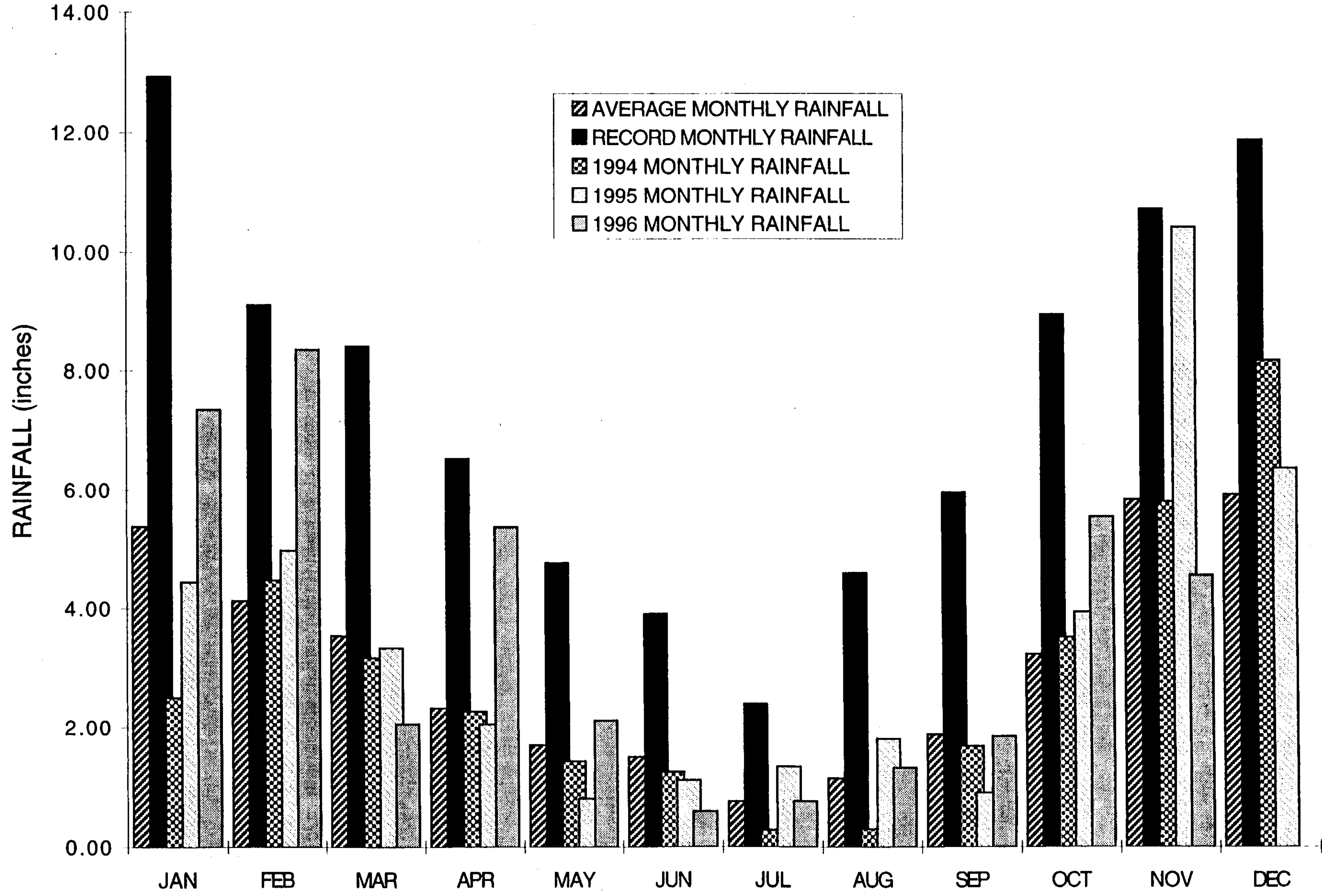

Figure 2: Record of monthly precipitation for the Puget Sound region 1994-1996 (National Weather Service at SEATAC Airport).

of spatial and temporal scales (Frissell et al, 1986). Natural processes also interact with human-induced disturbances. Physical habitat features also vary from site to site and may, therefore, vary in sensitivity to disturbance. Nevertheless, the long-term, geologic and geomorphic structure of the drainage basin can be viewed as a template which structures the complex response of the stream system (Frissell et al., 1986).

The natural geomorphic characteristics of each stream were determined before comparing the level of urbanization and degree of degradation among streams. If the morphological characteristics of each stream were known, then inferences could be drawn about their disturbance regimes and their expected response to urban impacts. Patterns of stream response to urban disturbance would be confounded or masked by natural variation without a systematic classification and stratification of data. A rationale was needed to establish that the study streams would naturally respond similarly to disturbance so that their response to urbanization could be assessed. Thus, the study was designed along the hierarchical framework of stream system classification outlined by Frissell et al. (1986). Streams were viewed in the context of their watersheds and landscape-level characteristics. This scheme emphasizes a habitat-centered view of the stream's relationship to its watershed over a range of scales in space and time. The stream network is hierarchically organized into levels: stream system, segment, reach, habitat feature (pool/riffle), and micro-habitat. Stream responses were assessed at various levels within this framework.

The Rosgen scheme, which classifies stream segments based on similar geomorphic characteristics, was chosen for comparison of stream segments (Rosgen, 1994). Ensuring that streams are of a similar type reduces the variation that might appear to be natural, but may actually be due to comparison of inherently different systems. Morphologic characteristics of the stream channel are determined by physical processes, primarily those of a fluvial nature. Channel morphologic characteristics are influenced by several variables (Leopold et al, 1964). These include channel width, depth, water velocity, discharge, gradient, floodplain features, streambed roughness, channel structure, longitudinal profile, sediment load, and sediment/substrata size. A change in any one of these variables initiates a series of adjustments in the other variables until the channel reaches a new equilibrium condition. These physical variables are used in the Rosgen classification system as "delineative criteria" (Rosgen, 1994), and represent the stream channel's dominant features. The Rosgen scheme is hierarchical, beginning with the stream segment as either a single or multiple channel. The segment is further classified by the entrenchment ratio, which is defined as the flood-prone width (FPW) to bankfull width (BFW) ratio and then by its width-depth (BFW/BFD) ratio. Finally, sinuosity and slope are determined. An additional level of classification based on dominant substrata size may also be used. Based on segment classification, the Rosgen system also provides interpretive information on sensitivity to disturbance, recovery potential, sediment dynamics, level of influence of riparian vegetation, streambed stability, and streambank erosion potential. This interpretive information can be used in designing a watershed management plan, cumulative effects analysis, and for restoration/enhancement guidance. A main criticism of the Rosgen method, as with other classification schemes, is that LWD is omitted, which reflects the non-forested environment in which it was designed. However, the classification tool is still useful.

Watershed Characterization

Watershed characterization is the description of the current natural and human-related attributes of the basin and includes:

- Dominant natural and human features of the watershed that affect ecosystem function and biological integrity.
- Cataloged and/or mapped watershed attributes such as geologic, soil, and topographic characteristics.
- Key land-use features and land-cover patterns of the basin including roads, forested areas, and development levels, usually on a map.
- Delineated municipal jurisdictions and regulatory responsibilities within each watershed.
- Current and historic salmonid utilization for each stream system.

- List of beneficial uses common to the watershed and their relative importance.
- Unique or critical resource issues and problems.
- Water resource management programs that currently exist.

The most up-to-date land-use information was obtained from each municipality in the form of basin plans, land-use maps, satellite imaging, aerial photos, and computer-digitized (GIS) data bases, when available. All land-use data, roads, and basin features were "ground-checked" in the field. Soil data were obtained from the US Soil Conservation Service (USSCS) surveys. Past water quality and hydrologic data were compiled from US Geologic Survey (USGS), county, and government agency records. USGS topographic maps were used to determine basin-level morphologic characteristics and to delineate stream network properties. Officials from governmental and non-governmental agencies were interviewed for additional and/or specific information. Citizen groups and individual land-owners were also consulted.

Reference Streams

Physical, chemical, and/or biological assessment of the degree of urbanization effect requires either a control or reference or at least an unbiased estimate of attainable conditions. Relatively undisturbed streams and watersheds were selected for reference or controls within the PSL. These "regional reference sites" should have the same land-surface form, underlying geology, soils, vegetation patterns, and climate as the streams and sub-basins under study (Hughes et al., 1986) and represent the optimal conditions against which urbanized streams are compared. Such reference streams in undeveloped or low-impact watersheds can also provide goals for preservation, enhancement and restoration.

These reference streams are not "pristine,"due to the history of timber harvest and regional development, but are assumed to be as near to naturally functioning and ecologically intact streams that exist and would provide the long-term stability and diverse habitat necessary to support a full-range of salmonid species (Peterson et al, 1992). The biota, chemical water quality constituents, and physical habitat characteristics of the regional reference sites serve as benchmarks for the disturbed streams and watersheds. Comparisons between reference and urbanized streams can be used to explain changes as a result of human activities, their probable causes, and the ecological implications.

Measures of Urbanization

Imperviousness

The most common measure of urbanization level is the percentage of watershed area covered by impervious surfaces. Impervious surface is defined as any surface that prevents or inhibits the natural infiltration process such as roads, parking lots, and roof tops. Vegetated areas such as lawns, golf courses, and parks can also be considered relatively impervious, due to the removal and compaction of surface soils during development. The use of imperviousness is consistent with the underlying relationship between amount of impervious surface and the magnitude of runoff. Figure 3 illustrates the increase in runoff coefficient with increasing imperviousness. The runoff coefficient represents the fraction of rainfall volume that is actually converted to stormwater runoff. The runoff coefficient, and therefore runoff volume, closely related to imperviousness, except at very low levels where soils and slope factors tend to dominate, and where excessive runoff is not a major problem (Schueler, 1994). The population density of an area is also highly correlated with percent impervious area (Stankowski, 1972).

The rationale for using imperviousness as an indicator of development is widely accepted. Imperviousness is integrative in nature and can represent an index of cumulative effects on aquatic resources irrespective of specific land-use factors or complex NPS pollution problems (Arnold and Gibbons, 1996). Although impervious surfaces do not generate pollution themselves, they accumulate pollutants, convey stormwater, and inhibit infiltration of runoff. They are, thus, a major contributor to the change in basin hydrologic regime and are a significant component of urban land-uses

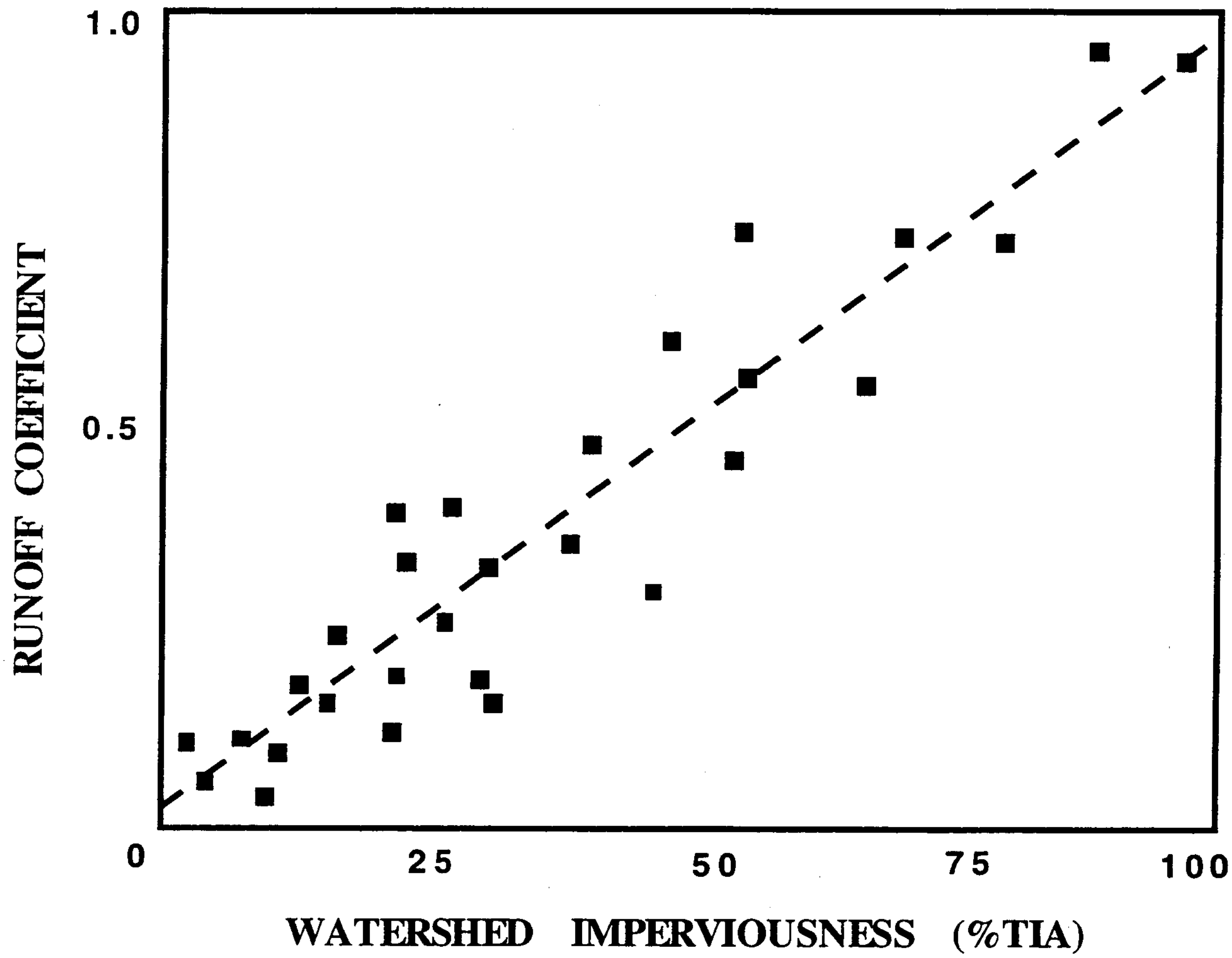

Figure 3: Runoff coefficient as a function of basin imperviousness (from Schueler, 1994).

that generate pollutants. Impervious surfaces are a major factor in the conveyance of stormwater and inhibit infiltration of runoff (Arnold and Gibbons, 1996).

The most common measure of imperviousness is percent total impervious area (%TIA) and is based on assigning a regionally-accepted, specific percent imperviousness to the various categories of land-use found within each basin (Alley and Veenhuis, 1983; Prych and Ebbert, 1986; Taylor, 1993; Schueler, 1994; Olthof, 1995). The extent of imperviousness in each basin was determined by first measuring the areas covered by each land-use type. The total area covered by impervious surface in each land-use category was estimated based on hydrologic studies of typical development patterns for these land-use types. The land-use-%TIA values were multiplied by their respective surface areas to obtain a final %TIA for each basin (Figure 4).

While the use of %TIA is a generally accepted index of urbanization, it is partly subjective. Division of each basin into polygons of land-use, calculating their areas, and assigning a land-use category assigned is a complex task requiring detailed land-use maps and/or aerial photographs. Land-use categories must then be assigned a value of % imperviousness. This is typically based on representative levels of TIA for that particular type of land-use and the corresponding run-off coefficients.

In addition to %TIA, the % of "effective" impervious area (%EIA) was often used. Effective impervious surface is directly connected to the drainage system and/or stream network. Impervious surfaces can be made ineffective by allowing or encouraging infiltration of the runoff before it reaches a conveyance system, thus reducing the total runoff reaching the stream. The most widely used land-use categories and their % TIA/EIA values currently used in the PSL region are shown in Table 3.

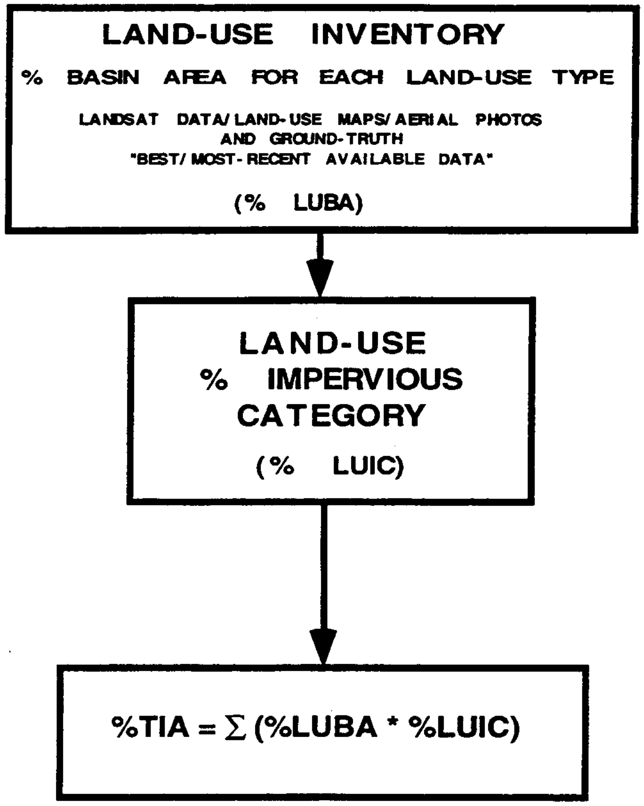

Figure 4: Total imperviousness calculation block-diagram.

TABLE 3: Land-use to % impervious conversions for PSL region.

LAND-USE CATEGORY / DESCRIPTION	TIA*	EIA**
FORESTED	0	0
AGRICULTURAL LAND PARKS / GOLF COURSES / OPEN-SPACE	5	1
LOW-DENSITY RESIDENTIAL RURAL	10	3
MEDIUM-DENSITY RESIDENTIAL SUBURBAN	35	24
HIGH-DENSITY RESIDENTIAL MULTI-FAMILY RESIDENTIAL URBAN	60	53
COMMERCIAL / INDUSTRIAL / MALLS BUSINESS PARKS / EDUCATIONAL FACILITIES	90	86
FREEWAYS / HIGHWAYS (I-5 / I-405 / I-90)	100	100

* Based on King County SWM estimates found in Taylor, (1993).

** Based on estimates from Beyerlein, (1996).

The simplest measure of urbanization would be land area within a watershed designated as either undeveloped (naturally forested) or developed, with the % of each "broad" category used as the level of urbanization. Although simple as well as representative, this method has the obvious drawback of "lumping" all development categories into a single classification, which would obviate the differential effect of land-use intensity. Various development categories and levels of %TIA can also be utilized in combination to form a tailored urbanization protocol. However comparison with previous or related projects would be difficult.

Obviously, there is justification for using an accepted measure of urbanization. Thus, % developed and % forested, in addition to the previously described %TIA, were used here. Specific land-use categories (% urban, % commercial, etc.) were also evaluated to assess cause-effect relationships with stream quality.

Typically, the largest contributors to impervious surface in urbanizing basins are roof-tops and road-surfaces, including roads, parking-lots, and driveways. Traditional zoning regulations emphasize the roof-top component of imperviousness, usually at the expense of roads. Housing density is normally regulated, but not road density. The roof-top impervious component is emphasized in the %TIA calculation. However, the road-surface component often substantially exceeds the roof-top component. A typical PSL area of mixed land-use (residential and commercial) may have an average of 60-70% of its impervious surface composed of roads and other paved areas (City of Olympia, 1994). This is a direct result of the rise in per-capita automobile ownership and increased usage of the car as the preferred mode of transportation.

Road Density

The importance of the road-component of imperviousness, is indicated by the wide range of impervious values for the same zoning category, which depends on the layout of streets (Schueler, 1994). Therefore, road-density is also proposed as an alternative, or as a compliment to %TIA as a measure of urbanization. Eaglin and Hubert (1993) found road density and road crossings to be strongly correlated with instream habitat degradation and fish abundance in a forested basin of the western US. Transportation-related imperviousness often exerted a greater hydrological impact on streams than roof-top runoff (Bannerman et al, 1993; Schueler, 1994; Arnold and Gibbons, 1996). Roads and parking-lots are usually directly connected to the storm drainage system and the nearest stream, whereas roof-top runoff tends to drain via more diffuse routes.

Road-density (km of road/km^2 of basin) was measured using the most current road map, aerial photos, or GIS database. The "baseline" road was considered to be a typical residential street. All roads were treated equally with the exception of multi-lane arterials or highways which were counted as a multiple of the baseline road. Variations in road surface materials (i.e. concrete, asphalt, gravel, or dirt) was not considered. Road-stream crossings were identified from aerial photos and field reconnaissance.

Stream Segment Delineation

Study streams were examined on two levels. Watershed-level sampling and surveys primarily included hydrologic variables and chemical water quality constituents. Each stream system was also sub-divided into "segments" for physical habitat and channel morphological characteristics. A combination of aerial photographs, topographic maps, land-use maps, and field reconnaissance were used to delineate stream segments. As was discussed earlier, all stream segments were classified based on a standard set of criteria (Rosgen, 1994) and were field-surveyed to verify current basin conditions. Morphological characteristics were used as the primary criteria for segment delineation. Segments were then sub-divided based on changes in dominant sub-basin land-use, as required. This was necessary based on the objectives of the study to assess the effect of urbanization.

Because of access limitations and logistical considerations, study segments were not always continuous, and in most cases only limited portions of a stream segment were surveyed (survey reach). Survey reach length for each segment varied from several hundred m to several km. Due to the morphologically-based determination of segment length as well as field constraints, statistically-minimum required segment length (sample size) was not determined prior to data collection. Variance for each segment was therefore estimated in order to calculate (with 90% confidence) a statistically adequate, minimum survey reach (Zar, 1984) for each major physical habitat parameter (e.g., LWD and pools). If the actual surveyed reach was less than the statistical minimum length, then that reach was not used in later analyses.

Watershed Characterization

Watershed-level information was derived primarily from the most current USGS topographic maps and aerial photographs. Basin plans, GIS-based maps, and data from cooperating agencies were also utilized. Drainage basin area and watershed boundaries were determined according to drainage patterns and contour lines, in accordance with standard methods (Dunne and Leopold, 1978). A standard "English-grid" was also used to determine sub-basin areas if no other data were available. Stream lengths, valley slopes, and stream gradients were derived from topographic maps using a map-wheel and slope-indicator template. Stream lengths were measured as "logical" extensions of blue lines on topographic maps, in accordance with Dunne and Leopold (1978). This technique was originally proven to be statistically accurate for field conditions by Morisawa (1957) and is the preferred method. This method involves inserting streams into the drainage network wherever there are "V-shaped" contours and extending channels just beyond the last contour.

Drainage Density

Drainage density (DD) was calculated based on natural conditions, using stream channel length indicated by the undeveloped basin topography, and once again, based on the current developed conditions. For the developed or "artificial" condition, roads that provided direct drainage paths for runoff into the stream system and storm sewer outfalls were included in the drainage network. With increased urbanization, there is usually a reduction in the natural drainage system, but a significant increase in overall drainage density due to the artificial component (Graf, 1977). Land-use data were compiled from a variety of sources in an attempt to develop the most accurate and up-to-date picture of the current conditions in each basin. Land-use maps, GIS-based data, data from LANDSAT imagery, and aerial photos were used to compile the data required to calculate the various measures of urbanization.

Riparian Integrity Assessment

Riparian integrity was assessed as the quantity and quality of riparian forest areas primarily from aerial photos. Field surveys were utilized to update the aerial photos where photo coverage was not up-to-date or adequate. The longitudinal integrity of the riparian corridor was determined as the number of significant breaks in the riparian zone/km, including breaks due to roads, trails, utility right-of-ways and storm sewer outfalls, regardless of the type or width of discontinuity.

Riparian buffer width was defined as the lateral distance from the bankfull streambank outward and tabulated as the mean width between right and left banks for each reach. This value was calculated by dividing the area of riparian forest area for the reach by twice the length of stream channel (Barton et al, 1985). A variety of recommended buffer widths exist for stream protection for each stream segment. The length of riparian buffer for each of five categories was determined and converted to a % of the total segment length. The five buffer width categories were:

a) <10 m
b) 10-30 m
c) 30-50 m
d) 50-100 m
e) >100 m

Riparian buffer quality was judged based on dominant forest composition (coniferous, mixed, or deciduous) and successional stage (old-growth, mature, young, shrub, or grass). LWD recruitment potential was scored based on both current instream and riparian conditions as well as the ability to meet future LWD requirements. Overall riparian quality for each segment was scored as either optimal (4), sub-optimal (3), marginal (2), or poor (1). Observed riparian characteristics were also noted. Again, all riparian measurements and conditions were validated in the field to ensure that aerial photo analysis was current and accurate.

Hydrologic Analysis

One of the most pervasive changes in stream environments brought on by urbanization is the increase in both magnitude and frequency of elevated flows. Post-development peak discharges typically increase by several times over pre-development levels with equivalent rainfall (Hollis, 1975), and the frequency of discharges at and close to bankfull levels increases substantially (Leopold, 1968). Urban development results in numerous new peak flows from small storms that previously produced little runoff (Booth, 1991). These conditions represent greater exertion of stream power after compared to before development. Being the amount of work done by the flow on the channel per unit time and a function of flow rate (Leopold, Wolman and Miller, 1992), stream power is a direct expression of the capability of elevated flows to erode the channel, stress habitat features and biota, and move living and non-living material.

The fundamental importance of stream power and recognition of its overall and event-by-event relative increase with urbanization led this research to choose relative stream power as a key hydrologic variable to explore. Relative stream power was expressed as the ratio of two-year peak flow to average winter baseflow (2-year peak:winter base). This ratio represents the short-term maximum increase in physical stress experienced by the channel and its habitats and biota over the hydrograph during a significant but relatively frequent event. The two-year frequency flow was chosen because of its rough general approximation of the dominant discharge, the discharge level equivalent in its effect to the range of discharges that govern the shape and size of the channel (Leopold, Wolman and Miller, 1992; Gordon, McMahon and Finlayson 1992). Since the majority of the streams in the study were not gauged, several methods of modeling stream hydrology were utilized to make estimates of the 2-year peak:winter base ratio: (1) Hydrologic Simulation Program - FORTRAN (HSPF), (2) King County Runoff Time Series (KCRTS), and (3) linear regression analysis.

HSPF is a continuous-simulation hydrologic model that generates stream flows accounting for surface, shallow subsurface, and groundwater flows. Soils, land cover, and slope are the three primary characteristics used to determine the hydrologic response of a system. The HSPF parameters used in western Washington classify soils as till, outwash, or wetland. Land cover categories available in HSPF include forest, grass/pasture, impervious, and saturated. Slopes are grouped into three categories; flat (0-6%), moderate (6-15%), and steep (>15%). A watershed is divided into subbasins and modeled as a number of discontinuous pervious and impervious land segments. The model is first calibrated against observed flows by adjusting model input parameters until the calculated model flows match the observed stream flows. The model is then verified by simulating discharge and checking it against an actual flow record to determine how well the model predicts observed flow.

KCRTS (King County, 1995b) was developed using HSPF and is also a continuous hydrologic model. KCRTS uses runoff files that contain runoff information simulated by HSPF for a variety of land cover conditions and soil types in King County. A shortened runoff file was developed using precipitation from eight water years, chosen as a good representation of the full record. Information on soil type and area of different land uses within a watershed is entered into KCRTS by the user. The time series created by KCRTS can then be analyzed for peak flows of different recurrence intervals. KCRTS is much simpler and faster to use than HSPF, yet it still gives the benefit of continuous hydrologic modeling.

Statistical linear regression equations are widely used to predict flood flows of a given recurrence interval at ungauged stream sites (Cummans et al., 1975; Panu and Smith, 1989). Using multiple hydrologic variables in the regression equation reflects the combined effect of several variables on flood peaks (Benson 1964). The best hydrologic variables to use vary by region, and can be determined statistically in the regression procedure. The stepwise regression method is recommended for choosing variables to include in a multiple linear regression (Draper and Smith 1981). This method inserts variables into the equation one at a time, determining the order by using the partial correlation coefficient to examine the importance of variables not yet in the equation (Draper and Smith 1981).

Hydrologic Modeling Methods

KCRTS was the primary hydrologic model used to obtain 2-year peak flow estimates. KCRTS was developed for use in King County, and the rainfall files in KCRTS are specific to locations in King County. KCRTS was also used, however, to generate flows for the Snohomish County streams, all of which are located very near King County. Table 4 gives the modeled 2-year peak flows, the method used to model each, and information used to justify the accuracy of modeled flows.

KCRTS was also used to generate 2-year peak flows for several streams in Kitsap county. The KCRTS scaling factor was adjusted based on 2-year, 24-hour precipitation data to account for stream location (Miller et al. 1973). Differences in rainfall intensities, dry periods before storms, and storm durations between King and Kitsap Counties were a concern. Scaling factors for storm sizes other than the 2-year, 24-hour storm were calculated to check for differences in rainfall intensities. All gave similar values to the 2-year, 24-hour storm. KCRTS-generated flows were compared with known flows at two sites, where information was available, to check for other differences between locations. Big Beef Creek in Kitsap County had gauge information available. The KCRTS-generated flow was 75% of the 2-year peak flow determined from gauge data. Percival Creek site #1, in the City of Olympia, was also checked. The KCRTS generated flow there was 68% of the HSPF-generated flow.

KCRTS did not model flows accurately for watersheds with high levels of urbanization. KCRTS assumes that all storm water flows directly to the stream channel at a rate that accounts only for different levels of surface roughness. While this assumption gave satisfactory results for basins that were not highly developed, using KCRTS for basins with high levels of impervious area produced higher flows than actual. In reality, most precipitation would not land on an impervious surface and flow directly into the stream channel. It would be stored in depressions, constructed storage areas, and behind culverts on its way to the channel. The modeled flow was higher than expected when these different types of storage were ignored. Alternative modeling methods were used for basins with high impervious areas (Kelsey, Juanita, Thornton, Schneider, and Des Moines Creeks).

Table 4. Two-Year Peak Flows for Puget Sound Lowland Study Streams Estimated by Hydrologic Modeling, with Data to Verify Accuracy

Stream	Site	2-Yr. Flow (cms)	Model Used	Justification Information	Source
Big Anderson	BA1/A	6.0	KCRTS*		
Big Bear	BB1 BB2 BB3 BB4/A	4.2 4.4 5.0 7.6	KCRTS	HSPF 2-year flows (1985-build out): Site 1/2/3 => 3.7-5.4 cms Site 4/A => 6.4-8.7 cms	(King County 1989)
Big Beef	BF1/A	12.1	KCRTS*	Gauge data (1971-1994), 13 years of data, not continuous 2-year peak flow = 16.1 cms	(USGS 1996)
Carey	CA1/A	5.0	KCRTS	HSPF 2-year flows (1989-build out):4.8-5.6 cms	(King County 1994)
Coal	CL1/A	7.7	KCRTS	HSPF flows (1983-build out): 1.5-year flow: 3.7 - 7.2 cms 25-year flow: 10.3 - 14.7 cms (KCRTS 25-year flow = 12.9 cms)	(City of Bellevue 1987)
Covington	CV1/A	6.3	KCRTS	Gauge data 2-year flow = 7.2 cms (Site 1/A is above gauge site)	(King County 1995c)
Des Moines	DMA	4.8	HSPF		(Hartley 1996)
Jenkins	JEA	4.0	HSPF		(King County 1995c)
Juanita	JU1 JU2 JU3/A JU4	3.7 5.2 5.3 5.4	Regression	Gauge data at mouth (1980-1990) 2-year flow = 6.9 cms	(USGS 1996)
Kelsey	KE1/A	6.0	Regression	Gauge data on Mercer Creek (1980-1994) 2-y flow =11.7 cms (Kelsey Creek at site 1/A drains 50% of the watershed upstream of the Mercer Creek gauge)	(USGS 1996)
Little Anderson	LA1	4.5	KCRTS*		
Little Bear	LBA LB1/X LB3/B LB2 LBC LB4/D	2.3 2.9 5.9 6.5 9.5 9.7	KCRTS		

* Located outside of King County. The KCRTS scale factor was calculated based on 2-year, 24-hour precipitation data.

Table 4. Two-Year Peak Flows for Puget Sound Lowland Study Streams Estimated by Hydrologic Modeling, with Data to Verify Accuracy (Continued)

Stream	Site	2-Yr. Flow (cms)	Model Used	Justification Information	Source
North	NOA NO1/X NOB NOC	8.1 12.4 15.0 19.6	KCRTS	HSPF 2-year flows (Site C is below HSPF site): 1985-86 => 12.7 cms 1989-90 => 19.5 cms Full build-out => 22.1 cms	(Snohomish County 1994a)
Percival	PE1 PE2	2.5 10.3	HSPF	HSPF 2-year flows (1990-build out): PE1: 2.3 - 2.7 cms PE2: 8.8 - 11.1 cms KCRTS* for PE1: 2-year flow = 1.7 cms (68% of HSPF flow)	(City of Olympia 1993)
Rock	RO1/A	2.3	HSPF		(King County 1993)
Schneider	SC1	0.8	Regression		
Seabeck	SE1	5.4	KCRTS*		
Stavis	ST1	5.9	KCRTS*		
Swamp	SWA SW1/B SWC SWD	11.7 12.9 22.5 23.3	KCRTS	HSPF 2-year flows (1990-build out): Site 1/B: 9.9 - 13.0 cms (Site 1/B is below HSPF site) Site D: 20.4- 24.1 cms (Site D is above HSPF site)	(Snohomish County 1994b)
Thornton	TH1 TH2 TH3 TH4/X	2.2 4.0 1.8 8.8	Regression		

* Located outside of King County. The KCRTS scale factor was calculated based on 2-year, 24-hour precipitation data.

HSPF data were available from previously prepared basin plans for many of the study streams (see Table 4). The 2-year peak flow values generated by HSPF were used when they were available and when KCRTS did not generate accurate flow values. For example, KCRTS consistently generated higher peak flows for Rock Creek than the HSPF model predicted. There are two diversions of water from Rock Creek that were not accounted for by KCRTS but were by HSPF. In other cases, flow values generated by

HSPF were used to verify flows generated by KCRTS. After the accuracy of KCRTS was verified, it could then be used to model flows for streams without verification data.

A stepwise linear regression was used to generate 2-year peak flows when HSPF information was not available and KCRTS was not applicable. The regression equation was calculated using SPSS statistical software. Independent variables entered into SPSS as possible predictors of 2-year peak flow included basin area, %TIA, effective impervious area (%EIA), mean annual precipitation, % till soils, % wetland soils, and % forest cover. Basin area and % till were the best predictors of 2-year peak storm flow. The equation generated through the linear regression procedure is as follows, with Q_2 in cms, and basin area in square kilometers:

$$Q_2 = 0.30*(\text{basin area})^{1.18}*(\%\ \text{till})^{2.78}$$

The four streams with mainly outwash soil (Jenkins, Covington, Rock, and Percival [upper part of the basin]) were not used to generate the regression equation. Outwash soil is more permeable than till soil, and less water in the outwash basins is expected to become runoff.

Inputs for KCRTS modeling and linear regression analysis include land use, %TIA, %EIA, soil type, and wetland area. Land use, %TIA, and %EIA were established as described earlier. Soil types for each basin were taken from U. S. Department of Agriculture soil surveys (Debose and Klungland, 1983; Snyder et al., 1973; Pringle, 1990), with the exception of the Kitsap County streams, which were from the Upper Hood Canal Watershed Report (Kitsap County, 1993). Each specific soil type was classified as either till or outwash (King County, 1995b). The percentage of till or outwash soil was assumed to be evenly distributed between the different land use types in the watershed. Except for King County, wetland areas were measured from maps (Kitsap County, 1993; Snohomish County, 1996; Thurston County, 1993) using a clear grid with equally spaced dots. In King County, The King County Wetlands Inventory (King County, 1991) gave the area of each wetland.

Average winter and summer base flows were calculated from flow measurements taken in this study during 1994, 1995, and 1996. Winter base flow was averaged from all measurements taken between October and April. Summer base flow was averaged from all measurements taken in August and September. Des Moines and Jenkins Creeks were added relatively late to the study, and no field measurements of flow were available for these two streams. Gauge data from the King County Hydrologic Monitoring Report (King County 1995a) were used to calculate average summer and winter base flow for these creeks. Gauge data from the same source were used to calculate average base flows for Kelsey and Juanita Creeks, as these data sets represent many more points than the field measurements taken as part of this study. If base flow measurements were not available at a particular site, the value from the nearest site on the same stream was used. Sites were fairly close together in all cases where this was necessary.

Physio-chemical Sampling and Analysis

Stream-flow data were obtained from cooperating municipalities, the USGS, and by direct measurement using a hand-held flow-meter (Marsh-McBirney Model 2000) according to the standard stream cross-sectional method (Leopold, 1968; Dunne and Leopold, 1978; Gordon et al, 1992). Flow was measured during summer and winter baseflow conditions, as well as during storm periods.

Chemical water quality data were compiled from federal, state, and local agencies, as well as from direct field measurement. Existing data available from these streams and sites are listed in Table 5 and were included in the data base. The following constituents were determined in grab samples during summer and winter baseflow, as well as flow-weighted composites during storm events. Because most existing data were collected without regard to flow, the existing data compiled were limited to baseflow conditions.

(a) Temperature

(b) pH

(c) Conductivity

(d) Total Suspended Solids (TSS)

(e) Alkalinity

(f) Total metals (Cu, Zn, Pb)

Table 5: Summary of Previous Water Quality Monitoring on Puget Sound Study Streams by Local Government Agencies.

stream	station #	flow	velocity	staff	temp	DO	pH	COND	HARD	NO_3	NH3	TP	TSS	TCu	TPb	TZn	TURB	time interval
Big Bear #1	J484							X		X	X	X	X					1/89 -- 6/95
Big Bear #1	0484							*		*	*	*	*					1/89 -- 6/95
Big Bear #2																		
Little Bear #1	0478							X		X	X	X	X					1/89 -- 6/95
Little Bear #1	lblu, lbld		*	*	*	*	*	*	*	*		*	*	*	*	*		9/93 -- 11/94
Little Bear #2	lblu		*	*	*	*	*	*	*	*		*	*	*	*	*		9/93 -- 11/94
North	nclu, ncld		*	*	*	*	*	*	*	*		*	*	*	*	*		5/92 -- 11/94
Swamp	sclu, scld		*	*	*	*	*	*	*	*		*	*	*	*	*		5/92 -- 11/94
Kelsey	D444							X		X	X	X	X					1/89 -- 6/95
Coal	0442							X		X	X	X	X					1/89 -- 6/95
Juanita	0446				X	X		X		X	X	X	X				X	1/94 -- 9/95
Thornton #1	0434							X		X	X	X	X					1/89 -- 6/95
Thornton #2																		
Covington	C320							X		X	X	X	X					1/89 -- 6/95
Carey																		
Rock																		
Green-Cove	GC5	X			X	X	X	X		X	X	X	X				X	1/94 -- 9/94
Green-Cove	GC7	*			*	*	*	*	*	*	*	*	*	*	*	*	*	1/94 -- 9/94
Percival #1	PERC T02	X			X	X	X	X	X	X	X	X	X	X	X	X	X	12/90 -- 9/94
Percival #2																		
Schneider	SCHN	X			X	X	X	X	X	X	X	X	X	X	X	X	X	1/93 -- 9/94
Seabeck																		
Stavis																		
Big Anderson																		
Big Beef																		
Little Anderson																		

X indicates data is available at or near our site.
* indicates data is available on the stream, but not at our site.

(g) Dissolved filtered metals(Cu, Zn, Pb)

(h) Nutrients - total and soluble reactive phosphorus (TP, SRP) and total nitrogen and nitrate (TN, NO_3-N)

(i) Dissolved Oxygen (DO)

Sediment samples were also gathered during low flow periods in the summer and fall of 1994 and analyzed for metals and petroleum-based hydrocarbons. Samples were collected, with a glass scoop to prevent contamination, in the top 10 cm from five pools upstream and downstream from each site.

Flow-weighted composite samples were gathered during storms of different magnitudes during the fall/winter storm seasons from 1994-1996 (Bryant, 1995). All laboratory-based chemical water quality analyses were performed by the King County (METRO) environmental laboratory. Analytical procedures and detection levels are shown in Table 6.

Instream Habitat Assessment

Field surveys were conducted during the summer seasons of 1994, 1995, and 1996 to develop a continuous record of reach-level features, including streambank conditions, LWD, and channel morphology. Detailed habitat-level physical characteristics, including pool and riffle and streambed substrata measurements were also determined. The current conditions of each stream segment was evaluated with the objective of relating the variability in stream conditions within the PSL region to the degree of upstream and adjacent watershed development. Details and background information on field sampling/survey procedures are contained in the PSL Habitat Assessment Protocols in May (1996; Appendix B). Habitat characteristics determined are all considered potential indices of urbanization and relate either directly or indirectly to salmonid habitat. Effective indices should be objective, consistent, and relatively sensitive to human disturbance. The characteristics determined include:

1) Riparian canopy closure (% shaded)

Table 6: Summary of Constituents measured in Water (in mg/L) and Sediment (in mg/Kg) in Puget Sound lowland streams, including chemical analysis method and minimum (MDL) and reportable (RDL) detection limits.

Parameters	MDL	RDL	Method
METALS			
Zinc, Total	0.005	0.025	* ICP
Zinc, Dissolved	0.005	0.025	ICP
OTHER			
Alkalinity	0.2	1	** titration method
Total Suspended Solids	0.5	1	** filtered and dried at 103-105°C
Ammonia Nitrogen	0.02	0.04	** automated phenate method
Nitrite + Nitrate Nitrogen	0.05	0.1	** automated cadmium reduction method
Total Phosphorus	0.005	0.01	** persulfate digestion, ascorbic acid method
Ortho Phosphorus	0.002	0.005	** automated ascorbic acid reduction method
SEDIMENT METALS			
Copper, Total	0.4	1.9	ICP
Lead, Total	2.9	14.4	ICP
Zinc, Total	0.5	2.4	ICP

* inductively coupled plasma-emission spectroscopy method, APHA, 1992.
** Standard Methods, APHA, 1992.

2) Riparian buffer quantity and quality (see above)
3) Streambank cover and erosional conditions
4) Riparian land-use and human impact level
5) Stream channel gradient, sinuosity, and confinement
6) Bankfull dimensions (BFW and BFD)
7) Floodplain characteristics (FPW)
8) Streambed substrata size distribution
9) Streambed substrata embeddedness
10) Spawning habitat quantity and quality
11) Rearing habitat quantity and quality
12) Large woody debris (LWD) quantity and quality

Riparian Canopy Conditions

Riparian canopy cover was measured at regular intervals in each stream segment using a standard spherical densiometer (Lemmon, 1957). This device measures angular canopy density and cover, both widely-accepted measures of shading related to light availability and water temperature. Riparian conditions, such as riparian width, dominant species and successional stage and noticeable human disturbance directly adjacent to the study reach were also recorded.

Channel Morphology

Stream channel morphologic characteristics, including bankfull dimensions, stream gradient, and floodplain character were recorded at regular stream-length intervals. Stream channel gradient was measured at 100 m intervals for each sample reach using a standard clinometer and values were checked against topographic maps. Stream channel confinement, whether due to natural geomorphic features or human intervention, was classified as unconfined, moderate, or confined. Channel sinuosity was also indexed as high, moderate, or low through the use of standard compass bearing and stream-length measurements. Bankfull width and depth (BFW and BFD) were measured every 50 m and averaged to give a dimension of the active channel for each sample reach. Flood-prone width (FPW) was also measured and averaged for each reach (Dunne and Leopold, 1978; Gordon et al, 1992; Rosgen, 1994).

Streambank conditions were also observed in order to quantify streambank erosion using a combined streambank stability index. Stream segments with >75% of the reach classified as stable were given a score of 4. Segments with 50- 75% stable banks were scored as 3, 25-50% as 2, and <25% as 1. Artificial streambank protection (rip-rap) was considered a sign of bank instability and scored as 1.

Wolman (1954) described a method for sampling streambed material which is quick, statistically-reliable, and provides information on the substrata particle size distribution and has been modified for fish habitat assessment (Kondolf and Li, 1992). The US Forest Service (Bevenger and King, 1995) modified the Wolman pebble-count to characterize a stream reach along a continuum of habitat features (pools, riffles, and glides) in order to detect the shift in substrata size distribution toward fine sediment in affected compared to reference stream reaches. Samples from several reference reaches are recommended so that reference condition variability can be well defined. However, a limited number of undeveloped, reference streams were available here. Substrata particle embeddedness was also measured as a simple average % for each stream reach (see May, 1996; Appendix B).

Streambed Scour and Aggradation

Several indicators were used to evaluate salmonid spawning and egg incubating habitat quantity and quality. Stream reaches were identified as likely salmonid spawning areas based on past spawning surveys and suitable habitat features. With the assistance of fisheries biologists, the optimal pool tailouts and low-gradient riffles were identified as potential spawning habitat for coho salmon and cutthroat trout. Direct measurement of embryo mortality was not feasible, therefore physical measurements were used to indicate indirect effects of fine sediment deposition via intragravel DO and streambed instability via scour and fill caused by urbanization.

Streambed scour and fill, which can be a major cause of mortality for incubating salmonid embryos (Nawa and Frissell, 1993), was determined along transects across identified spawning habitat areas. Scour monitors, similar to the original design of Tripp and Poulin (1986) and improved upon by Quinn and Peterson (1994), were installed in the streambed. These devices, also referred to as "sliding-bead" monitors (Nawa and Frissell, 1993), are relatively non-invasive to install and measure the effect of peak flows on the intragravel zone and provide a localized assessment of streambed instability (Figure 5a). This technique measures streambed alteration as discrete scour-fill events during peak flow storm events/periods (Jackson and Beschta, 1982).

Scour monitors were installed in the streambed at close intervals across the transect; normally 1-2 per site due to the small stream size. Peak flows scoured the streambed and exposed each successive "bead" to flow turbulence and bedload movement until that bead(s) was freed from the streambed (Figure 5b). After high flow events, scour and fill were determined by counting the number of balls floating and/or amount of cable buried. The number of previously buried beads that moved to the end of the "tail" wire indicated the maximum depth of scour for that storm event. The amount of post-storm fill was determined by measuring the depth to which the exposed beads and tail were buried. Net change in streambed topography was then calculated (Figure 5b).

The monitors were checked after major storm events, or at least monthly during the incubation period. Scour monitor data were evaluated for potential effect on real or potential salmonid redds. Depth is considered to be controlled by the magnitude and duration of stream discharge and substrata texture. Scour to a depth of 20-25 cm is generally recognized as detrimental to salmonid eggs or embryos (Crisp and Carling, 1989). Shallower scour events, more frequent scour events, and excessive aggradation may also have negative effects on salmonid STE (survival to embryo; Lisle, 1989).

The greatest mortality in salmonid populations occurs between spawning and emergence (Quinn and Peterson, 1994). Fine sediment is a natural component of stream substrata; however, excessive deposition of fines has been identified as a major factor in reduced survival to embryo. The relationship between fine sediment and STE is highly

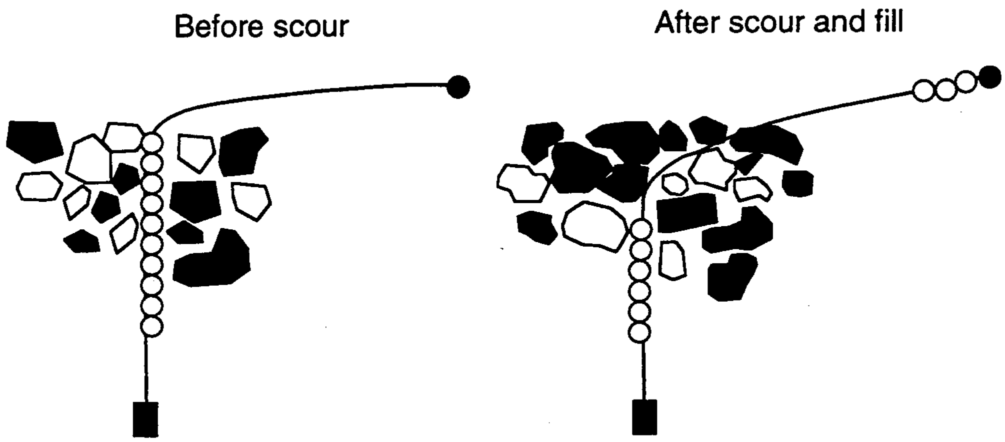

Figure 5a: Sliding-bead type scour monitor(in-situ).

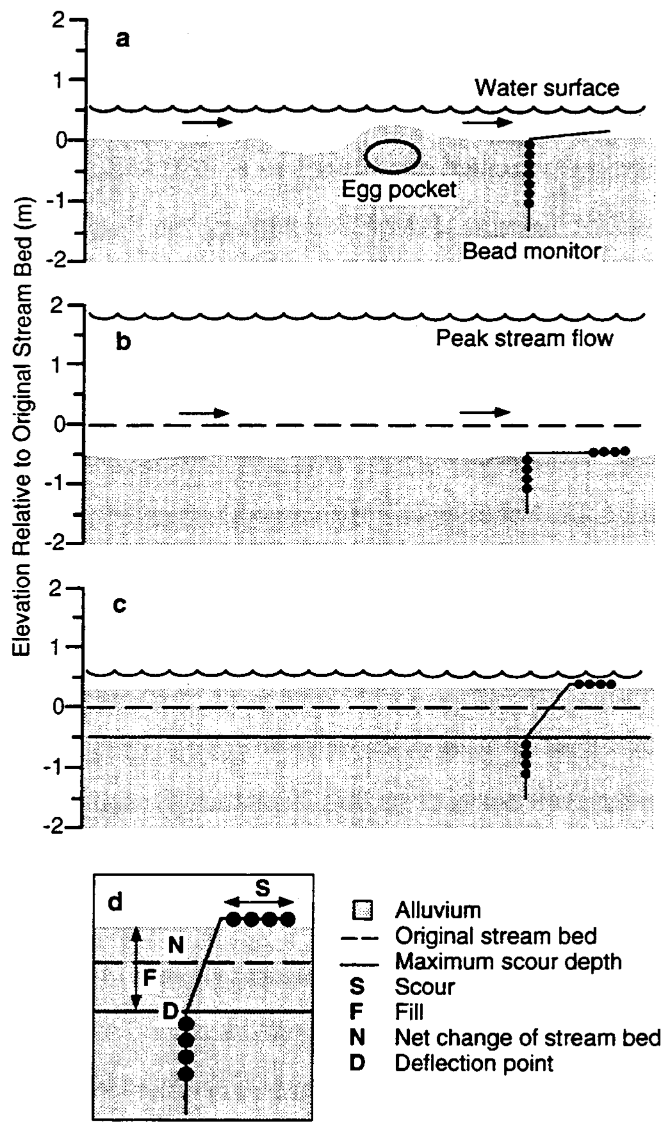

Figure 5b: Streambed scour and fill as measured by a sliding-bead scour monitor.
(a) Scour monitor installed in streambed near salmonid redd
(b) Maximum streambed scour at peak flow during a large storm
* Scoured beads slide down to the end of the wire
* Deep enough scour may wash out salmonid redd
(c) Post-storm sediment aggradation buries scour monitor wire
(d) Measurement of scour and fill (aggradation)
(modified from Nawa and Frissell, 1993)

variable (Chapman, 1988). Also, the relationship has a large slope, such that a small change in % fines results in a substantial change in embryo mortality (Tagart, 1984). Although the exact mechanism of fine sediment-induced mortality is not known, reduced DO has been strongly implicated (Alderdice et al., 1958; Coble, 1961; McNeil, 1966; Koski, 1966 and 1975; Tagard, 1984; Chapman, 1988). Fine sediment has also been well recognized as detrimental to benthic macroinvertebrates (Borchardt and Statzner, 1990; Chutter, 1969; Hynes, 1960).

To investigate the effect of fine sediment on benthic macroinvertebrates, standard McNeil sediment core samples were also obtained for each survey reach and the percent fines (<0.85 mm) was determined (McNeil and Ahnell, 1964). Samples were collected in 1994 and 1996 during summer low flow to ensure that results were unbiased by high flows. Samples were collected from the same or nearest riffles as were those for macroinvertebrates.Two samples were collected from the mid-line of each riffle that satisfied the following criteria: a) covered with flowing water; b) areas without predominantly sand/silt deposits or large substratum particles; or c) areas generally undisturbed.

Samples were sorted through US Standard Sieves with the following sizes: 0.063, 0.125, 0.25, 0.5, 0.85, 1.0, 2.0, 4.0, 8.0, 16.0, 31.5 and 63.0 mm. Smaller size particles have been shown to be insignificant so were discarded. The samples were processed in the field and volumes determined according to McNeil and Ahnell (1964). A detailed explanation of the volumetric processor was given by Olthof (1994). The percent organic material contained in sediment from 0.065-0.85 mm was determined as ash-free dry weight.

Integravel DO (IGDO)

The effect of fine sediment on IGDO and, hence, on spawning and incubating habitat was determined after a technique by Maret et al. (1993). The simple, inexpensive IGDO monitor device consisted of an aquarium air-stone attached to a section of 1/8-inch diameter plastic tubing and sealed with a (bulldog) compression-clip (Figure 6).

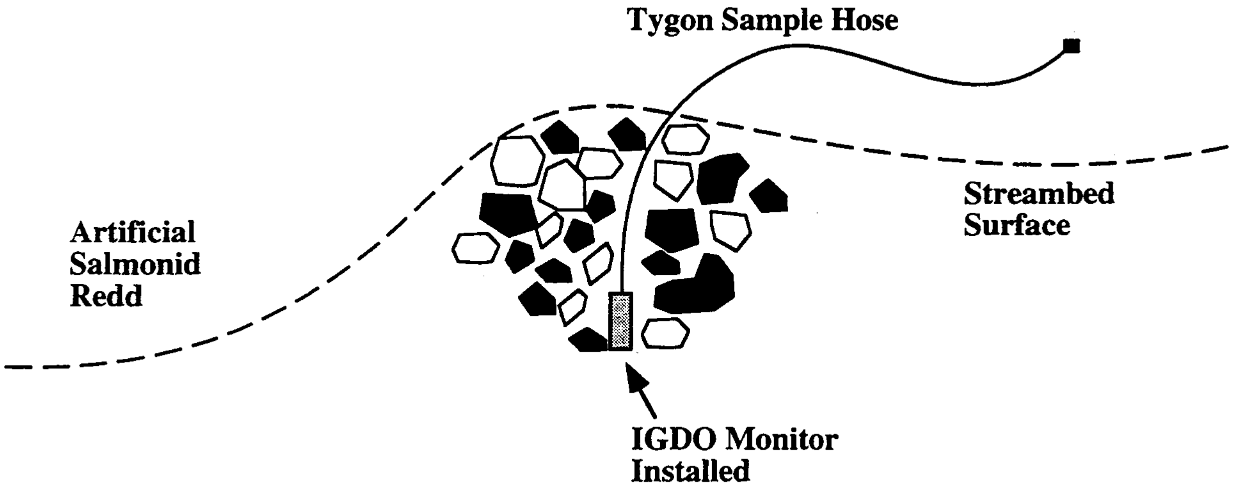

Figure 6: Intragravel dissolved oxygen (IGDO) monitor.

Replicate monitors (at least two/site) were installed in several pool-tailouts in each stream reach in early fall and monitored throughout the normal egg incubation period (October-April). Installation involved digging an artificial redd and burying the device, simulating the construction of an egg-pocket, and then covering the egg-pocket as do female salmonids. This action creates an incubating environment very low in fine sediment and highly permeable to flow and DO exchange (Figure 7). Water samples were drawn monthly using a peristaltic pump attached to a battery-powered drill, which prevented entrainment of atmospheric oxygen. DO was determined by the Winkler method.

Salmonid Rearing Habitat

Stream habitat survey protocols were designed after those used to assess physical habitat in forested streams (Bisson et al, 1982; Lisle, 1987; Hankin and Reeves, 1988; Robison and Beschta, 1990; Peterson et al, 1992; Ralph et al, 1994; TFW, 1994). The three main features were recorded: pools (rearing habitat), riffles (spawning habitat), and LWD. Habitat was assessed during the low-flow period between June and September. Over 100 stream segments were surveyed during the summers of 1994-1996. Each segment varied in length based on various practical and logistical constraints, but at least 25% of each stream segment was sampled. Very long (> 2 km) segments were usually sub-divided and sampled at multiple locations to be representative.

Habitat surface area and types were also recorded (Bisson et al, 1982) and tallied as scour, dam or plunge pools, etc., and high-gradient or low-gradient riffles. Residual pool depths (RPD) were determined according to Lisle (1987). RPD is defined as the difference between maximum and tailout depths, which estimates pool depth at no-flow/drought condition. The amount (%) of cover on each pool was also visually estimated. This percent pool cover was visually estimated based on the amount of over-hanging vegetation, under-cut banks, LWD, and other cover features. The following data were recorded from these observations:

1) # Pools/km
2) Pool-spacing

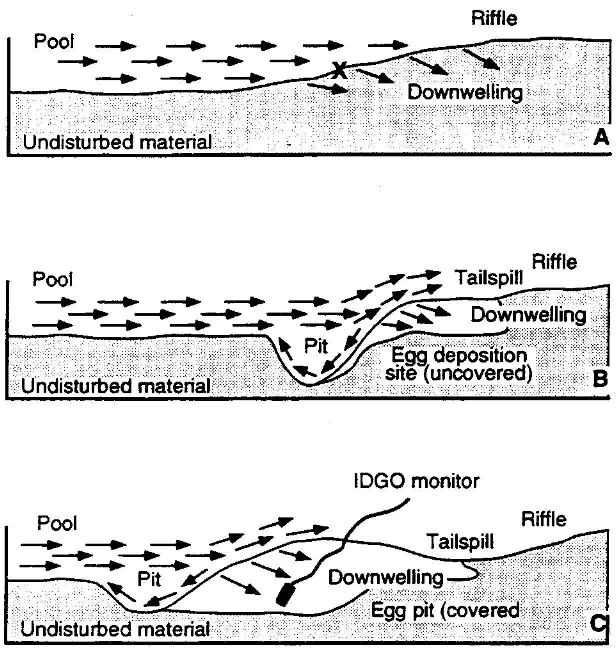

Figure 7: Architecture of a typical salmonid redd with intragravel dissolved oxygen (IGDO) monitor installed.
(A) Streambed topography near pool-tailout. Likely spawning area is marked with "X" (area of flow into gravel)
(B) Redd construction creates a low-flow zone, facilitating egg deposition and fertilization (fine sediment flushed from pocket)
(C) Egg-pocket covered by upstream digging activity and down-welling flow maximized by redd topography. Induced flow flushes fines, provided oxygenated surface water to developing embryos, and removes metabolic wastes.
(modified from Bjornn and Reiser, 1991)

3) Pool area/km and average pool area
4) Mean residual pool depth (RPD)
5) % pool area
6) % riffle area

The number, size, in-channel position, and quality of LWD were also determined within the active BFW channel. LWD was recorded as any organic debris >0.1 m in diameter and > 1 m in length was recorded. Each piece of LWD was measured and its volume calculated, but pieces were often estimated in debris jams. Artificial log weirs and deflectors comprised a portion of LWD in many urban streams and were identified as such. LWD was classified according to (log or root-wad) species (coniferous, deciduous, or man-made), decay condition, stability, and location in the active channel. A LWD quality rating was determined. The following LWD values based on these characteristics were obtained:

1) # LWD/km
2) LWD frequency
3) LWD volume/km
4) Mean LWD volume index
5) % pools formed by LWD
6) LWD quality (position, species, and decomposition)

Qualitative Habitat Assessment

Although quantitative data are relied on most heavily, most agencies responsible for stream assessment in the PNW use some form of qualitative assessment. A qualitative habitat assessment was designed using a combination of indicators (metrics) from existing state and federal survey forms (Plafkin et al., 1989; Rankin, 1989; Plotnikoff, 1993). For this survey, each habitat parameter was graded as optimal, sub-optimal, marginal, or poor (see Appendix B in May, 1996). A score of 4, 3, 2, or 1, respectively, was subjectively assigned for each indicator. Scores for all 15 metrics were summed for a total qualitative habitat index (QHI). A maximum score of 60 was possible and the minimum was 15. Scores were determined by the same person by following the quantitative assessment but before compiling the data, in order to avoid bias. All QHI surveys were completed by the same individual.

Biological Indicators

Coho:Cutthroat Ratio

Data on salmonid fishes were highly variable and influenced by several factors beyond the investigator's control, including climate, oceanographic conditions, fishing pressure, and predation. The level of effort by agencies or groups performing fish-count surveys also affected the results. The most recent salmonid abundance data (specifically for juvenile coho and cutthroat) were obtained from local jurisdictions and tribal fisheries agencies. Data were screened to ensure use of similar methods and an equivalent level of effort. Usable data were available for only 11 of the 120 stream-segments. Juvenile coho typically far outnumbered cutthroat in natural PSL streams (Lucchetti and Fuerstenberg, 1993). Therefore, the ratio of juvenile coho to cutthroat abundance was used as an index of the condition of salmonid habitat.

Because of the difficulty involved with obtaining fish data, benthic (aquatic) macroinvertebrates were chosen as the primary biological indicator. These organisms, composed primarily of aquatic insects, are a main component of the salmonid food-web, are affected by physical habitat disturbance similar to salmonids, are relatively long-lived (~ 1 year), are sedentary, and are easily sampled. For these reasons, the assemblage of aquatic macroinvertebrates integrates the effects of environmental conditions and stress, and are thus, considered excellent indicators.

Macroinvertebrate Field Methods

All invertebrate samples were collected from riffle habitats, based on cobble size, cover, flow, and slope. Macroinvertebrates were collected using standarized sample procedures to obtain a quantitative sample of the macroinvertebrate assemblages existing within the riffle. Each riffle site was sampled one time with three replicate samples using a Surber sampler (500 μm mesh, 300 mm x 300 mm frame, 230 mm bag depth and a detachable sample bucket). Replicate sampling proceeded from downstream to upstream along the mid-line of the riffle—one sample near the tail, one near the middle, and one near the head of the riffle. All rocks within the frame of the Surber sampler were cleaned by hand and removed from the sampling area. A metal spike was used to stir up the substrate to a depth of 10 centimeters beneath the armor layer. Each sample was rinsed through a 1-cm soil sieve into a plastic wash pan to catch large rocks, organic matter and large invertebrates. All invertebrates found in the 1-cm sieve or that remained on the Surber were placed directly into the 250 ml sample jar.

Invertebrates were separated from the sand and gravel by stirring the contents of the plastic wash pan. Floating organic matter was poured into a 500 μm soil sieve then transferred into a sampling jar and preserved with ethanol (95% - residual water in the sample diluted the ethanol to near 70%). Other invertebrates were removed from non-organic residue by visual examination.

Laboratory procedures removed remaining debris. Identifications were made using keys (Merritt and Cummins, 1984; Pennak 1989; Stewart and Stark 1993; Wiggins 1977).

Invertebrates were identified to genus with the following exceptions: Phylum: Nematoda; Order: Pelecypoda, Gastropoda, Oligochaeta, Amphipoda, and Isopoda; Order: Colembolla and Odonata; Family: Chironomidae, Ceratopogonidae, Muscidae, Planariidae; and Class: Hirudinea. All taxa were assigned to functional feeding groups; other classifications used in data analysis included tolerant and intolerant (sensitivity to organic waste and oxygen depletion), long-lived, and clingers using data available in Merritt and Cummins (1984) and Wisseman (1996).

Macroinvertebrate Data Analysis and Metric Testing

Initially, macroinvertebrate data from 1994 were divided into two sets. Data from all but four streams (Big Bear, Little Bear, Juanita, and Thornton Creeks) were used to evaluate the regional applicability of B-IBI metrics used in earlier studies (Kerans and Karr 1994, DeShon 1995, Fore et al., 1996) and to generate a B-IBI applicable to Puget Sound Streams. Data from the four streams with multiple sample sites were used as an independent test of that B-IBI. Because two sites in set 2 (Thornton -3 and Juanita -3) were in the most influenced watersheds in the sample streams, they were included in set 1 for metric testing. Additional metric testing was possible after a second year of data collection and, when integrated with recent studies in other regions, resulted in a 10-metric B-IBI.

Biological attributes must meet several rigorous standards before they can be included in a multimetric biological index: 1) biologically and ecologically meaningful; 2) increase or decrease as the human influence increases; 3) insensitive to sampling variation; 4) distinct separation between the best available sites and the most degraded sites; 5) rank or classify similar locations or situations together (Cook, 1976; Karr et al., 1986; Barbour et al., 1995; Fore et al., 1996).

To determine if a metric meets these criteria each potential metric was plotted against the gradient of urbanization as measured by impervious area. The graphical presentation must demonstrate a quantitative change across a gradient of human influence, indicating a biologically meaningful response to degradation. Least and most degraded sites in this study were judged based on general familiarity with watershed and instream conditions. The least degraded [Carey (CA), Covington (CV), and Rock (RO)] and most [Juanita (JU), Kelsey (KE), and Thornton (TH)] degraded streams were associated with the lowest and highest impervious areas. Thus, we use impervious area to ordinate our study streams along a gradient of human impact in the watershed. For more detail on the method, see Karr and Chu (1995).

Statistical Methods

A variety of graphical and statistical techniques were used in identifying relationships linking watershed conditions, stream characteristics, and biological variables. Microsoft EXCEL spreadsheet software and S+ statistical software were the primary tools used in data analysis. Correlation analysis was performed as appropriate using the Spearman (nonparametric-rank) or Pearson (parametric) techniques (Zar, 1984) on S+ (a significance level of 0.05 was used throughout). Normally testing was done using the Lilliefors test, which is a form of the Kolmogorov-Smirnoff test used when mean and variance of the distribution are unknown (Zar, 1984). Because most variables were not normally distributed, threshold testing was done using the Mann-Whitney or Kruskal-Wallis rank-tests (Zar, 1984). Multivariate (least-squares, linear, step-wise) regression was used to analyze the linkages between measures of basin urbanization and selected chemical water quality variables. The fit of the regression equations was evaluated using the coefficient of determination (r^2) and residual analysis (Zar, 1984). For variables other than chemical water quality, the linkages were not as direct, so regression analysis was not used for defining most relationships.

In order to identify which physical, chemical, and biological variables were the most influential in determining instream conditions and to assist in developing a set of stream quality indices, another form of multivariate statistical analysis was used. A form of partial least-squares (PLS) correlation was used to identify which stream-quality variables best described instream conditions and best reflected the effects of development. PLS is a method of data reduction designed for the analysis of cause and effect in complex systems under indirect observation and is a hybrid of regression analysis, based on the singular-value decomposition (SVD) technique. PLS applies to studies in which cause and effect are each measured variously and redundantly (Streissguth et al., 1993). This method is most useful when multiple "blocks" of indicators are collected and a summary of the predictive interrelationships is desired. Each indicator or metric measures some aspect of the relationship underlying the block as a whole. For this study, the blocks are the urban-impact, physical, chemical, and the biological variables. The relationships consisted of (1) urbanization (cause) and physical habitat (effect), (2) urbanization (cause) and biological integrity (effect), (3) urbanization (cause) and physio-chemical condition (effect), (4) physio-chemical condition (cause) and physical habitat (effect), (5) physio-chemical condition (cause) and biological integrity (effect), and (6) physical habitat (cause) and biological integrity (effect). The goal was not to explain the correlations among indicators of the same block, but instead to describe the linear combination of indicators (or transformed indicators) in each block which predicted the other block (i.e. urbanization metrics were weighted in proportion to their correlation with the sum of either physical habitat, physio-chemical, or biological variables). This analysis technique should identify the most statistically significant stream quality indices.

In addition, each stream quality variable was evaluated individually to assess how well it responded to urbanization level. A useful stream quality variable was one that accurately reflected the stream's response over a range of human disturbance. The individual variables were originally chosen because they were physically, chemically, and/or ecologically meaningful. Each of the potential urban stream indices have usually proven useful in evaluating the effects of forestry practices in streams in the PNW and have been shown to be relatively insensitive to sampling variation (Peterson et al., 1992). To be useful, each variable should display a correlation with the gradient of development. Effective parameters should also be sensitive to changes in intensity and/or specific aspects of urbanization. A "good" stream quality variable must also be able to clearly distinguish between the least and most degraded sites.

After identifying and assessing the individual variables as potential indices, a suite of indices was selected as an overall stream quality index for these streams. A combination of an integrated index of stream quality should provide a more robust index, because no single variable can adequately indicate the full range or variety of disturbance.

RESULTS

Watershed Urbanization

Impervious Area

Basin land-use was tabulated and the percentage of impervious surface area (%TIA and %EIA) was calculated as part of the watershed analysis process. Land-use data for each stream basin are shown in Table 1. Stream segment and sub-basin data are summarized in Table 7. Figure 8 shows the stream basin %TIA, arranged from least to most developed and ranged from undeveloped, reference streams (< 5 %TIA) to those draining highly urbanized basins (> 45 %TIA). For most analyses, development level was considered a continuous variable based on imperviousness (%TIA). For some purposes, data were discretized into land-use categories based on a range of %TIA values. Although the impact of human activities depends on more than %TIA, this was the principal measure of urbanization used here. The %EIA was also calculated for each basin based on typical land-use values for the PSL region. Effective and total imperviousness were found to be highly correlated and therefore %TIA was used as the primary independent variable. Streams will usually be listed in order of their urbanization level (%TIA).

Some variables are best analyzed on a watershed-level, whereas a smaller scale is advantageous for others. Data were analyzed at two levels; 1) watershed or basin-level variables include most hydrologic and chemical water quality variables and, 2) stream-segment-level variables include the bulk of the physical habitat and biological variables. Some overlap does exist between the two levels of analysis. Regardless of the analysis level, the purpose was to relate or link basin-level processes to both physical habitat and biological characteristics, as well as to relate physical habitat characteristics directly to biological indicators in order to evaluate the response of stream variables to urbanization and to identify useful stream quality indices (Figure 9).

Although impervious surface area is the most commonly accepted measure of development intensity, alternative measures were also investigated. The simplest

Table 7: Stream sub-basin characteristics for the Puget Sound lowlands.

STREAM	SEGMENT (Segment #)	STREAM SEGMENT LENGTH(km)	ROSGEN STREAM TYPE	CUM UPSTREAM SUB-BASIN AREA(km^2)	CUMULATIVE UPSTREAM %TIA	BMP QTY (#)	BMP VOLUME (acre-ft)
OLYMPIA and THURSTON COUNTY							
Green Cove	Lower (1)	1.5	B4	10.8	8.1	14	1
Green Cove	Middle (2)	1.0	C4	8.8	8.8	4	1
Green Cove	Upper (3)	2.5	E5	6.3	10.2	6	1
Percival	Lower (1)	1.5	C3	21.2	21.8	66	89
Percival	Middle (2)	1.0	B4	5.8	12.4	5	1
Percival	Upper (3)	2.0	E4	4.7	11.1	2	0
Percival	Black Lake Ditch (4)	4.5	F5	13.1	24.0	13	80
Schneider	Mainstem (1)	1.8	B4	2.7	42.2	4	0
CITY of BELLEVUE							
Coal	Below I-405 (1)	1.0	F5	16.8	20.8	3	0
Coal	Parkway to I-405 (2)	2.0	B4	16.2	20.3	39	1
Coal	Newcastle to Parkway (3)	4.0	C4	13.1	19.0	11	1
Coal	Couger Mt. Headwaters (4)	4.0	A4	5.5	10.3	7	0
Kelsey	Headwaters/Larson Lake (1)	1.5	DA6	1.8	48.0	15	1
Kelsey	Upper Mainstem(2)	0.8	C4	2.5	41.5	3	0
Kelsey	Upstream Bel-Red (3)	0.3	C5	3.0	43.7	2	0
Kelsey	Downstream Bel-Red (4)	0.7	F3	12.3	47.6	20	1
Kelsey	Middle Mainstem (5)	0.8	G4	12.8	45.9	13	1
Kelsey	Lower Mainstem-Golf (6)	1.3	E4	14.3	50.0	3	0
Kelsey	Lower Mainstem-Park (7)	0.8	E5	15.6	49.2	2	0
Kelsey	Lower Valley/Sears Tribs (8)	1.3	E5	6.8	48.0	66	2
Kelsey	Upper Valley Trib (9)	2.2	C4	4.5	27.3	31	1
Kelsey	West/Goff Tribs (10)	3.1	G4	5.6	54.4	67	2
CITY of SEATTLE							
Thornton	Lower Mainstem (1)	2.0	G6	29.7	55.4	4	0
Thornton	South Branch (2)	4.5	B5	10.1	60.6	23	1
Thornton	North Branch (3)	6.5	C5	15.5	52.7	57	2
KITSAP COUNTY							
Big Anderson	Lower Mainstem (1)	1.0	D4	15.5	1.2	0	0
Big Anderson	Upper Mainstem (2)	1.7	C4	13.3	1.2	0	0
Big Anderson	North Branch (3)	5.0	B3	4.5	1.2	0	0
Big Anderson	South Branch (4)	5.0	B3	5.5	1.2	0	0
Big Anderson	Tributary 0413 (5)	1.0	B4	3.3	1.2	0	0
Big Beef	Ravine/Below Lake (1)	12.0	C4	34.3	3.1	0	0
Big Beef	Above Lake Symington (2)	2.0	C4	15.8	4.1	0	0
Big Beef	Headwater Wetlands (3)	6.0	C4	12.0	4.1	0	0
Little Anderson	Lower Mainstem (1)	1.0	DA4	16.1	3.4	0	0
Little Anderson	Middle Mainstem (2)	1.0	B3	15.0	3.4	0	0
Little Anderson	Upper Branches/Tribs (3)	3.5	B4	10.0	3.4	0	0
Stavis	Lower Mainstem(1)	1.0	C4	17.0	1.5	0	0
Stavis	Upper East Branch (2)	8.0	B4	7.0	1.5	0	0
Stavis	Upper West Branch (3)	4.8	B4	6.5	1.5	0	0
Seabeck	Lower Mainstem (1)	1.2	F3	18.3	2.7	0	0
Seabeck	Upper East Branch (2)	1.5	F4	5.0	2.7	0	0
Seabeck	Upper West Branch (3)	5.3	B4	7.0	2.7	0	0
Seabeck	Seabeck Heights Tributary (4)	2.0	G4	4.0	2.7	0	3
KING COUNTY							
Rock	Lower (1)	2.3	B3	33.6	3.2	0	0
Rock	Upper (2)	6.0	C4	29.6	3.1	0	0
Juanita	Lower (1)	2.5	B5	18.4	45.4	3	40
Juanita	Upper (2)	4.5	B4	12.0	44.3	3	56
Covington	Lower/Soos Creek(1)	2.0	B3	57.0	3.9	1	8
Covington	Middle/Black diamond Rd (2)	3.0	C4	49.8	3.9	0	6
Covington	Below Lake Sawyer (3)	5.0	C5	39.8	3.9	0	4
Covington	Above Lake Sawyer (4)	5.0	E5	22.8	3.2	0	2
Carey	Lower (1)	4.0	C4	20.2	1.3	0	0
Carey	Upper (2)	7.0	B4	10.4	0.8	0	0
Big Bear	Lower Mainstem/HWY-520 (1)	1.5	F6	89.3	10.9	2	4
Big Bear	Lower Mainstem/Redmond (2)	1.0	F6	86.1	9.6	4	9
Big Bear	Lower Mainstem/Evans (3)	1.0	E5	83.5	9.3	2	3
Big Bear	Lower Mainstem/Novelty Hill (4)	3.5	E5	80.1	8.8	1	2
Big Bear	McWhirter Tributary (5)	4.0	G3	76.1	3.9	0	2
Big Bear	Lower Mainstem/Avondale (6)	5.5	F4	72.5	7.8	0	7
Big Bear	Lower Cottage Lake Creek(7a)	3.5	C4	32.1	8.6	0	2
Big Bear	Middle Cottage Lake Creek (7b)	4.0	E5	24.9	7.7	0	9
Big Bear	Daniels Creek (7c)	3.0	E5	19.6	3.1	0	8
Big Bear	Crystal/Headwater Wetlands (7d)	1.5	DA	11.1	2.1	0	2
Big Bear	Lower BB Branch (8)	2.0	C4	37.7	6.6	0	0
Big Bear	Seidel Creek Tributary(9)	3.5	G4	33.8	4.8	0	2
Big Bear	Middle BB Branch (10)	2.0	C4	28.6	5.5	0	0
Big Bear	Struve Creek Tributary(11)	3.0	G4	24.4	4.5	0	2
Big Bear	Upper BB Branch(12)	4.5	C4	18.9	3.3	0	0
Big Bear	Paradise Lake Outlet (13)	5.0	E5	15.0	1.9	0	2
Big Bear	Paradise Valley Wetlands (14)	5.5	DA	10.0	1.2	0	8

Table 7: Stream sub-basin characteristics for the Puget Sound lowlands.

		STREAM SEGMENT	ROSGEN STREAM	CUM UPSTREAM SUB-BASIN	CUMULATIVE UPSTREAM	BMP QTY	BMP VOLUME
STREAM	SEGMENT (Segment #)	LENGTH(km)	TYPE	AREA(km^2)	%TIA	(#)	(acre-ft)
KING COUNTY (con't)							
Jenkins	Lower (1)	2.0	G3	41.3	13.1	0	44
Jenkins	Middle (2)	7.0	DA4	20.0	13.1	0	44
Jenkins	Upper (3)	2.0	B3	6.5	13.1	0	44
Jenkins	CranmarTrib (4)	6.0	E4	15.3	13.1	0	44
Miller	Lower Mainstem/Below STP(1a)	1.0	C4	16.5	49.4	0	0
Miller	Lower Mainstem/Above STP(1b)	5.0	B3	14.8	49.4	1	12
Miller	Walker Creek Tributary (2)	4.0	C4	4.3	42.6	2	23
Miller	Middle Mainstem(3)	5.0	C5	10.8	49.3	2	24
Miller	Upper/Headwaters (4)	5.0	E5	6.5	52.9	4	42
DesMoines	Lower Mainstem/Park (1)	0.6	F4	15.2	49.1	2	0
DesMoines	Middle Mainstem/Below Plant (2)	0.8	B3	13.2	49.8	2	12
DesMoines	Middle Mainstem/Above Plant (3)	1.6	B3	11.9	49.8	2	16
DesMoines	Riparian Wetland @ 200th (4)	0.5	E4	10.1	49.8	2	0
DesMoines	Golf Course/Runway/Fuel Tanks (5)	3.5	F5	9.0	55.0	3	3
SNOHOMISH COUNTY							
Little Bear	Upper/Headwaters (1)	4.0	C4	7.9	4.4	0	4
Little Bear	Middle/Rural (2)	3.0	C4	11.9	4.1	0	0
Little Bear	Middle/Undeveloped (3)	1.0	C4	14.0	3.8	0	0
Little Bear	Middle/Suburban (4)	4.0	F4	18.9	5.1	1	3
Little Bear	Great Dane Creek (5)	4.0	C4	5.9	5.4	0	2
Little Bear	Parallel to SR-9 (6)	2.0	F4	29.2	9.9	3	5
Little Bear	Rowlands Creek/KC (7)	1.5	F4	33.2	10.0	0	1
Little Bear	HWY-522/KC (8)	2.5	F5	35.4	12.0	2	5
Little Bear	Lower/Woodenville/KC (9)	3.0	F5	37.6	13.8	3	23
North	Everett/Headwaters (1)	1.5	E6	4.6	48.5	5	3
North	Upper/Above I-5 (2)	3.0	C5	8.1	44.4	4	3
North	McCollum/Below I-5 (3)	2.0	C4	10.1	41.0	1	1
North	Mainstem/Above Mill Crk (4)	5.0	C4	17.4	34.5	0	5
North	Wetland/Below Mill Crk (5)	1.5	C5	21.9	29.2	0	12
North	Silver Lake (6a)	3.0	E6	4.8	36.8	1	10
North	Ruggs Lake (6b)	4.0	E6	7.3	32.2	1	1
North	Thomas Lake (6c)	6.0	E5	10.9	25.5	0	6
North	Upper Penny Creek/Golf (6d)	4.0	E5	12.3	25.4	0	9
North	Lower Penny Creek/Res (6e)	3.0	B4	13.7	25.4	0	3
North	Middle Mainstem (7)	3.0	C4	44.7	25.7	0	20
North	Upper Silver/Tambark (8a)	5.0	B4	4.5	16.2	0	0
North	Middle Silver/Tambark (8b)	3.0	B4	8.5	16.2	2	17
North	Lower Silver/Tambark (8c)	3.0	F5	10.5	16.2	2	11
North	Lower Mainstem (9)	5.0	E4	70.7	26.2	2	17
North	Lower/KC (10)	3.0	F4	72.4	26.4	3	5
Swamp	Everett Headwaters (1)	5.0	C5	6.0	32.8	2	15
Swamp	Below Lake Stickney (2)	4.0	C5	15.1	29.3	1	4
Swamp	Wetland Above I-5 (3)	3.0	C5	20.7	26.2	0	2
Swamp	Butternut/Parallel I-5 (4)	2.0	C4	21.9	25.8	0	2
Swamp	Mainstem Below I-5/405 (5)	4.0	C4	26.6	24.4	0	14
Swamp	Martha Lake Creek (6)	5.0	C5	31.5	20.9	0	10
Swamp	Lower Scriber Creek (7)	5.0	E6	18.6	48.8	3	15
Swamp	Upper Scriber Creek (8)	4.0	F5	10.0	54.7	4	22
Swamp	Mainstem/Below Scriber(9a)	3.0	C4	51.5	32.4	1	10
Swamp	Middle Mainstem/Wallace(9b)	1.0	C4	56.0	31.5	1	12
Swamp	Lower/KC (10)	4.0	F5	61.6	31.1	3	24

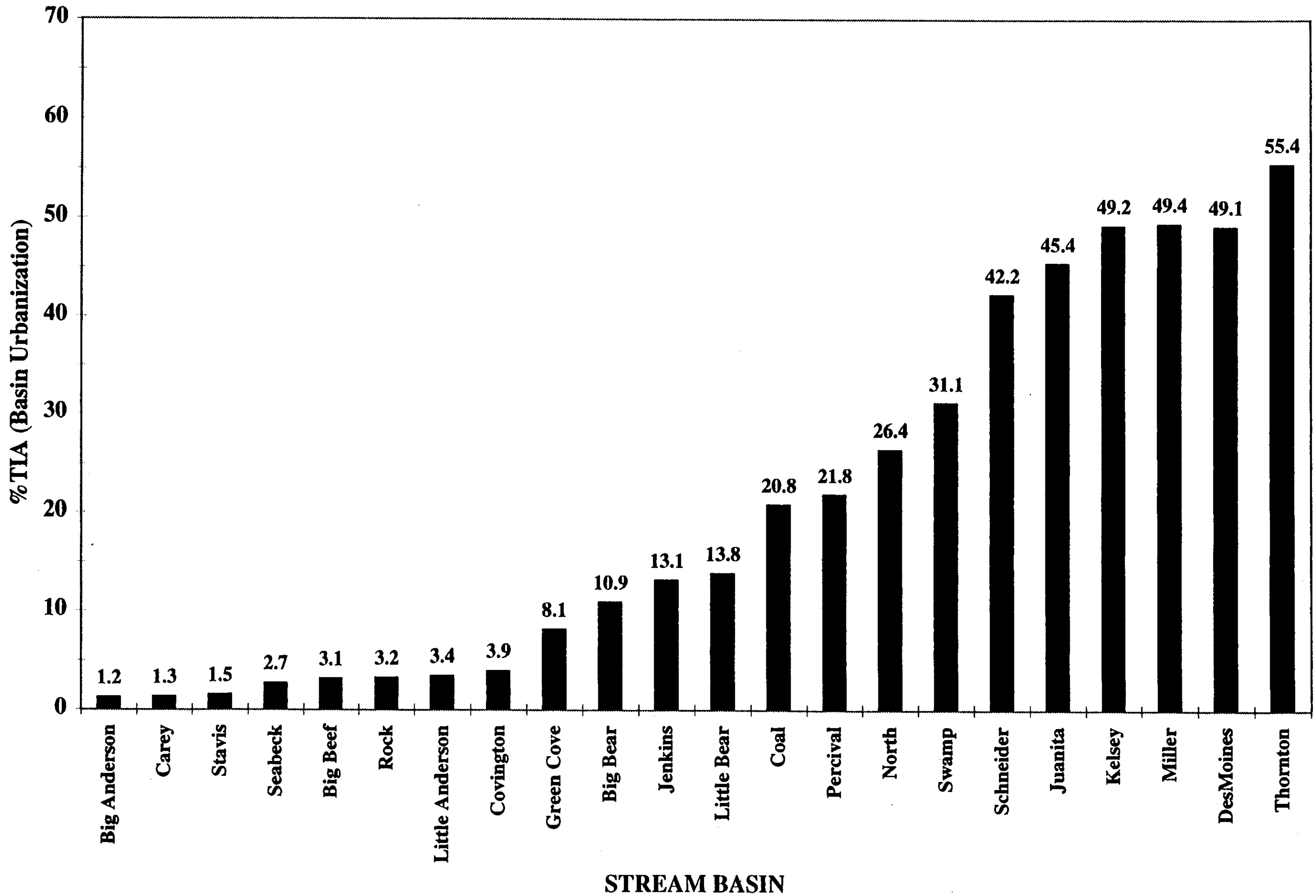

Figure 8: Watershed total impervious area (%TIA) for Puget Sound lowland study streams.

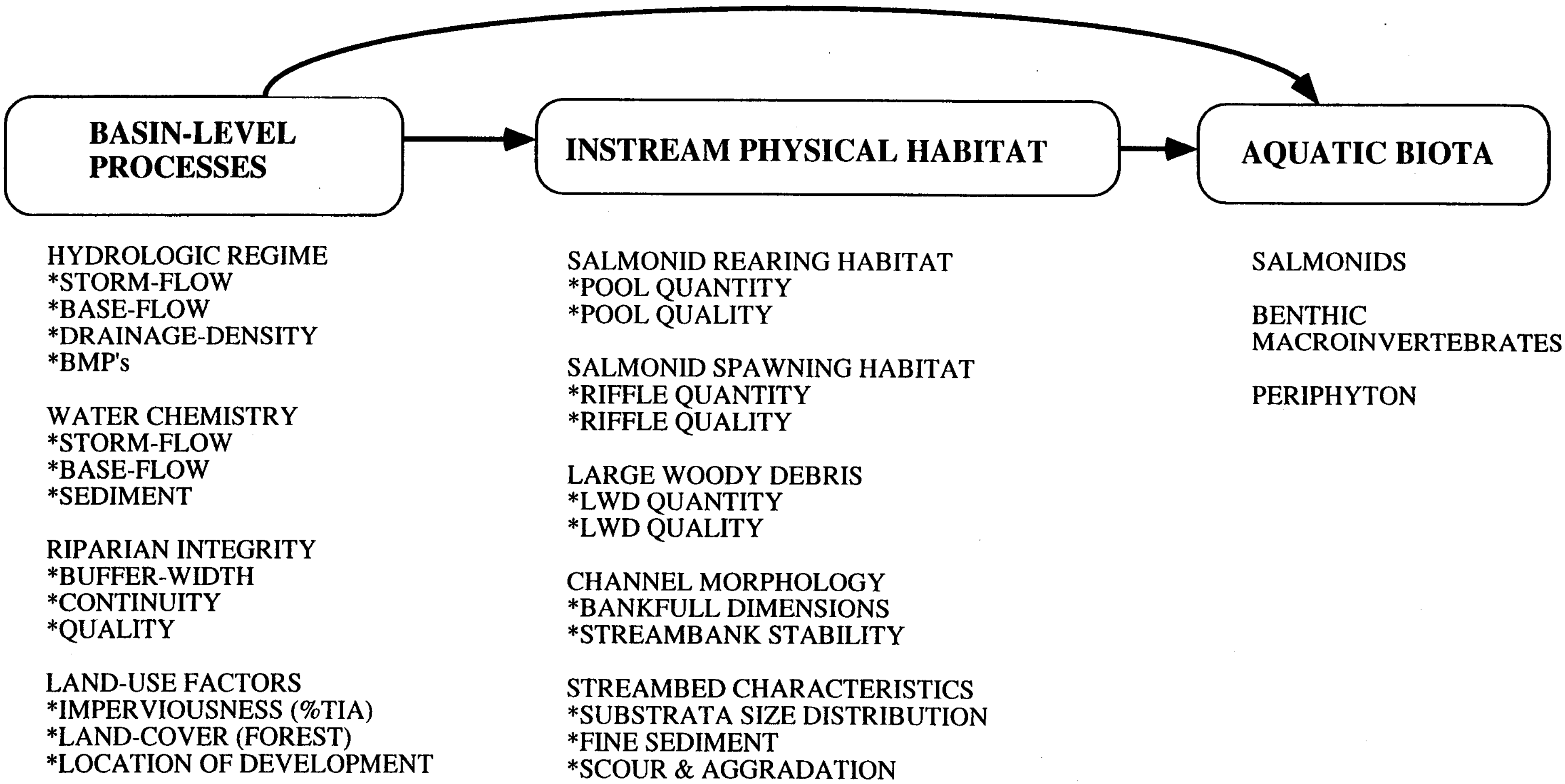

Figure 9 : Conceptual relationships among variables analyzed

measure is the fraction of the watershed developed. Any human activity that involved land-clearing or replaced the natural vegetation was considered development. Land-use categories included as developed are parks, agricultural or pasture lands, golf courses, as well as impervious surfaces. As expected, the % of basin area developed was strongly related to %TIA. A direct relationship between development and imperviousness has usually been found; for these watersheds the relationship was:

$$\textbf{\%TIA = 0.59*(\% Developed)-11.17} \quad [r^2 = 0.95]$$

These results indicate that a simple measurement of developed area may be adequate to characterize urbanization. However, a more detailed land-use characterization may be necessary for some purposes, such as hydrologic modeling.

Road Density

Road density is another alternative to imperviousness as a measure of urbanization. This measure is particularly relevant, because transportation-related activity affects a large fraction of urbanized area. Road-density was strongly related to the level of development or imperviousness either on a watershed level (Figure 10a) or on a stream-segment scale Figure 10b). Results illustrate that a range of road densities existed during the initial phases of development. For streams with a %TIA <10%, the road-density ranged from 1 to 5 km/km^2 of basin. This indicates that there is some flexibility in initial integration of roads into the development process and, thus, represents a potential for impervious surface reduction.

In addition to road-density, the presence of a major highway in the stream catchment is also indicative of higher levels of development (Table 1). A major arterial was present in all but one of the most highly developed watersheds. While an obvious part of all development (i.e. more people necessitates more and "better" road systems), this observation suggests that roads may be related to watershed and stream degradation. Along with more roads in a watershed comes more road-crossings (bridges and culverts). Each road-crossing breaks the continuity of the riparian corridor and allows an

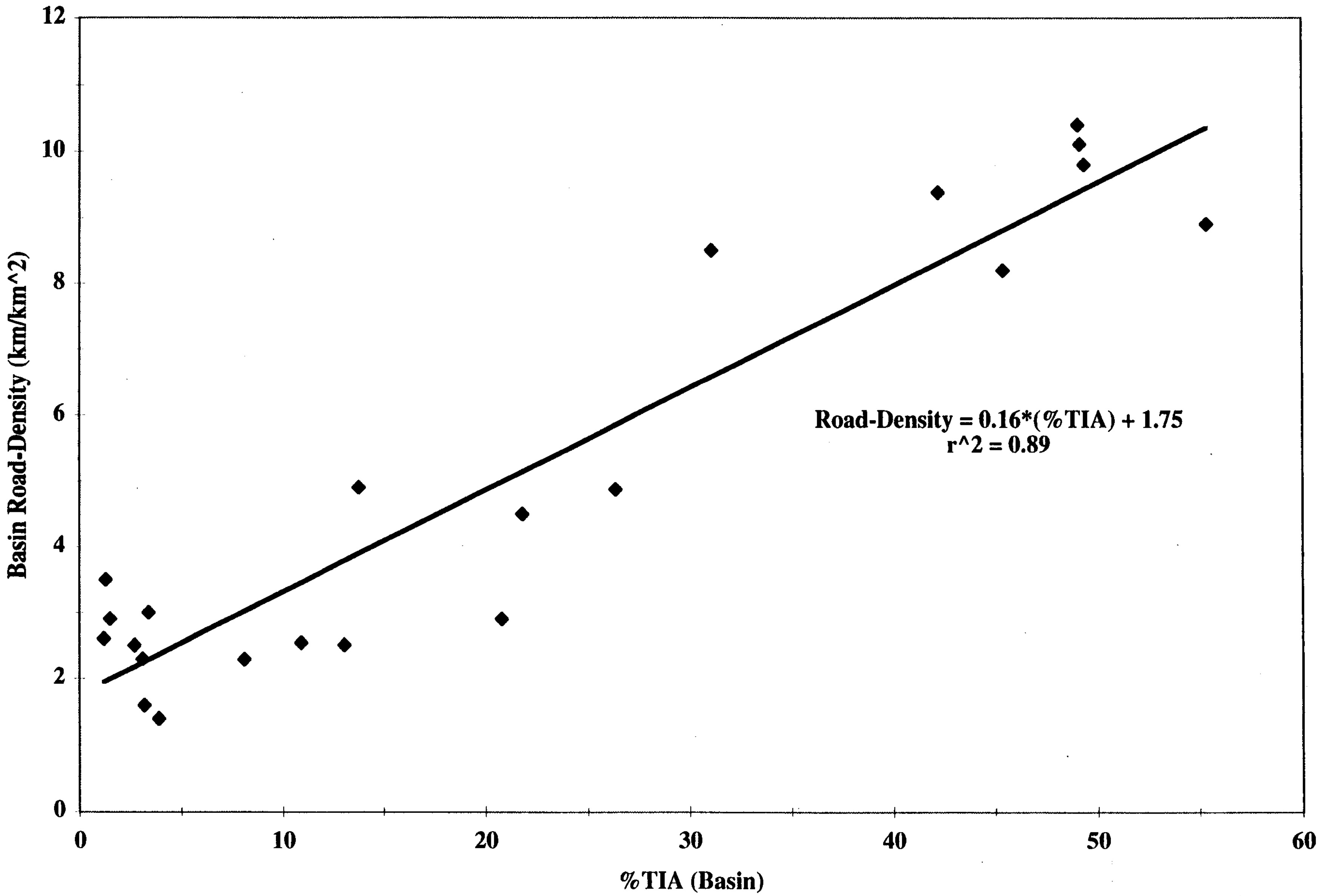

Figure 10a: Relationship between urbanization (%TIA) and road-density in Puget Sound lowland streams (watershed scale).

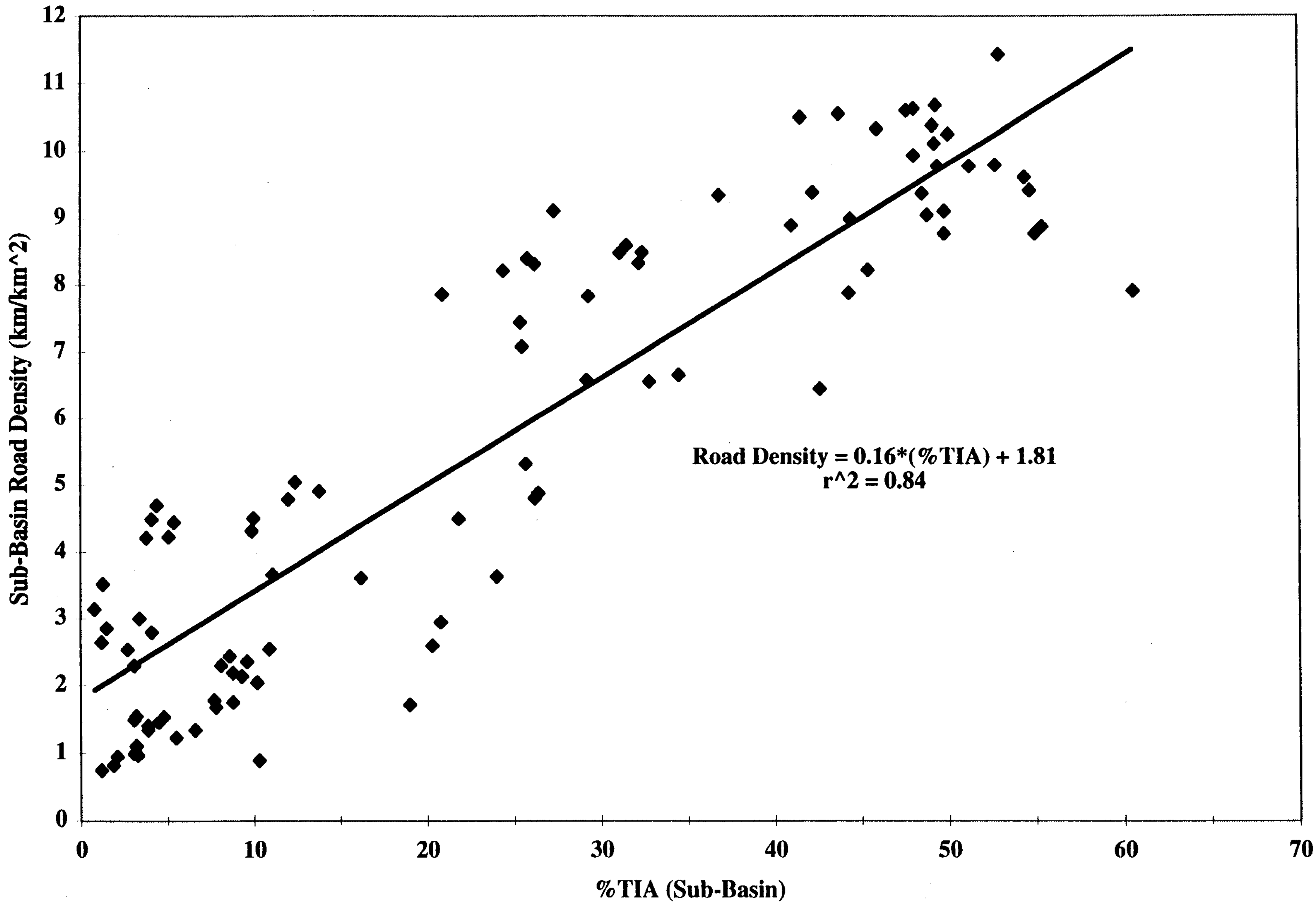

Figure 10b: Relationship between urbanization (%TIA) and road-density in Puget Sound lowland streams (sub-basin scale).

opportunity for stormwater runoff to flow directly into the stream via roadside ditches. With few exceptions, road construction keeps pace with development and, because roads are directly connected with a stream network, may be the critical component of the hydrologic problem.

Forested versus Developed Area

The fraction of natural, forested land remaining is also a measure the level of human activity within the basin. The natural wetland area was also included in the % forested category. Percent forested is the inverse of % developed and is a useful indicator of urban impact, as well as a measure of the amount of "natural" land-cover remaining. The % of natural forest cover remaining reflects the loss of natural vegetative cover to intercept and infiltrate precipitation. For the PSL streams, the point at which imperviousness overwhelms the natural vegetative cover appears to occur when forested area drops below 30% of the total watershed area. That typically occurs when the basin shifts from rural-dominated land-use to mostly suburban development. At this point there is usually a dramatic rise in commercial use and multi-family housing becomes significant. As with impervious surface, the location as well as the magnitude of forest area is important. In general, those basins with a high level of imperviousness concentrated in their headwaters, had more water quality problems and lower overall stream quality.

Basin Hydrologic Regime

Watershed Drainage Density

Analysis of the drainage-density (DD) for each watershed showed some significant differences between the natural, pre-development, and the developed DD. Basin area ranged from approximately 3 km^2 to 90 km^2, while that for natural DD was only 0.3-1.3 km/km^2. However, there was no clear relationship found between either basin area or %TIA and natural DD. The developed basin DD was only slightly correlated with %TIA. However, the ratio of urban to natural DD, which normalized for the change in DD with development, indicated a strong linear relationship with imperviousness (Figure 11a). This implies that the DD ratio may be a useful index of basin urbanization and potential hydrologic effect on streams. Figure 11a also shows the inverse relationship between DD and forested area. At %TIA) of 5, which approximates the break between undeveloped streams and those that are beginning to show the effects of urbanization, the DD ratio was less than 1.25 (i.e., <25% increase in DD with urbanization). Under ideal conditions the DD ratio would be unity, indicating no anthropogenic changes in the stream network. At the point where %TIA exceeds forest area (suburban growth zone), the urban DD was more than 50% greater than natural conditions, with one exception. Figure 11b shows the direct effect of the artificial drainage network (roads and stormwater outfalls) on basin drainage density. Road crossings and stormwater pipes funnel runoff directly into the stream channel with little or no infiltration.

The effect of development-driven changes in DD on stream flow were illustrated by comparing DD to predicted flows using several hydrologic simulation models (Cooper, 1996). The results included a estimation of the current 2-year stormflow as well as the mean winter baseflow. A weak relationship between winter baseflow and %TIA was observed, possibly due to increased surface runoff and loss of groundwater recharge. Using these values in a dimensionless ratio (2-year stormflow to winter-baseflow), a strong relationship with %TIA was demonstrated (Figure 12).

Stormflow:Base Flow Ratio

The patterns of response to storm events was markedly different for urbanized and non-urbanized streams. The more urbanized streams generally showed a "flashy" response to storm events. To illustrate this, a fairly typical multi-day storm in January, 1994, was chosen. An antecedent dry period of about one week preceded this storm and the storm rainfall pattern was fairly consistent over the PSL. During this storm, 0.5 inches of rain fell on 1/22/94, followed by 0.3 inches on 1/23/94, with low to moderate (< 0.1 in.) rainfall for the next two days and only trace precipitation after 1/26/94. Figure 13

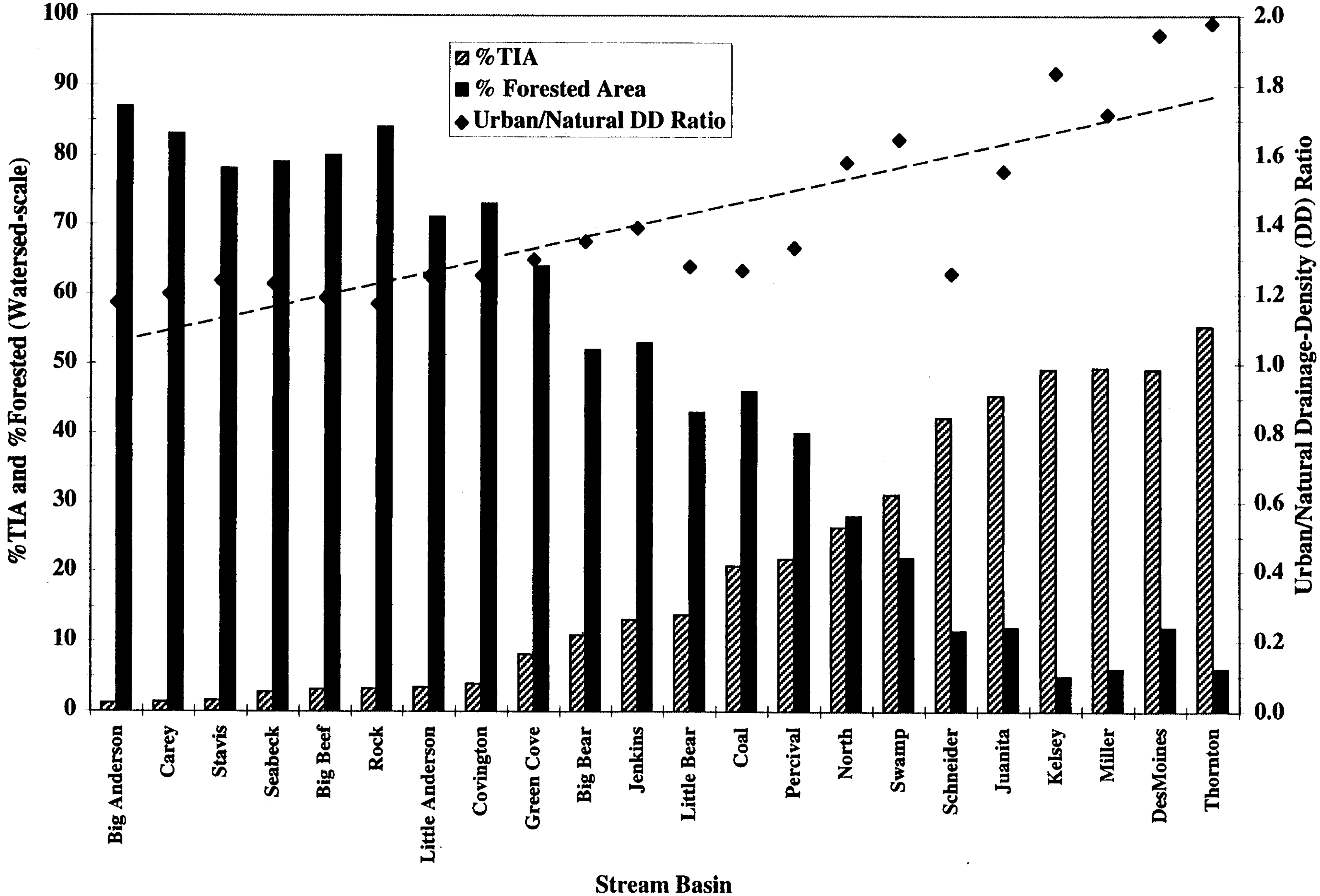

Figure 11a: Relationship between watershed land-cover and drainage-density (DD) in Puget Sound lowland streams. Dashed line shows correlation (r = 0.85) between DD and urbanization (%TIA).

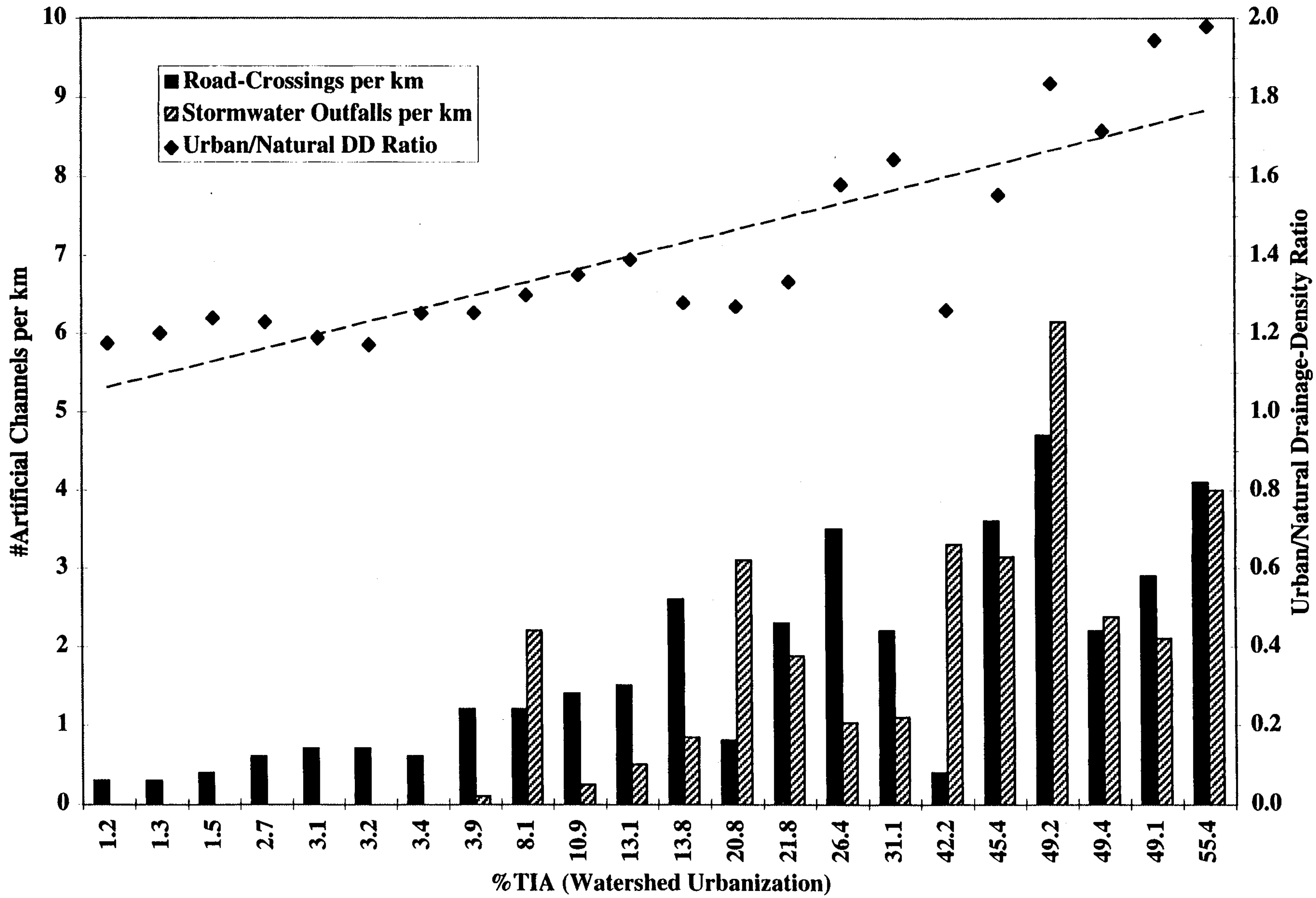

Figure 11b: Relationship between watershed drainage-density (DD) and the artificial stormwater drainage network in Puget Sound lowland streams. Dashed line shows correlation (r = 0.85) between DD and imperviousness (%TIA).

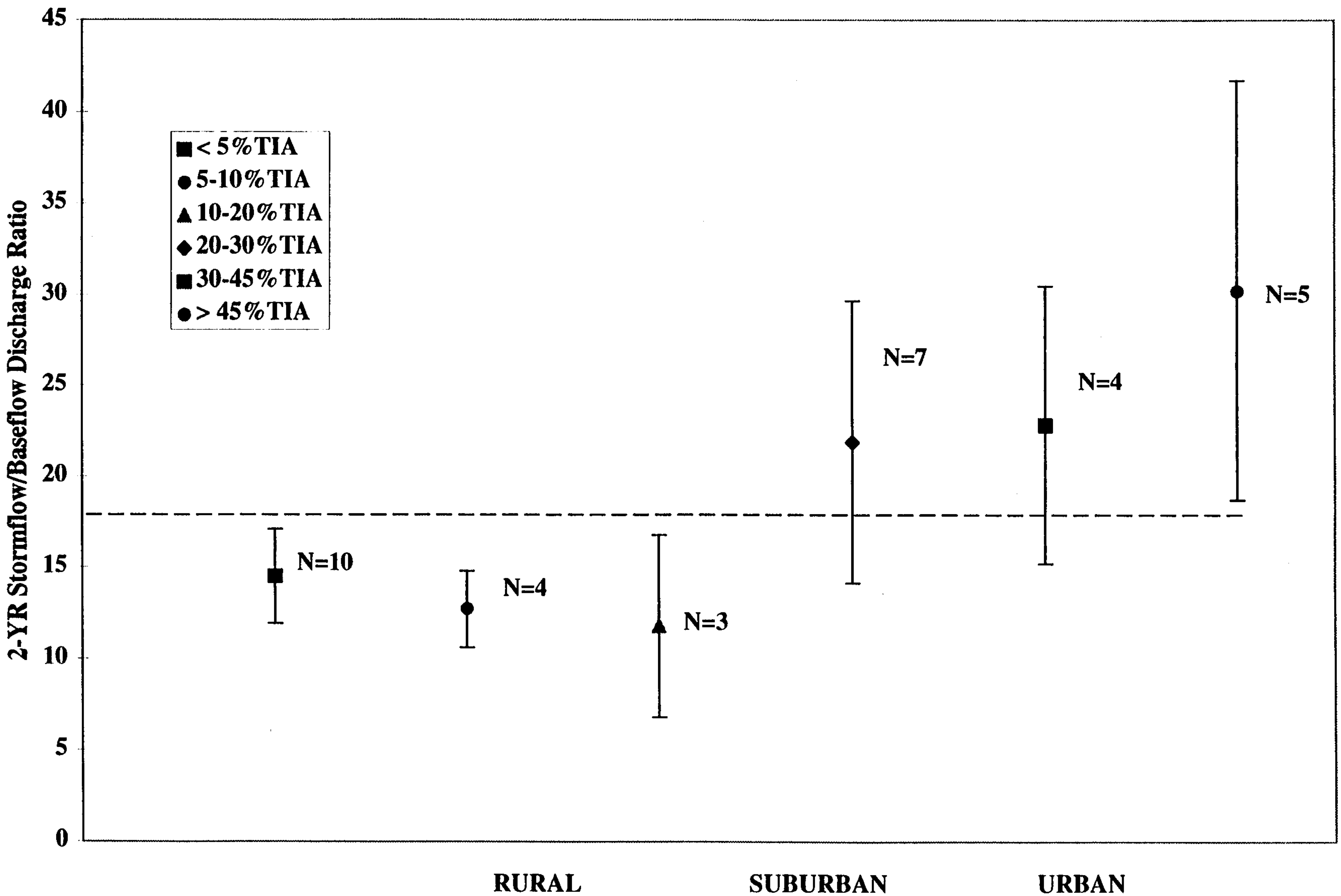

Figure 12: Relationship between watershed hydrologic regime (2-year stormflow to winter baseflow ratio) and basin urbanization (%TIA) in PSL streams.

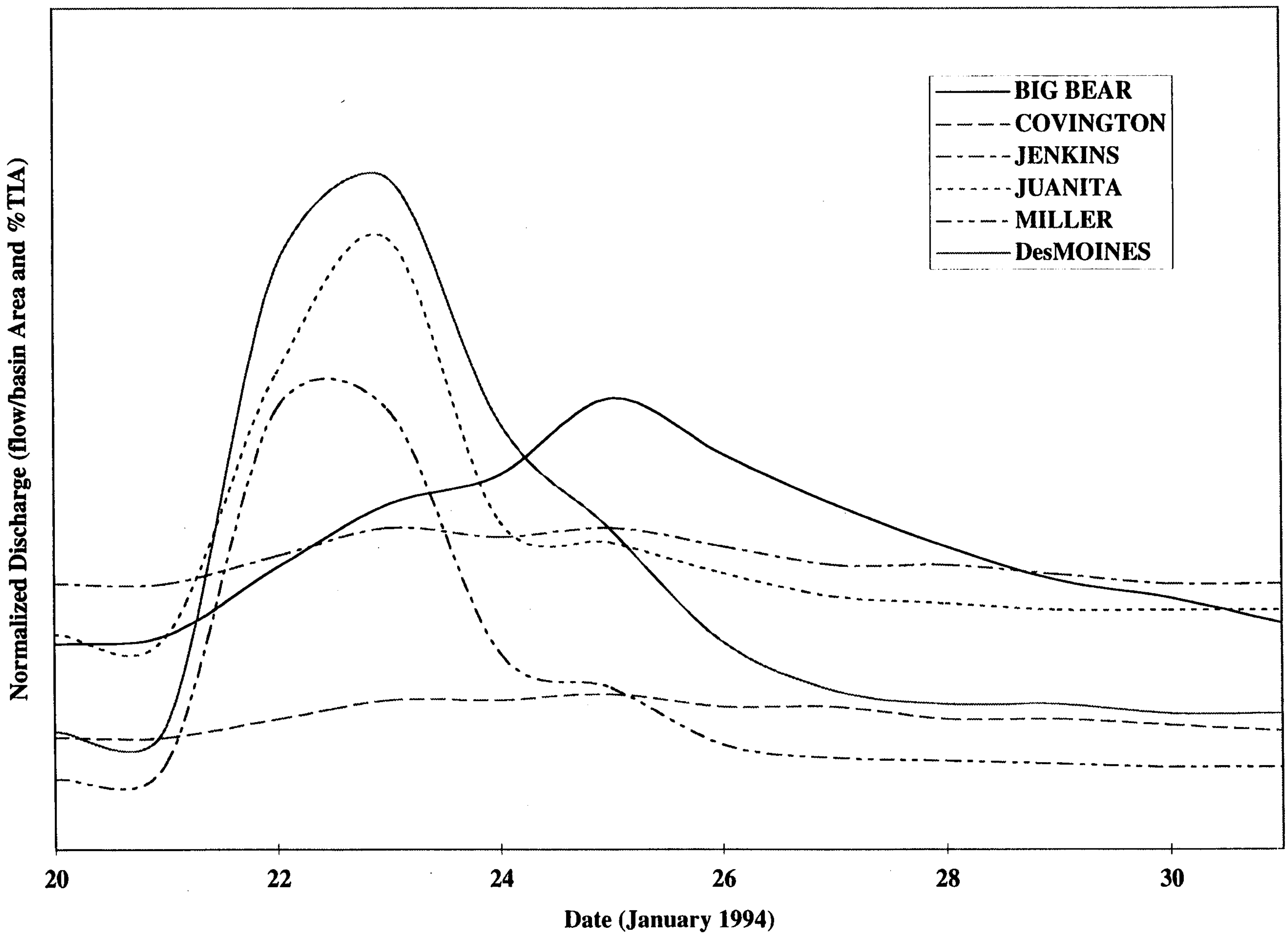

Figure 13: Typical winter rain-storm hydrographs for representative Puget Sound lowland streams (normalized for basin area and total impervious area).

shows the hydrograph for this storm for six streams with the curves normalized for basin area and %TIA. The three heavily urbanized (> 40 %TIA) basins (Juanita, Miller, and DesMoines) all responded to this typical storm with rapidly rising and falling hydrographs characteristic of highly impervious catchments. Each of the urbanized streams returned to baseflow within 24 hours of the rainfall end. In contrast, the less urbanized basins (Covington, 4 %TIA; Big Bear, 11 %TIA; Jenkins, 13 %TIA) showed a much slower response to the storm event and took several days following the storm to return to baseflow conditions. Big Bear Creek, with a much larger basin had a higher hydrograph peak than the other two rural streams, yet it too responded much more slowly to the storm than the three urbanized streams. Jenkins and Covington Creeks also have headwater lakes and intact headwater wetlands as does Big Bear Creek. Covington and Jenkins are also dominated by outwash soils as opposed to the till-dominated soils of the other four streams. Outwash soils tend to be more permeable than glacial tills and thus provide better infiltration capacity. All three of the urbanized streams have lost much of their headwater wetland area to development (especially Des Moines Creek which has the runways for SEATAC Airport located in its headwaters). Wetlands tend to moderate streamflows and are recognized for their natural flood-control capabilities. The riparian corridors of these urbanized streams are also significantly altered. All three of the urbanized streams have a 2-year stormflow to baseflow ratio > 20 whereas the rural basins were well under 20.

Urbanization had no detectable effect on summer low-flow, possibly due to the limited continuous flow data base (only 5 streams). Urbanization usually decreases summer baseflow, however, that is not always the case. In addition to %TIA, other basin characteristics (soils, topography, etc.) also effect summer low-flow levels and could be confounding the results.

Flood Frequency

The frequency of high-flow or flooding events has often been used as an indicator of urbanization. Continuous discharge data (in the form of mean daily discharge) were available for only five of the study streams, and for a sufficient period of record (1988-1994), to allow for a modified flood-frequency analysis. This time period was determined to be long enough so that it included representative seasonal rainfall patterns and short enough such that basin development did not change significantly. Included in this data-set were Covington (4 %TIA), Big Bear (11 %TIA), Jenkins (13 %TIA), Miller (49 %TIA), and Des Moines (50 %TIA) Creeks. Table 8 shows the frequency of mean daily discharges (MDD) greater than the average MDD over that six-year period. Representative flood flows were tabulated for 3X, 5X, and 10X the normal MDD. Although the data-set is limited, this analysis indicates that overbank, flood flows occur with greater frequency in urbanized PSL streams than in streams draining undeveloped basins. This agrees with the results of previous studies in the PSL (Richey, 1982; Booth, 1991; Law, 1994). Small (3XMDD) high-flow events are, on average, about twice as frequent in urban basins than in rural catchments. Moderate (5XMDD) flow events appear to be 3-times more frequent in the highly urbanized watersheds, as do large (10XMDD) flooding events.

Annual maximum, mean, and minimum MDD were determined from the continuous discharge records for the five streams. The ratio of maximum to mean and maximum to minimum annual flows were calculated to normalize flows for differences in both watershed size and precipitation pattern (Table 9). Urban streams (> 40 %TIA) had significantly larger Max:Mean and Max:Min flow ratios than rural streams (< 15 %TIA). Urban stream ratios were on average about twice that of rural catchments. This confirms the earlier reported results comparing modeled 2-Year stormflows and baseflow discharges. Urbanization clearly resulted in greater streamflow fluctuation, which represents a natural disturbance to the stream channel.

TABLE 8: Frequency of flooding events during the period 1988-1994.

Stream Basin (%TIA)	3XMDD	5XMDD	10XMDD
Covington (4%)	20	6	0
Big Bear (11%)	13	1	1
Jenkins (13%)	18	5	0
Miller (49%)	45	22	6
Des Moines (50%)	36	19	3

MDD = mean daily discharge
%TIA = total impervious area

TABLE 9: Mean Daily Discharge (MDD) ratios based on USGS data for two periods of stream gauging (1945-1979 and 1988-1994).

Stream Basin	Period of Record	Max:Mean Ratio	Max:Min Ratio
Covington	1988-1994	2.17	3.93
Big Bear	1988-1994	2.22	4.05
Jenkins	1988-1994	1.66	2.51
Miller*	1988-1994	4.89	13.78
Des Moines*	1988-1994	4.09	10.89
Kelsey	1945-1979	1.34	1.96
May	1964-1979	1.19	1.55
Juanita	1945-1979	1.48	2.15
North	1945-1973	1.71	2.65
Swamp	1964-1979	1.32	2.25
Rock	1945-1973	1.53	3.13
Big Beef	1969-1979	1.52	3.97

* = Streams that were already urbanized prior to the period of record
MDD = mean daily discharge
%TIA = total impervious area

Continuous discharge records were available (USGS) on four streams for a longer period (1945-1979). Max:Mean and Max:Min ratios for this data-set indicate that prior to the beginning of the major development surge in the early 1980s, there was a minimal difference in the ratios among those streams (Table 9), many of which are currently urbanized. Unfortunately, the same streams are not represented in both data-sets. Reliable %TIA data were not available for the older period of record.

Relative Stream Power

Hydrologic Observations and Model Predictions

No relationship between measured summer base flow normalized for basin area and %TIA was evident for the study streams, although others have observed an inverse relation between summer base flow per unit basin area and urbanization (Klein, 1979; Leopold, 1968; Richey, 1982). Winter base flow per unit basin area generally decreased with increasing %TIA. However five stream segments that receive relatively high rainfall or have an artificially augmented water source exhibited higher values of winter base flow/unit basin area than others at an approximately equivalent urbanization level.

The 2-year peak flow rates estimated by modeling were also normalized for basin area and compared to %TIA. Normalized 2-year peak flow generally increased with increasing %TIA, although the trend was not pronounced. The three basins with predominantly outwash soil had the lowest normalized 2-year peak flows, indicating the importance of a factor other than urbanization. Other factors, such as differences in rainfall intensities, storm antecedent dry periods, and lag times between basins, also may have masked the increased peak flows that occurred in each basin with urbanization when the range of study streams was considered.

The ratio of 2-year peak flow to winter base flow (flow ratio), representing relative stream power, increased with increasing %TIA (Figure 14). Use of a flow ratio compensates for variations among watersheds, such as meteorological characteristics, soil type and area. This ratio reflects the combined effects of winter base flow decreasing and 2-year peak flow increasing with increased urbanization in a watershed. Similar to the relationship with %TIA, the flow ratio also increased with increasing road density (r = 0.66) and increasing urban/natural drainage density ratio (r = .67).

Relative Stream Power Related to Habitat

With increasing flow ratio there was generally less large woody debris (LWD) in the study streams (Figure 15). However, low quantities of LWD occurred at all values of the flow ratio. At flow ratios less than 15, there were stream reaches with over 250 pieces of LWD/km. Between flow ratios of 15 and 35, there were no reaches with more than 250 pieces of LWD/km. At flow ratios higher than 35, no reach had more than 100 pieces of LWD/km.

Lack of wood in streams with high flow ratios may have been observed for several reasons. Increased peak flows can wash wood out of streams. Wood in the stream can become stranded above the bankfull level due to higher flows, rendering it useless as a habitat forming element. Alternatively, changes in the hydrologic regime may not have been the cause of decreased LWD. People often remove wood from streams. The riparian zone may have been removed, and no new wood would have entered the stream. The amount of wood in any stream, even those considered pristine, in the Puget Sound lowland area has probably been altered by humans. Logging operations in the region have removed wood from many streams and sources of new wood from the adjacent land.

The number of pools/km of stream generally decreased with increasing flow ratio (Figure 16). At flow ratios less than 15, there were stream reaches with more than 70 pools/km. With flow ratios between 15 and 35, only one of 16 stream reaches had more than 70 pools/km. At flow ratios greater than 35, all reaches had less than 35 pools/km. The altered flow regime in urban streams appeared to affect the occurrence of pools because of the decreased the persistence of LWD in a channel, an important determinant of pool formation.

The percentage of pool cover is a good overall indicator of the habitat quality of a stream reach. This characteristic includes any form of cover over, on or in a pool, for

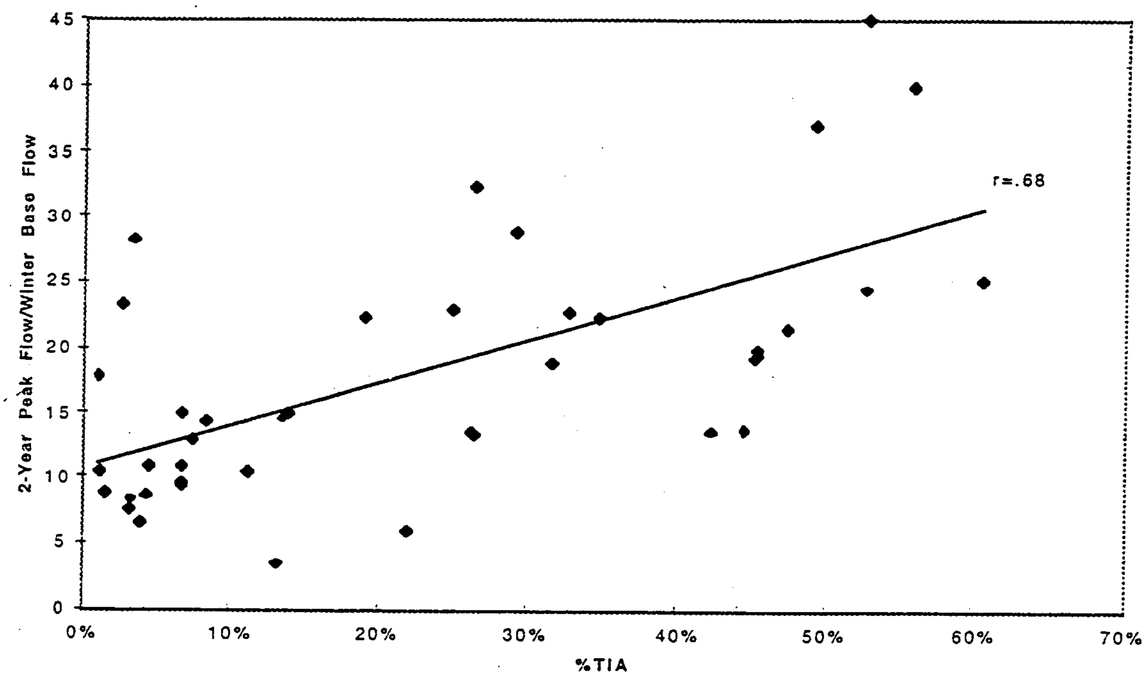

Figure 14. Relationship between %TIA and 2-Year Peak Flow/Winter Base Flow for Puget Sound Lowland Study Streams

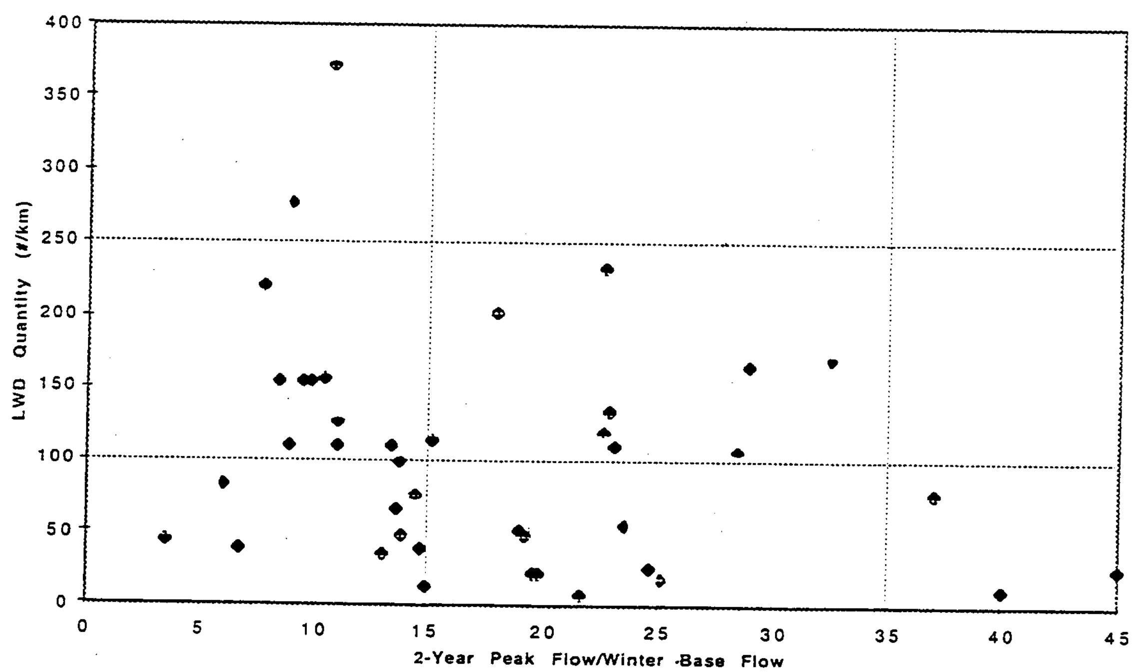

Figure 15. Relationship Between 2-Year Peak Flow/Winter Base Flow and LWD Quantity for Puget Sound Lowland Study Streams

example, LWD, rootwads, overhanging banks or vegetation, and large rocks. Average percent pool cover generally declined with increasing flow ratios (Figure 17). The stream reaches that had the most pool cover (>70%) occurred only with flow ratios less than 15, except for Swamp (SW) Creek site C. At flow ratios greater than 35, the average percent pool cover for any stream reach was always less than 35%. Therefore, low flow ratios did not guarantee a high percentage of pool cover. However, the results indicate that extensive pool cover is almost sure to be lacking with high flow ratios.

Increased peak flows may have contributed to loss of cover in several ways. LWD, brushy debris, and large rocks within the stream channel could have been washed out by high flows. Overhanging banks and terrestrial vegetation could have been lost due to erosion of the stream bank caused by an altered hydrologic regime and loss of riparian vegetation.

The trend towards decreased pool cover with increasing flow ratios may have been a direct result of urbanization, and not necessarily or entirely related to flow. For example, more urbanized streams had higher flow ratios, but generally also had degraded riparian areas. Less overhanging terrestrial vegetation, and therefore less pool cover, prevails in this situation, but not necessarily due to increased flows.

Substratum embeddedness, or the degree to which substratum particles are firmly enclosed in the stream bed, may also be affected by flow patterns in a stream. Substratum embeddedness generally increased with increasing flow ratios (Figure 18). At flow ratios below 13, substratum embeddedness never exceeded 20%. For flow ratios between 13 and 24, substratum embeddedness ranged from 5% to 50%. Above a flow ratio of 24, substratum embeddedness was always at least 20%, and reached as high as 60%.

Fine sediments are defined here as particles with a diameter less than 0.85 mm. The largest proportions of fine sediments (>20%) occurred in three of the most highly urbanized streams. All had flow ratios greater than 19, Juanita (JU), Kelsey (KE), and Thornton (TH) Creeks (Figure 19). The other two measurements greater than 20% occurred in streams considered to be outliers with regard to % fines. Fine sediments in Stavis (ST) Creek were influenced by tidal action. Big Bear (BB) Creek site A was

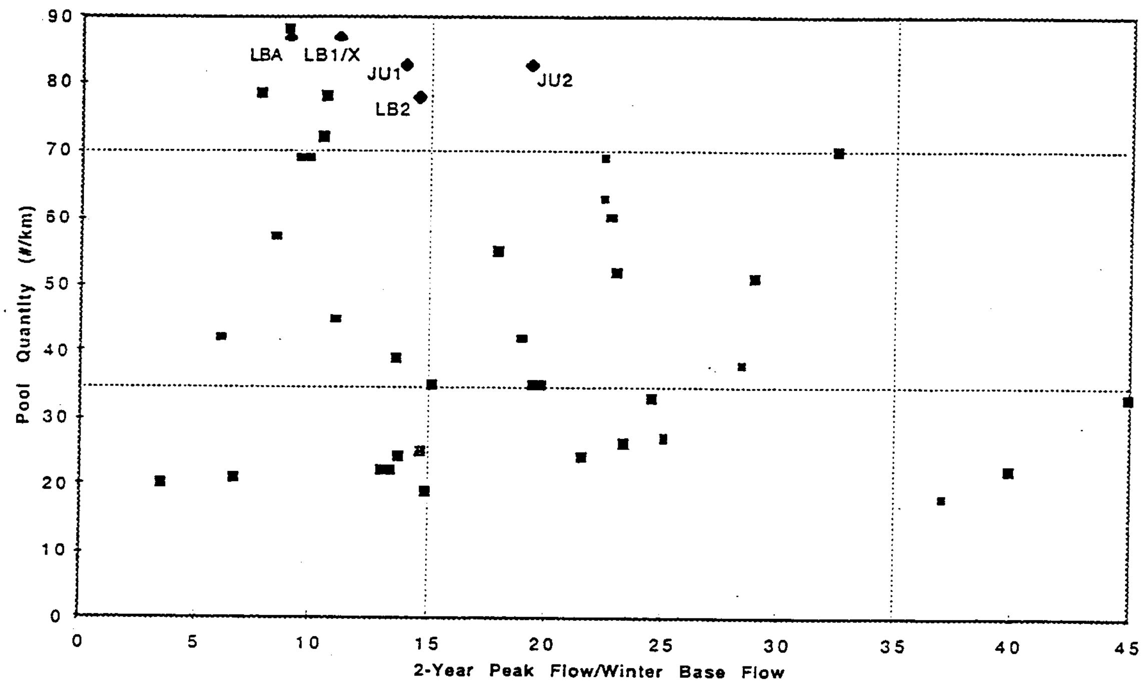

Figure 16. Relationship Between 2-Year Peak Flow/Winter Base Flow and Pool Quantity for Puget Sound Lowland Study Streams

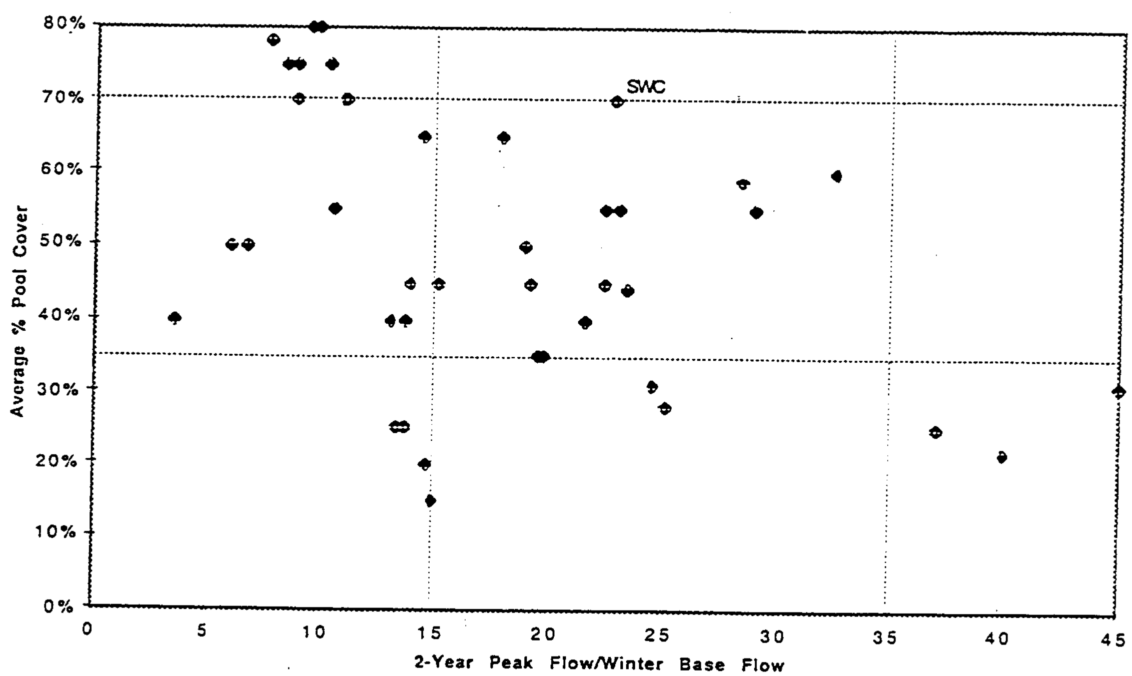

Figure 17. Relationship Between 2-Year Peak Flow/Winter Base Flow and % Pool Cover for Puget Sound Lowland Study Streams

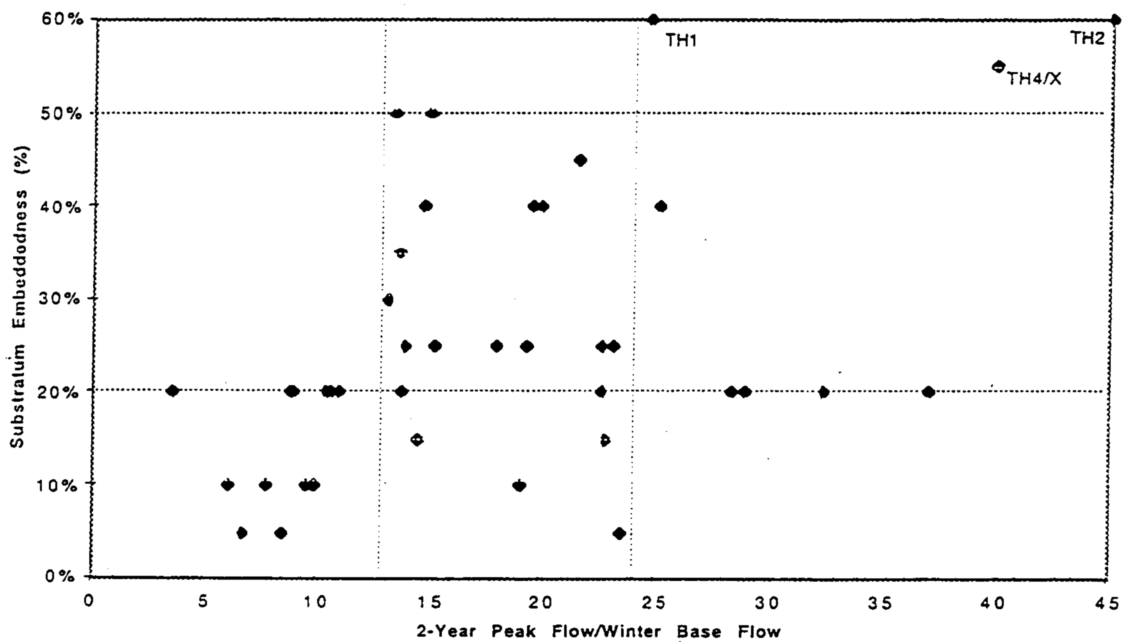

Figure 18. Relationship Between 2-Year Peak Flow/Winter Base Flow and Substratum Embeddedness for Puget Sound Lowland Study Streams

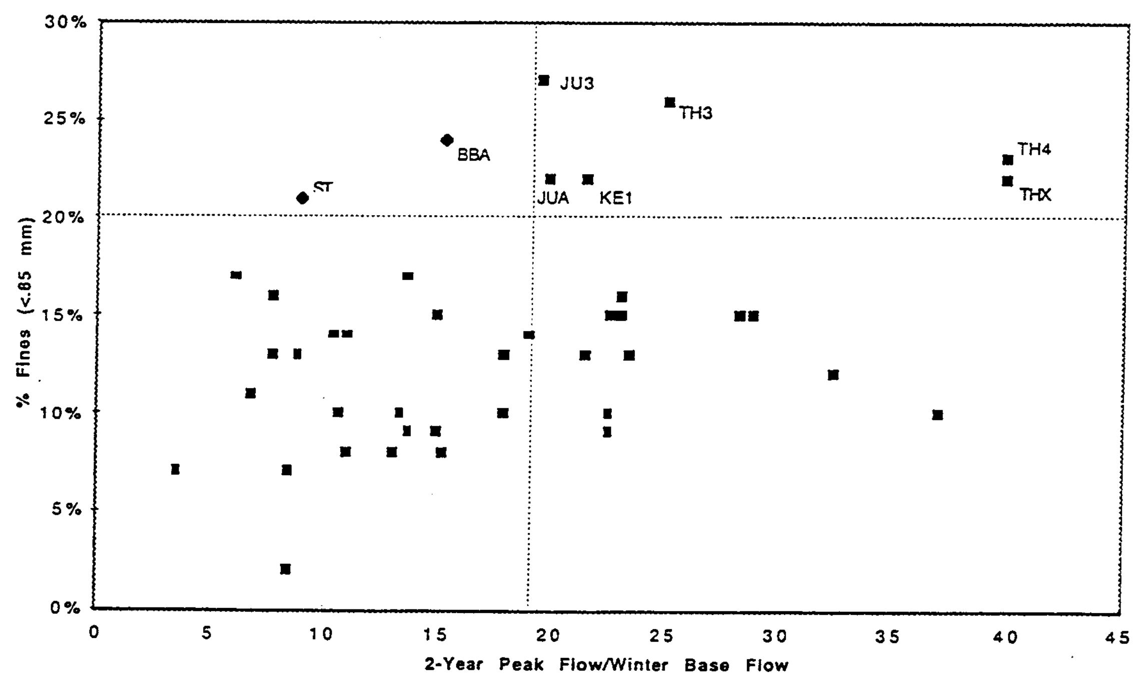

Figure 19. Relationship Between 2-Year Peak Flow/Winter Base Flow and % Fines for Puget Sound Lowland Study Streams

considered an outlier due to sampling site error. Increased storm flows and decreased base flows, as expressed by the flow ratio, did not appear to fully explain the high amounts of fine sediment in streams. Several streams with high flow ratios had low amounts of fine sediment, such as Des Moines Creek and Swamp Creek site A. Factors such as the level of urbanization, the average slope, the quality of the riparian zone, and the amount of debris (such as LWD) in a reach may better explain the presence or absence of fine sediments.

Relative Stream Power Related to Biological Indices

The increased duration and magnitude of peak flows occurring with urbanization are not only a concern because they alter in-stream habitat. Stream organisms may be physically displaced by high flows.

Figure 20 illustrates the relationship between relative stream power and the coho/cutthroat trout ratio. Above a flow ratio of approximately 20, cutthroat trout dominated in all streams for which data were available. This study has demonstrated that habitat features that affect fishes, such as pool quantity and pool cover, were probably affected by the altered hydrologic regime in urban watersheds, and that in turn these altered habitat conditions probably affected fishes. The question remains, however, as to the relative importance of this factor versus the direct effect of an altered flow regime that physically displaces fish from streams. No biologic data were collected as part of this study to answer this question. However, Scott et al. (1986) in their study of Kelsey Creek found no evidence that fish were being displaced by the increased magnitude or duration of peak storm flows. Instead, changes in habitat were blamed for the disappearance of an evenly distributed fish community and the shift to a community dominated by cutthroat. This evidence points to changes in habitat, and not physical displacement by high flows, as the most severe impact an altered hydrologic regime has on fish communities in urban streams.

B-IBI scores generally declined with increasing values of the flow ratio, with the exception of Little Anderson (LA) Creek (Figure 21). Little Anderson Creek has several roads in its headwaters that may partially explain the high flows that occurred in this stream. However, the reason for the high flow ratio and high B-IBI score associated with Little Anderson Creek was not entirely clear.

All B-IBI scores greater than 35 occurred at flow ratios less than 20 (except for Little Anderson Creek). Between flow ratios of 20 and 35, all B-IBI scores were less than 35 (except for Little Anderson Creek). At flow ratios greater than 35, all B-IBI scores were less than 20.

Similar to fish abundance, the question remains as to whether the relationship between the B-IBI scores and flow ratios was observed because of the effects of flow on habitat features, or if, instead, macroinvertebrates were being washed from streams. This study has demonstrated that increased flows due to urbanization may alter the amount of wood in a stream and substratum embeddedness. These habitat changes, in turn, probably affect the macroinvertebrate community.

No biologic data were collected as part of this study to determine if macroinvertebrates were physically displaced from streams by high flows. However, other studies have examined the direct effects of high flow on macroinvertebrates. Borchardt and Statzner (1990) found that during high stream flows large macroinvertebrate population losses occurred if no pore spaces were available in the substratum as a refuge. Perhaps, high flows combined with altered habitat were largely responsible for lowered B-IBI scores in Puget Sound lowland urban creeks. Macroinvertebrates could have been washed out of streams because they had no refuge during high flows. Statzner et al. (1988) found that most macroinvertebrates can withstand much higher velocities than those normally occurring in streams. It is the energy costs of withstanding high flow forces for long periods that can cause organisms to drift. While the evidence is that fishes are relatively unaffected directly by high flows, it appears that the direct effects on macroinvertebrates can be more significant.

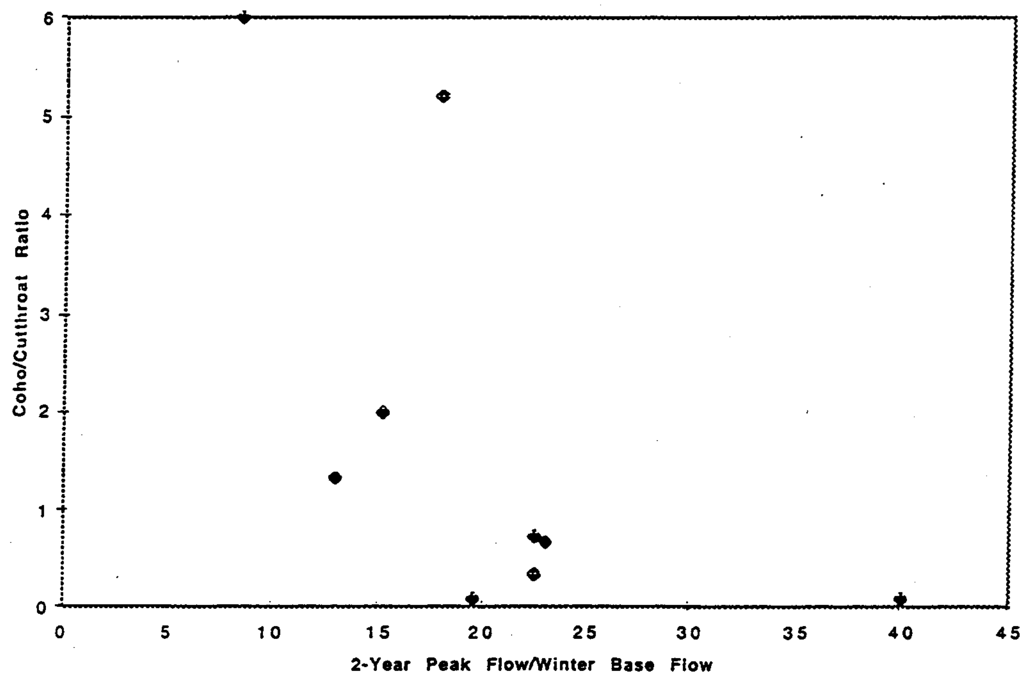

Figure 20. Relationship Between 2-Year Peak Flow/Winter Base Flow and Coho/Cutthroat Ratio for Puget Sound Lowland Study Streams

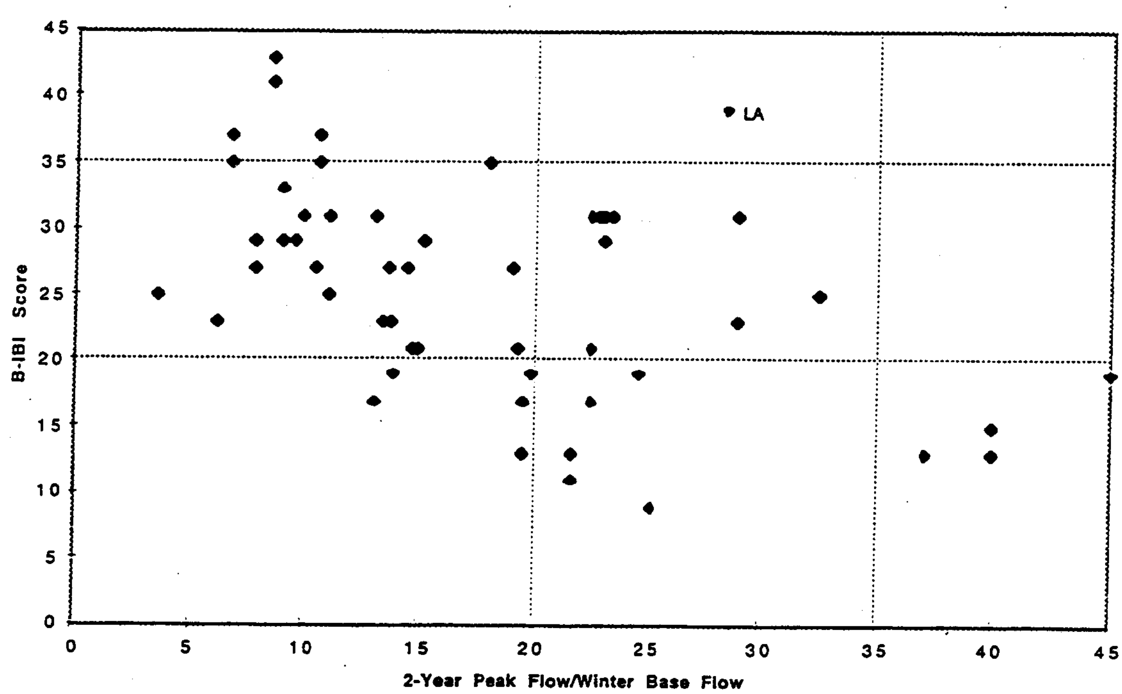

Figure 21. Relationship Between 2-Year Peak Flow/Winter Base Flow and B-IBI Score for Puget Sound Lowland Study Streams

Summary of Effects of Relative Stream Power

Figure 22 indicates the flow ratio magnitudes where changes in habitat and biotic integrity became apparent in the study streams. Habitat changes began to occur with flow ratios as low as 13. Flow ratios of 20 or less appeared to maintain levels of biotic integrity close to those found in unimpacted streams in the region. However, a flow ratio less than 20 did not guarantee a stream with high biotic integrity. With flow ratios between 20 and 35, habitat conditions and biologic communities were degraded from the most pristine levels, but not to the degree seen at flow ratios greater than 35. When flow ratios exceeded 35, in-stream habitat and biotic integrity were highly degraded.

Artificial Drainage Network

Another important aspect in urbanized basins is the artificial drainage network and artificial stormwater storage volume. Data on location and technical characteristics of stormwater facilities was difficult to obtain; its quality and availability varied among municipalities. Most retention/detention (R/D) facilities are designed primarily for quantity-control. The number of stormwater outfalls/km of stream was tabulated in the field. As expected, the number of drainage system links to the stream channel network strongly related to development level (Figure 13b). Stormdrains are another path for untreated runoff to enter the stream channel, in addition to road-crossings and drainage ditches. The number and total volume of retention/detention (R/D) facilities was also tabulated for each stream (Table 7).

The Percival Creek watershed had by-far the largest storage volume of the PSL streams. This reflects the City of Olympia's strong commitment to large regional detention facilities rather than multiple, smaller R/D ponds. Percival Creek watershed contains the Yauger Park multi-use R/D facility and the newly constructed Mottman Road stormwater treatment site on Black Lake Ditch. In contrast, the Kelsey Creek watershed in the City of Bellevue had numerous smaller R/D facilities, reflecting both the built-out condition of the watershed (lack of space for large facilities) and the city policy of stormwater controls for individual developments. Older developed basins tend to have

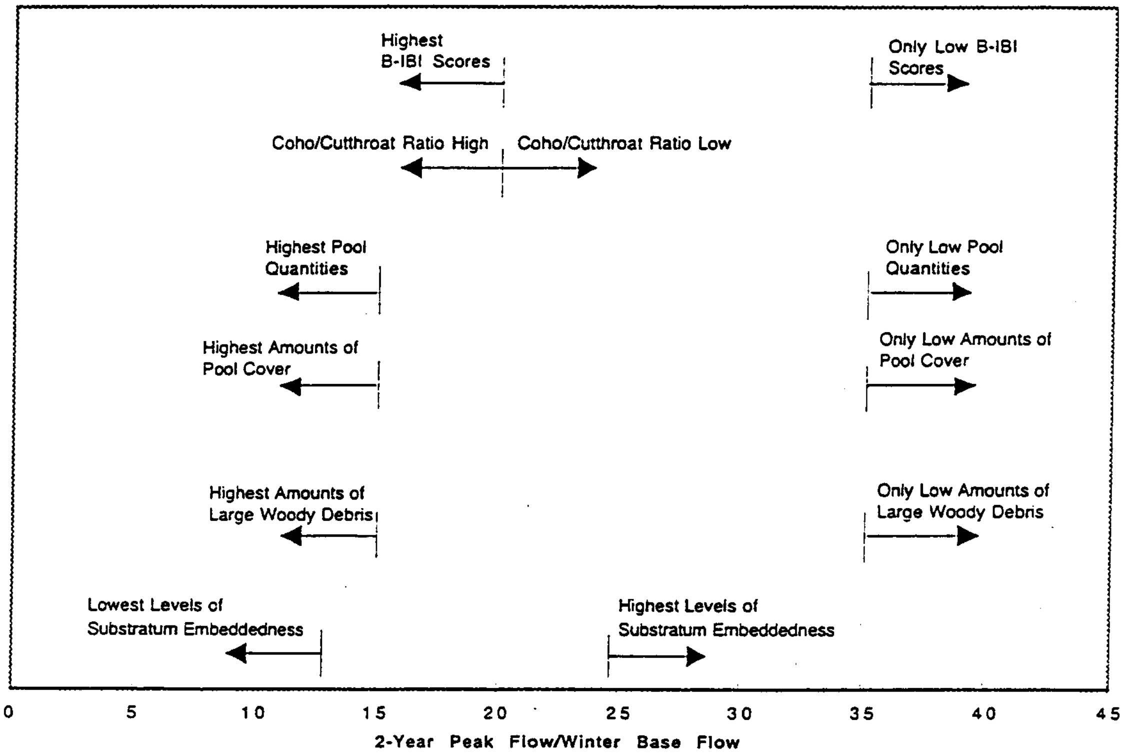

Figure 22. Habitat and Biological Effects of an Altered Stream Hydrologic Regime for Puget Sound Lowland Study Streams

numerous small stormwater facilities, whereas more recently developed basins and certain municipalities were dominated by large, regional stormwater facilities.

Stream Classification

An average of 32% of total stream lengths was physically surveyed. This met the goal of at least 25%, which previous studies had indicated as the minimum needed for a "representative" sampling of channel characteristics and stream habitat (see Habitat Assessment Protocols - Appendix B, May, 1996). In addition, all stream-segments less than 0.5 km in length were fully surveyed.

The distribution of characteristics is complex and "patchy". While 100% coverage of many segments would be required to obtain reliable data, that was impossible to achieve for most streams due to logistic and cost constraints. Nevertheless, delineating stream segments based on land-use, as well as geomorphic factors helped to sort out cause and effect relationships between human impacts and physical conditions. Dividing each stream into segments also allowed for smaller, more manageable, and "representative" units to survey.

Each stream segment was classified (or "typed") using the Rosgen method (Rosgen, 1994). All segments had a gradient less than 3.5%, with the majority of lowland streams less than 2%. All segments could be characterized broadly as the "pool-riffle" type. The Rosgen "B", "C", and "E" types are the most common in the PSL, along with several "F" and "G" types that have evolved due to incision of the streambed by high flows. Braided "DA" type channels were also found in streams with intact estuary outlets (Table 7). A similar range of geomorphic characteristics existed over the range of channel types. Dominant streambed material consisted of gravel mixed with sand and cobble. Most stream types had comparable natural shear stress components, due to similar slopes, streambed roughness, and flows. Also, most streams had highly erodable streambanks due to glacially-derived soils. Stream structure was dependent on woody material (LWD) and was, therefore, sensitive to disturbances in the riparian zone (Rosgen, 1994). The location and frequency of pool-riffle sequences is dependent on

LWD. Stream types observed encompassed the range of channel evolutionary changes common to low gradient, pool-riffle types (Rosgen, 1994). Over time, stream channels may widen and/or down-cut, and most migrate within their floodplain.

A wide variety of soil types (based on drainage characteristics) were observed. Till soils, deposited by glaciation (Vashon) dominate most basins with outwash soils being common in stream channels. Hydric soils were also common in wetland areas. The basins of Covington, Percival, and Jenkins Creeks consisted of predominantly outwash soil types and may, therefore, have some unique hydrologic characteristics. In general, no significant, underlying relationship was found between soil-type and stream quality. Nevertheless, till soils were significant in the determination of overall watershed runoff characteristics and hydrologic modeling. Generally, outwash (higher permeability) soils tended to tolerate runoff more than till soils (Cooper, 1996).

Channel Morphological Characteristics

Bankfull Dimensions

Stream size is generally proportional to basin drainage area (Leopold et al, 1964). Higher peak flows in urbanized streams tends to produce channels with larger dimensions than those in undeveloped, natural watersheds (Booth, 1990). Bankfull width (BFW) and depth (BFD) were measured at specified intervals and were only weakly related to basin area. Even the width to depth (BFW/BFD) ratio showed little relationship to basin area, while flood-prone width (FPW) was only slightly related with basin area.

The main reason for the poor relation between basin size and stream size is probably due to the dominating effect of urbanization on stream channel and floodplain dimensions. Cooper (1996) found that channel cross-sectional areas of these streams was slightly related to the 2-year stormflow to winter baseflow ratio and the relationship held even for highly-urbanized, constrained streams. These stream channels tended to deepen (or incise) to accommodate increased flows, because they were unable to widen their due to installed bank reinforcement. Exceptions to this pattern were several reaches along

Swamp Creek. These segments had fairly wide, continuous riparian corridors and upstream riparian wetlands to buffer the effects of runoff and high flows.

The hydrologic effects of urbanization usually widen and/or deepen stream channels, while the encroachment of development often has an opposite effect on channel width. All but one of the most urbanized streams (Schneider; > 40 %TIA) had streambank reinforcement (rip-rap) present over a majority of the surveyed reaches. In addition, streams in the moderate development range (20-40 %TIA) had rip-rap as a common streambank feature in reaches flowing through suburban and urban sub-basins. The floodplain area for all streams over 20 %TIA was also severely constrained or non-existent.

BFW was normalized for basin area and compared to upstream cumulative imperviousness (%TIA). Under natural conditions, natural variability in BFW should be small (Leopold, 1994). This is generally the case even for urbanized PSL streams (Figure 23). However, there are several interesting exceptions at either end of the development spectrum. Two segments of Stavis and Big Anderson Creeks had significantly larger BFWs than most of the other undeveloped segments. However, the sub-basins of these segments had been logged within the past 10 years, which may explain the channel widening. Schneider, Kelsey (upper), and Miller (Walker tributary) Creeks, which were at the urbanized end of the development spectrum, all had wider BFWs than would be expected for their basin size. These segments are all located in highly developed sub-basins and have relatively unconstrained channels and/or intact floodplains, thus allowing the stream to enlarge in proportion to the dominant discharge. The results for BFD were similar except that fewer incised channel segments were noted. This may be a result of the selection of stream segments based primarily on potential fish utilization. Incision is generally more common in steep, constrained tributary channels (Booth, 1990) which are typically not prime fish habitat area. Schneider, Coal, Miller, Des Moines, and Percival Creeks had deeply incised channels.

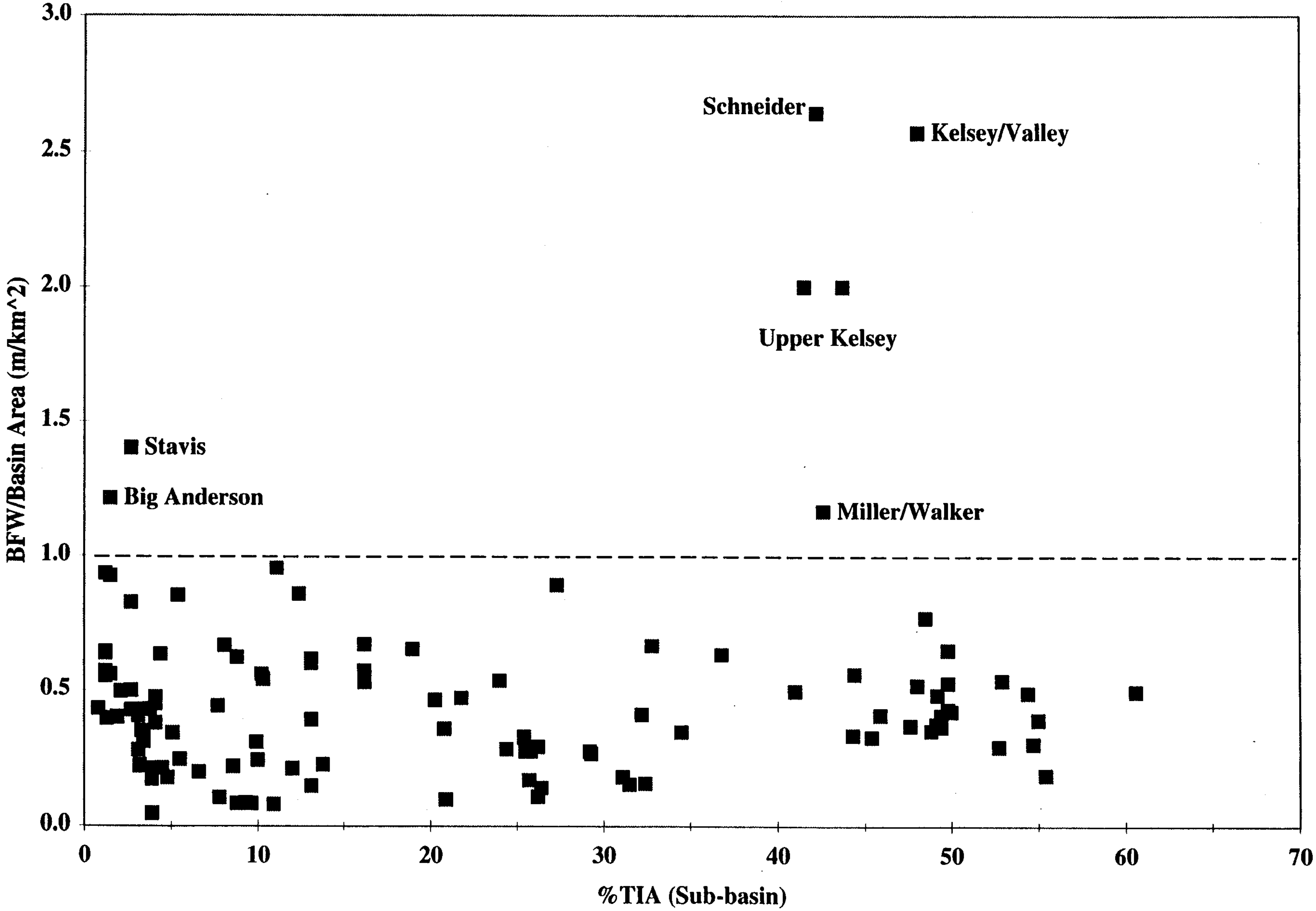

Figure 23: Relationship between channel bankfull width (BFW) and sub-basin urbanization (%TIA) in Puget Sound lowland streams.

Channel Gradient

Gradient can have a major influence on channel shape and other instream characteristics. Stream gradient is an important component of stream power and basal shear-stress and is recognized as a key indicator of channel incision susceptibility (Booth, 1990). However, with the exception of fine sediment deposition, gradient was unrelated to any measurement of channel size, physical habitat variables, or biological indicators. This is likely due to the lack of natural variation in stream gradient for PSL streams.

Channel Confinement

Confinement was classed as channelized, confined, moderate, or unconfined. Urban or suburban streams tended to be more confined and had smaller (or non-existent) floodplains than natural or rural basins. Only one third of all segments (n=120) were classified as channelized or confined, while 54% of suburban (%TIA > 20) and urban (%TIA > 40) segments (n=68) were classified as channelized or confined. Of the urban segments only (n=26), 81% were classed as channelized or confined. Nearly half (46%) of the heavily urbanized (%TIA > 45%) segments were channelized. As indicated earlier, channel confinement probably affects stream channel dimensions significantly and may prevent urbanized stream channels from reaching a dimension "equilibrium". This may at least partially explain why BFW and BFD, while useful morphological parameters, may not be particularly indicative of urbanization. However, as discharge increases with urbanization and the channel is constrained from natural expansion, the increase in flow energy can cause streambed scour and streambank erosion.

Channel Sinuosity

There was no apparent relationship between development and sinuosity, even though streams in the less developed watersheds tended to be more meandering . Streams that were heavily affected by urbanization were more channelized and less sinuous. However, sinuosity also varied naturally with stream type and basin topography. As with stream channel dimensions, sinuosity was a useful classification variable but did not strongly indicate urbanization.

Streambank Stability

Streambank erosion has been used as an indicator of development impact (Booth and Reinelt, 1993), although it and hillslope mass-wasting are natural processes. Stream segments were classified according to the presence of streambank erosion as well as the quantity and quality of streambank cover (see methods section). A combined streambank stability index was assigned as follows; segments with >75% classified as stable were scored 4, between 50% and 75% stable banks were scored as 3, 25-50% as 2, and <25% as 1. Artificial streambank protection (rip-rap) was considered a sign of bank instability and given a 1.

Only two undeveloped, reference (%TIA < 5%) segments had a stability rating less than 3. One of these segments drained a new residential development in the Seabeck sub-basin and is the site of the only stormwater retention facility in the Hood Canal watershed. The other segment is located in a section of Covington Creek that has significant residential development along the creek with little or no riparian buffer. In the 5-10% TIA range, the streambank ratings were generally 3 or 4 with the exception of those in the lower mainstem of Big Bear Creek. That section is along Avondale Road (a major thoroughfare) into Redmond. The creek is channelized in several areas, is mostly devoid of riparian forest, and is constrained by development and roads. In addition, rip-rap was common in this section. Between 10-30% of sub-basin TIA contained a fairly even mixture of streambank conditions from stable and natural to highly eroded or artificially "protected". There were no segments with a stability rating of 4 and very few with a 3 above a sub-basin TIA of 30%. These extreme values were found only in segments with intact and wide riparian corridors (Miller, Des Moines, North and Swamp). Artificial streambank protection (rip-rap) was a common feature of all highly-urbanized (TIA > 45%) streams.

Streambank stability was inversely related with cumulative upstream basin %TIA and more closely correlated with development within the segment itself, perhaps reflecting the local effects of construction and other human activities. Streambank stability is also influenced by the state of riparian vegetation surrounding the stream. Streambank stability was strongly related to the width (>30m) of the riparian buffer and inversely related to the number of breaks in the riparian corridor. Although not the total cause, urbanization and loss of riparian vegetation contribute substantially to streambank instability. Other stream corridor characteristics, such as soil-type and valley hillslope gradient, also contribute to the stability potential and bank condition.

Streambed Conditions

Scour and Aggradation: Streambed stability was determined using bead-type scour meters installed in salmonid spawning riffles in selected reaches during the 1994-95 and 1995-96 storm seasons. These sites also corresponded to the macroinvertebrate and IGDO sample sites. No consistent relationship was found between the upstream level of urbanization (%TIA) and the magnitude of scour and fill in downstream reaches (Figure 24). As would be expected, larger scour and/or fill events usually resulted from larger storms and the resultant higher flows. However, not all streams experienced significant scour from the large storm events and undeveloped streams were as likely as urbanized streams to exhibit significant scour (Figure 25a). In some cases, undeveloped streams experienced more scour than urbanized streams for the same storm (Figure 25b). Scour data are included in May (1996; Appendix D).

Local streambed and channel conditions affected the vulnerability of channels to scour. For example, scour meters in lower Thornton Creek, which was the most urbanized stream with the most severe storm flows, was also extensively channalized and confined and, thus, showed little or no scour or fill regardless of storm intensity (May, 1996; Appendix D). The streambed in this section was extremely consolidated and "armored"- no longer susceptible to scour.

Magnitude of scour was similar from year to year. Reaches that experienced net aggradation over the period of the first storm season also aggraded during the next period. The same was true for reaches that usually scoured (May, 1996; Appendix D). Thus, site location had an important effect using this method to stream stability.

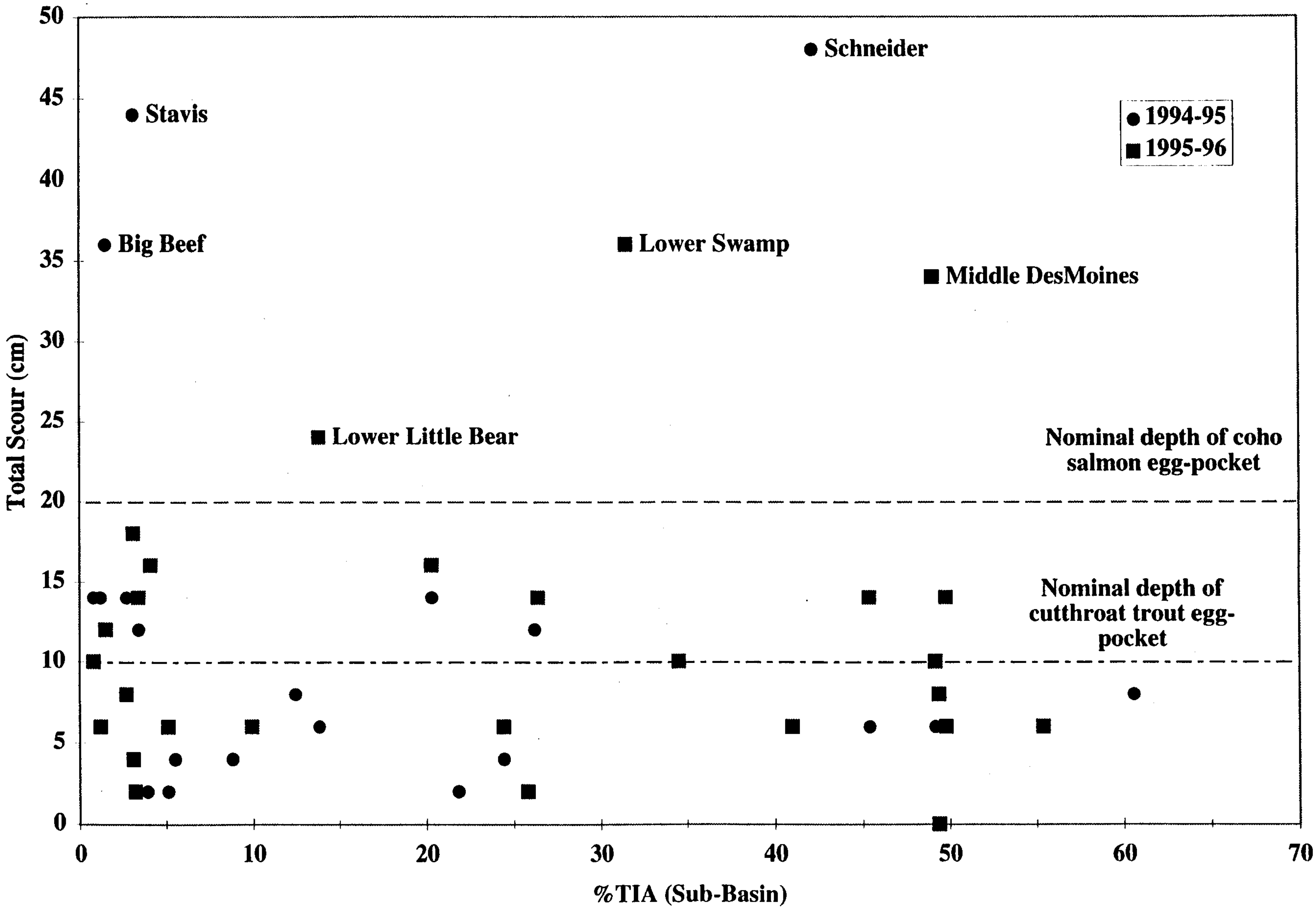

Figure 24: Mean total scour for monitored salmonid spawning sites during two storm seasons. Dashed-lines indicate typical depths of native salmonid redds.

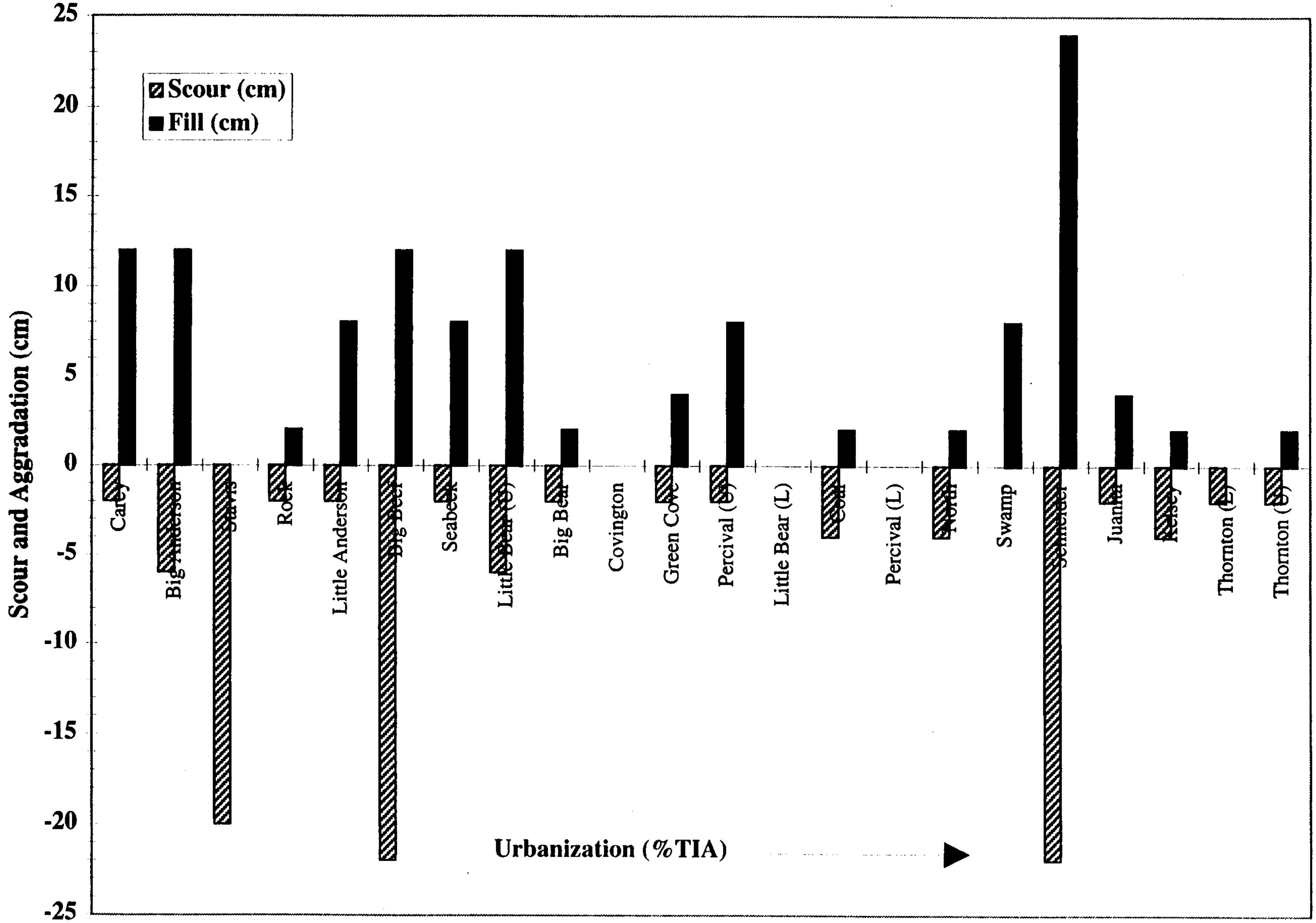

Figure 25a: Streambed scour and aggradation in Puget Sound lowland streams for a typical multi-day storm event (12/94).

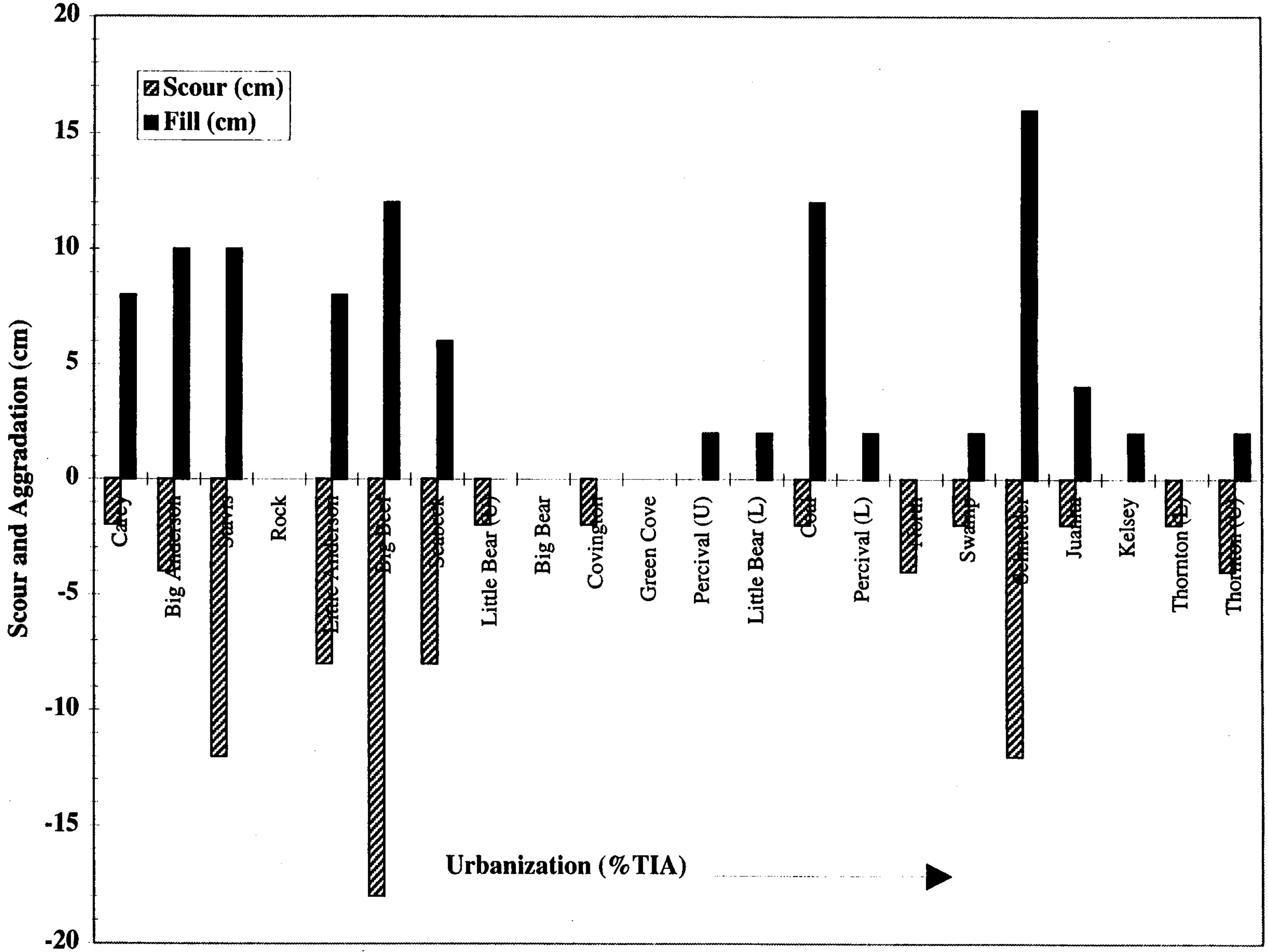

Figure 25b: Streambed scour and aggradation in Puget Sound lowland streams for a typical multi-day storm event (11/95).

A "scour-threshold" of 20 cm, which is the nominal depth of large salmonid redds, was used to indicate excessive scour (dashed line in Figure 24) for the entire monitoring period (winter storm season / salmonid egg incubation period). Several reaches with severe scour (> 20 cm) were observed, especially during large storm events in urban creeks with unconsolidated substrata (e.g., Schneider 94-95; Lower Swamp 95-96; and Des Moines 95-96). On the other hand, a few undeveloped creeks also had significant scour and/or fill episodes during major storms (Stavis and Big Beef 94-95). Again, location of meters may have been as important to streambed change as the magnitude of discharge. The overall potential for destruction of egg-pockets for cutthroat trout (10 cm nominal scour depth) appeared to be much higher (Figure 24). Neither of these scour thresholds were related to development level.

The available stream power and basal shear stress may be the most significant factors affecting streambed stability. Stream power (SP) is proportional to discharge and slope and would be expected to increase with urbanization, all else being equal, because flows tend to increase with urbanization. Cooper (1996) found this to be the case for these streams. Shear stress is dependent on slope, flow velocity, and bed-roughness. Critical basal shear stress determines the onset of streambed particle motion and the magnitude of scour and/or aggradation. Finding no significant relationship between the 2-year stormflow to winter-baseflow ratio and any of the scour monitor measurements (i.e. total scour, total fill, and net scour/fill) was not surprising, because local slope and streambed roughness and, hence, scour and fill are also highly variable. This result emphasizes the importance of site with scour and aggradation events.

Particle-size Distribution: Partical size was investigated using the modified Wolman (zig-zag) pebble-count, which is a time-tested, quick, simple, and inexpensive technique for characterizing streambed materials. The simple pebble-count method was used to assess its adequacy to determine effects of development. Urbanization was expected to shift the substrata composition toward finer particles. Pebble counts were made at the stream-segment leve. Stream reaches were selected (44 of 120 total segments were sampled) to cover a range of upstream urbanization.

As with streambed scour and aggradation, sediment particle size (using standard particle size parameters) was not strongly related to upstream %TIA (Olthof, 1994; Wydzga, 1997). Other factors than simply imperviousness affect partical, including location in the stream system, discharge history, and underlying geologic structure (outwash streams tend to have larger substrata). However, pebble-count D50 was inversely correlated with the 2-year stormflow/winter baseflow ratio Cooper (1996).

A shift in the lower end of a pebble-count cumulative frequency curve would indicate an increase in streambed fines (Potyondy and Hardy, 1994). The results here did show, to some extent, a shift toward finer particle size due to upstream development. shows the results of. There was no consistent pattern between zig-zag pebble-counts for selected stream segments upstream cumulative %TIA for either of the standard measures of average particle size (D50 - median particle size or GMPS - geometric mean particle size; Table 10). However, the D10 (10th percentile particle size) values indicated that more developed streams had a larger % of particles in the finest size class, indicating a possible shift toward finer particle size with urbanization. A D10 of 1 mm indicates that at least 10% of the sampled particles were smaller than 2 mm (the limit of accuracy). However, particle size is affected by other factors. Of the 22 suburban (20 < %TIA < 40) and urban (%TIA > 40) stream segments sampled, only 9 had a D10 greater than 1 mm, including lower Thornton Creek with a D10 = 2 mm. The remaining 8 segments, which were surrounded by intact riparian forest, had a D10 > 4 mm. That is some evidence for the mitigating effect of buffers.

Several stream segments, which had upstream %TIA < 5 and > 80% of their upstream riparian corridor >30m wide and in natural condition, were selected to examine the distribution of fines.. Each of these "reference" segments also had a streambank stability rating of 3 or 4 (good to excellent). Covington Creek was included in this analysis as a representative of streams dominated by outwash soil, compared to other PSL watersheds with predominantly till soils. Only 3 of the total 19 reaches showed a significant shift toward finer particle size, had a basin development %TIA < 10 (2 of the 3 were > 8% with the other located below a man-made lake with significant streambank erosion). Table 10 shows those reaches with an apparent shift toward fine sediment (frequency distributions are in Appendix E of May, 1996).

Pebble-count D10 values, in the same stream reaches, were directly related to % fines (Wydzga, 1996; Figure 26). Only pebble-count data collected from the same reach as the McNeil samples were considered. The results indicate that pebble-counts may be an effective and less-costly alternative to the more time-consuming McNeil sampling.

Pebble-count data for reaches within the same stream showed the effect of development (May, 1996; Appendix E). There was a shift toward more fine sediment in Little Bear Creek progressing downstream (most development is in the lower watershed). However, there were exceptions to the general trend toward finer particles as urbanization increased. Particle size distribution for Swamp and North Creeks showed the local effects of hydrologic, geomorphic, and land-use changes progressing from upstream down to the mouth. There were frequent changes in development intensity and land-use along these creeks. As with streambed stability, local factors had a significant influence on stream channel characteristics. Miller Creek, located within a highly developed basin, had a wide, intact riparian corridor in its lower portion, which would be expected to buffer the stream from the input of fine sediment. However, that was not supported by pebble-count data. The high fraction of fine sediment in the substratum at the lower end of the stream, may have originated from the upper basin and been transported downstream and/or may have resulted from local streambank erosion during excessive stormflows. In contrast, the Hood Canal (Kitsap) streams, dominated by forest and rural land-use, showed very similar particle size distributions and little fine sediment impact .

Chemical Water Quality

Base-flow

With the exception of conductivity (COND), alkalinity and sediment metal concentration (lead and zinc), strong relationships were not observed between base-flow water chemical constituents and development (%TIA). Using data collected during 1994-1996, together with that obtained from local agencies, COND for winter and summer

TABLE 10: Results of zig-zag pebble-count (mm) for selected reaches in Puget Sound lowland streams (1996).

STREAM SEGMENT	%TIA	D10 (mm)	D50 (mm)	Geometric Mean Particle Size
Thornton (South Branch)	60.6	1*	32	29
Thornton (Lower Mainstem)	55.4	2*	20	15
DesMoines (Middle/Above STP)	55.0	1*	45	28
Thornton (North Branch)	52.7	1*	14	13
DesMoines (Middle/Below STP)	51.0	4	26	19
DesMoines (Lower)	49.7	6	28	25
Miller (Middle)	49.4	1*	40	31
Miller (Lower)	49.3	1*	16	6
Kelsey (Upper)	48.0	6	40	31
Kelsey (Middle)	48.0	9	50	43
Kelsey (Lower)	47.3	1*	31	27
Juanita (Lower)	45.4	1*	24	15
North (McCollum/Upper)	41.0	1*	21	7
North (Mill Creek/Middle)	34.5	9	31	28
Swamp (Below Scriber Confluence)	32.4	10	63	47
Swamp (Wallace/Lower)	31.5	8	40	35
North (KC/Lower)	26.4	1*	25	11
North (Thrashers/Middle)	25.7	1*	31	27
Swamp (Butternut/Middle)	25.8	7	30	27
Swamp (Cypress/Middle)	24.4	1*	43	36
Coal (Lower)	20.8	1*	38	23
Coal (Middle)	20.3	1*	33	8
Coal (Upper)	19.0	1*	38	20
Little Bear (Lower)	13.8	1*	43	9
Jenkins (Lower)	13.1	14	75	57
Jenkins (Middle)	13.1	3	17	17
Jenkins (Upper)	13.1	7	30	27
Little Bear (Middle)	9.9	1*	34	29
Big Bear (Cottage/Upper)	8.6	1*	29	24
Big Bear (Cottage/Lower)	7.7	3	20	14
Big Bear (Middle Mainstem)	6.6	11	34	33
Big Bear (Upper Mainstem)	5.5	8	37	31
Little Bear (Upper)	4.1	6	26	21
Big Beef (Upper)	4.1	8	33	28
Covington (Lower)	3.9	14	70	52
Covington (Middle)	3.9	13	50	41
Little Anderson (Middle)	3.4	10	48	33
Rock (Lower)	3.2	20	66	56
Rock (Middle)	3.2	19	86	71
Big Beef (Lower)	3.1	1*	27	20
Seabeck (Lower)	2.7	4	43	34
Stavis (Middle)	1.5	6	42	32
Carey (Middle)	1.3	4	30	24
Big Anderson (Lower)	1.2	10	40	35

* Indicates that pebble-count cumulative frequency curve indicated a shift in D10 toward finer sediment when compared to reference reaches (significant at the 0.05 level). D10 is 10th percentile particle size (D50 is median particle size).

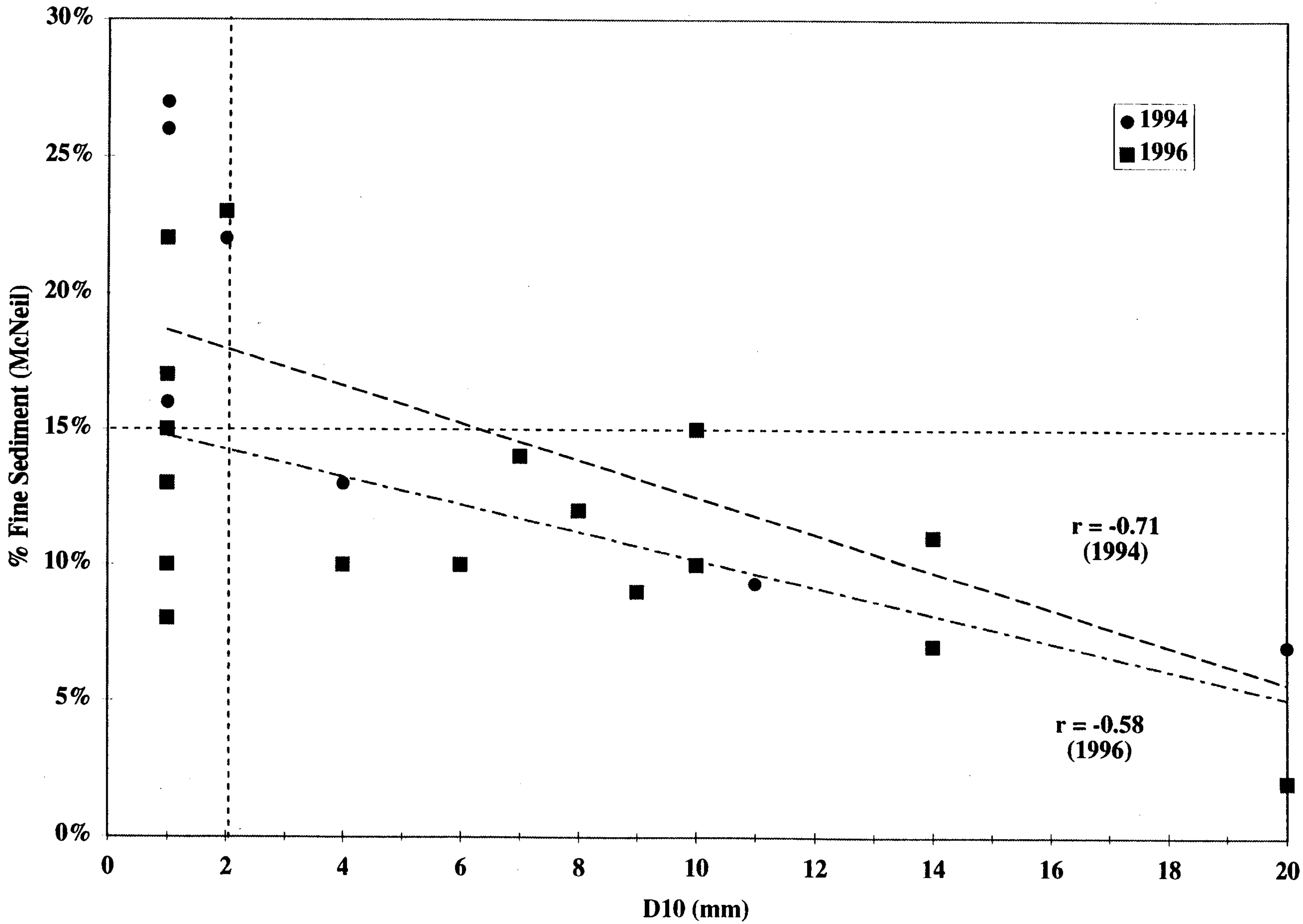

Figure 26: Relationship between substrata fine sediment (% fines) and pebble-count D10 particle size (mm). Fine sediment (McNeil) data from Olthof (1994) and Wydzga (1996). Dashed lines represent possible urban stream thresholds.

base flow combined was directly related to watershed urbanization (%TIA) ($r = 0.56$). However, Coal Creek had an exceptionally high COND and alkalinity, which was probably due to the residual effects of extensive coal-mining operations in its headwaters. The creek was used as a drainage system for mine shafts and its head waters still run underground through the abandoned mines. Concentrations of hardness ions (Ca^{2+}, Mg^{2+}), as well as Na^{+}, were up to three times those in other PSL streams. Hence, removal of Coal Creek data resulted in a much stronger relationship ($r = 0.91$; Figure 27). Winter and summer base-flow COND values are shown in Table 11.

TABLE 11: Baseflow conductivity (COND; µS/cm) for Puget Sound lowland streams (1994-1996).

Stream Basin	%TIA	Mean	COV(%)	n
Big Anderson	1.2	64	30.7	15
Carey	1.3	84	12.0	15
Stavis	1.5	86	49.1	15
Seabeck	2.7	71	31.0	15
Big Beef	3.1	67	28.6	15
Rock	3.2	89	16.9	13
Little Anderson	3.4	88	23.3	15
Covington	3.9	111	29.7	11
Green Cove	8.1	76	34.9	8
Big Bear (Upper)	9.9	89	14.2	12
Big Bear (Lower)	10.9	91	12.7	13
Little Bear (Upper)	5.1	105	15.9	15
Little Bear (Middle)	10.1	103	15.3	5
Little Bear (Lower)	13.8	124	12.6	15
Coal	**20.8**	**449**	**47.8**	**16**
Percival (Upper)	8.8	92	25.5	8
Percival (Lower)	21.8	91	20.8	8
North (Upper/McCollum)	41.0	159	39.1	6
North (Mill Creek)	29.2	142	14.0	6
North (Thrashers)	25.7	145	24.3	16
North (Lower/County	26.4	147	18.6	6
Swamp (Upper/Butternut)	25.8	135	16.3	6
Swamp (Cypress)	24.4	154	20.8	16
Swamp (Scriber)	32.4	171	8.2	6
Swamp (Lower/Wallace)	31.5	157	10.4	6
Schneider	42.2	131	17.8	8
Juanita	45.4	170	19.8	15
Kelsey	49.2	173	36.1	13
Thornton (Upper)	60.6	256	11.1	15
Thornton (Lower)	55.4	233	14.1	15

COV = coefficient of variation

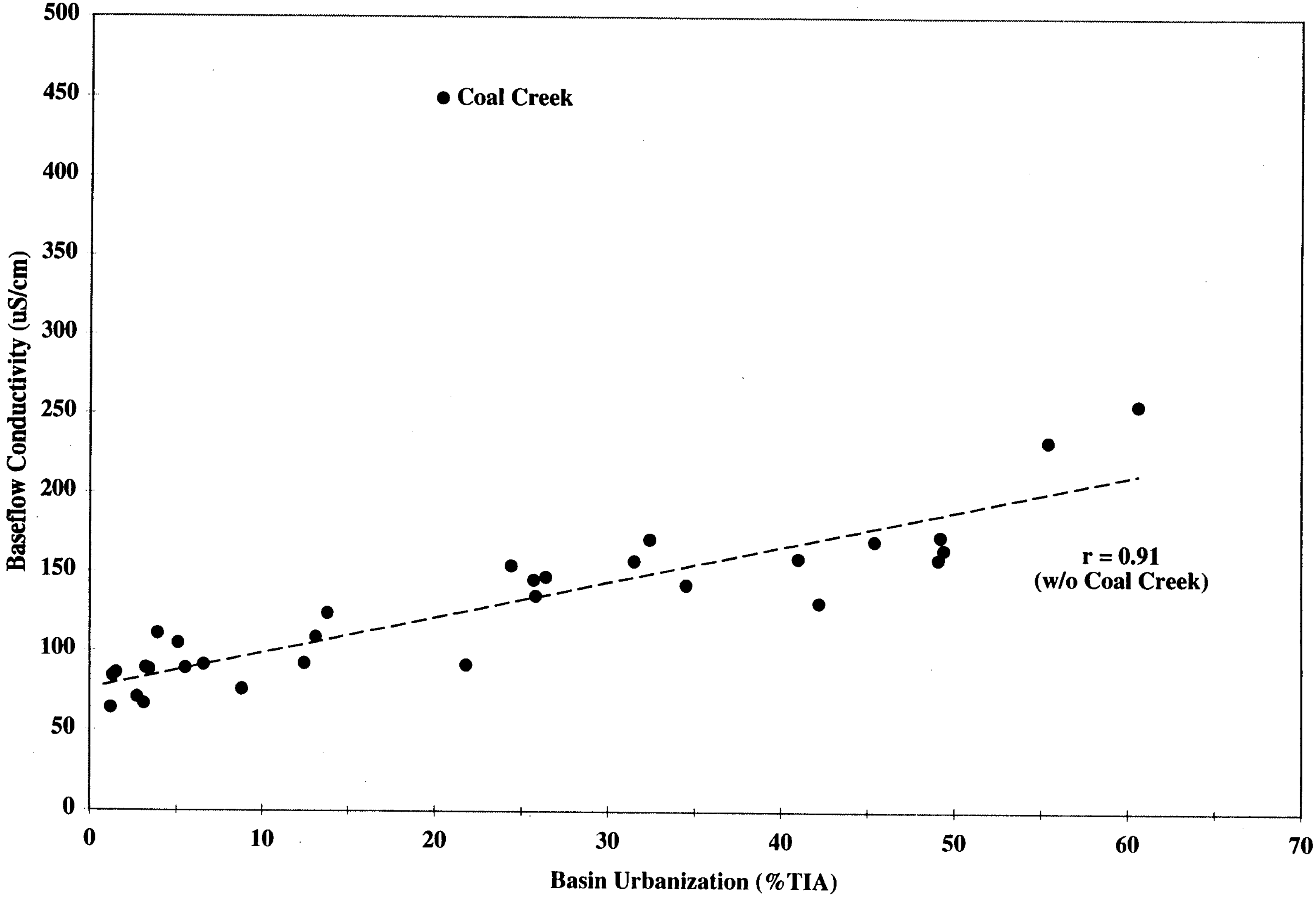

Figure 27: Relationship between watershed imperviousness (%TIA) and baseflow conductivity for Puget Sound lowland streams.

Surface DO was never a major problem even in the most urbanized streams. Water quality standards for Class AA in Washington waters state that DO shall exceed 9.5 mg/L and be <110% of saturation, temperature shall not exceed 16°C and pH shall be between 6.5 and 8.5. Results show that < 5% of almost 1000 samples taken over the 3 study years exceeded the standard. Low DOs were typically observed in areas of low flow, with an open canopy, and/or during a period of high summer temperature. In-stream DO was not related to urbanization. The lack of in-stream DO problems apparently indicate that BOD from either point or non-point sources (i.e., sewage or septic system leachate) and secondarily from inorganic nutrients via algae are relatively insignificant in these streams.

In only a few instances was the standard for pH violated (<1% of all samples) and half of those samples were from Coal Creek. Although temperature was determined only periodically, the data nevertheless indicated that high summer values may be a local problem only in the most urbanized streams. All but one stream with higher-than acceptable temperatures were in watersheds with TIA > 40%. The lone exception to this was Big Beef Creek at a site downstream from the very shallow and exposed Symington Reservoir (Bahls, 1993). High in-stream temperature has usually been linked with the loss of riparian canopy cover, which blocks the warming radiation input and is a common problem on urban streams. While neither lethal or sub-lethal limits for salmonids were exceeded, elevated temperature is nevertheless an additional stress to biota in stream with highly urbanized watersheds.

Storm-Flow

Event Mean Concentrations: Streams were sampled during selected storm events to obtain flow-weighted composite samples from which to calculate event-mean concentrations (EMC) for various constituents. Although conductivity was a good indicator of urbanization level under base-flow conditions, it was unrelated to urbanization during storm conditions (Bryant, 1995).

Storm EMCs for TSS for all storm-sizes were significantly greater than those during base-flow (Table 12). Moreover, the large-storm EMCs for TSS were at least an order of magnitude greater than the small or medium-sized storms (Table 12). TSS was not significantly different between small and medium storms, but there were significant differences between both small and large, as well as medium and large storms. Storms of greater magnitude, duration, and intensity created higher concentrations of TSS due to the greater stream power available for transporting sediment. TSS can originate directly from stormwater runoff, streambank erosion, and/or from resuspension of streambed sediment by storm discharge-induced turbulence. Larger storms also have the potential to flow into the floodplain and transfer debris or sediments into or out of the stream. The relationship between TSS EMC and development level (%TIA) indicates that, for all storm categories, there was a greater concentration of suspended solids in streams draining more urbanized watersheds (Table 12). Variability of TSS was very high (see coefficients of variation in Table 12) indicating that a large range of concentrations are possible for any given stream, development level, and storm combination. However, TSS was very low in PSL streams during normal base-flow conditions regardless of level of watershed development (Table 12).

TABLE 12: Summary of total suspended solids (TSS) data for Puget Sound lowland streams (1994-1996).

	Winter Baseflow	Small Storm EMC	Medium Storm EMC	Large Storm EMC
Mean (mg/L)	2.7	19.0	20.3	125.2
COV (%)	55	113	92	127
n	23	20	35	28
***r (%TIA)**	0.21	0.72**	0.64**	0.41**
***r (Precip)**	N/A	-0.65**	-0.15	0.48**

COV = coefficient of variation; EMC = event-mean concentration
* Spearman rank-correlation
** Significant at 0.05 level

Concentrations of nutrients were significantly different between base-flow and storm events of all sizes. With the exception of NO_3-N, all nutrients showed larger concentrations during storm than base-flow. Only total phosphorus (TP) showed a significant difference between stormflow levels. N and P concentrations were usually related directly with urbanization (%TIA), although several of the relationships were not significant nor consistent with storm magnitude. Table 13 summarizes the results of correlation analysis between nutrients and %TIA. TP increases as storm magnitude, intensity, and duration increased; much the same as TSS concentration (Table 14). This was not surprising, because a large fraction of TP is often associated with particulate matter. Step-wise linear regression also confirmed that precipitation magnitude and %TIA were the primary determinants of stormflow TP (EMC) ($r^2 = 0.44$).

Zinc (Zn) was considered representative of "urban" trace metals. Pb (lead) and Cu (copper) were also determined, but were usually in lower concentrations, relative to critical levels. The sources for these metals are primarily vehicles, water conveyance systems and commercial/industrial nonpoint activity. The difference between base-flow and stormflow TZn was significant, although no significant difference was detected in

TABLE 13: Correlation (Spearman) coefficients for nutrients vs. basin urbanization (%TIA) in Puget Sound lowland streams (1994-96).

Nutrient Constituent	Winter Baseflow	Small Storm EMC	Medium Storm EMC	Large Storm EMC
TP	0.75*	0.69*	0.49*	0.42*
SRP	0.56*	0.72*	0.56*	0.50*
NH_3-N	0.66*	0.82*	0.20	0.25
NO_X-N	-0.04	-0.10	-0.37	-0.25

* Significant at 0.05 level (EMC = event-mean concentration)

TABLE 14: Total phosphorus (TP) summary data for Puget Sound lowland streams (1994-96).

	Winter Baseflow	Small Storm EMC	Medium Storm EMC	Large Storm EMC
Mean (ug/L)	44.4	66.8	100.9	174.7
COV (%)	45	79	88	62
n	11	16	33	27
***r (%TIA)**	0.75**	0.69**	0.49**	0.42**
***r (Precip)**	N/A	-0.51**	-0.26	0.38**

COV = coefficient of variation; EMC = event-mean concentration
* Spearman rank-correlation
** Significant at 0.05 level

TZn between storm sizes (Table 15). However, TZn concentrations correlated well with level of watershed urbanization (Figure 28). Step-wise linear regression showed tha precipitation magnitude, %TIA, and riparian buffer width were the primary determinants of TZn EMC ($r^2 = 0.68$). Dissolved Zn (DZn) was about half that of TZn. No significant difference was detected in DZn between baseflow and stormflow, nor among stormflow levels (Table 16). The relationship between DZn and urbanization (%TIA) was very similar to that for TZn.

Trace metals were often below detection limits, especially in streams that were less urbanized. For example, more than 50% of the TZn concentrations from streams draining watersheds with TIA <20 % were <MDL (5 μg/L). This is one indication that chemical water quality does not pose a serious problem in most rural and suburban streams in the PSL and may partly explain no significant difference detected in metal content among storm sizes. More importantly, trace metal concentrations were below both state and federal water quality criteria thresholds in all but a very few highly urbanized streams.

TABLE 15: Total zinc (TZn) summary data for Puget Sound lowland streams (1994-96).

	Winter Baseflow	Small Storm EMC	Medium Storm EMC	Large Storm EMC
Mean (μg/L)	10.8	30.7	24.5	26.5
COV (%)	52	86	90	102
n	6	10	18	25
***r (%TIA)**	0.66	0.83**	0.75**	0.57**
***r (Precip)**	N/A	-0.71**	-0.71**	-0.71**

COV = coefficient of variation; EMC = event-mean concentration
* Spearman rank-correlation
** Significant at 0.05 level

TABLE 16: Dissolved zinc (DZn) summary data for Puget Sound lowland streams (1994-96).

	Winter Baseflow	Small Storm EMC	Medium Storm EMC	Large Storm EMC
Mean (μg/L)	8.7	9.1	16.2	10.9
COV (%)	37	60	77	88
n	6	11	9	13
***r (%TIA)**	0.78**	0.75**	0.51	0.55**
***r (Precip)**	N/A	-0.24	-0.60	-0.11

COV = coefficient of variation; EMC = event-mean concentration
* Spearman rank-correlation
** Significant at 0.05 level

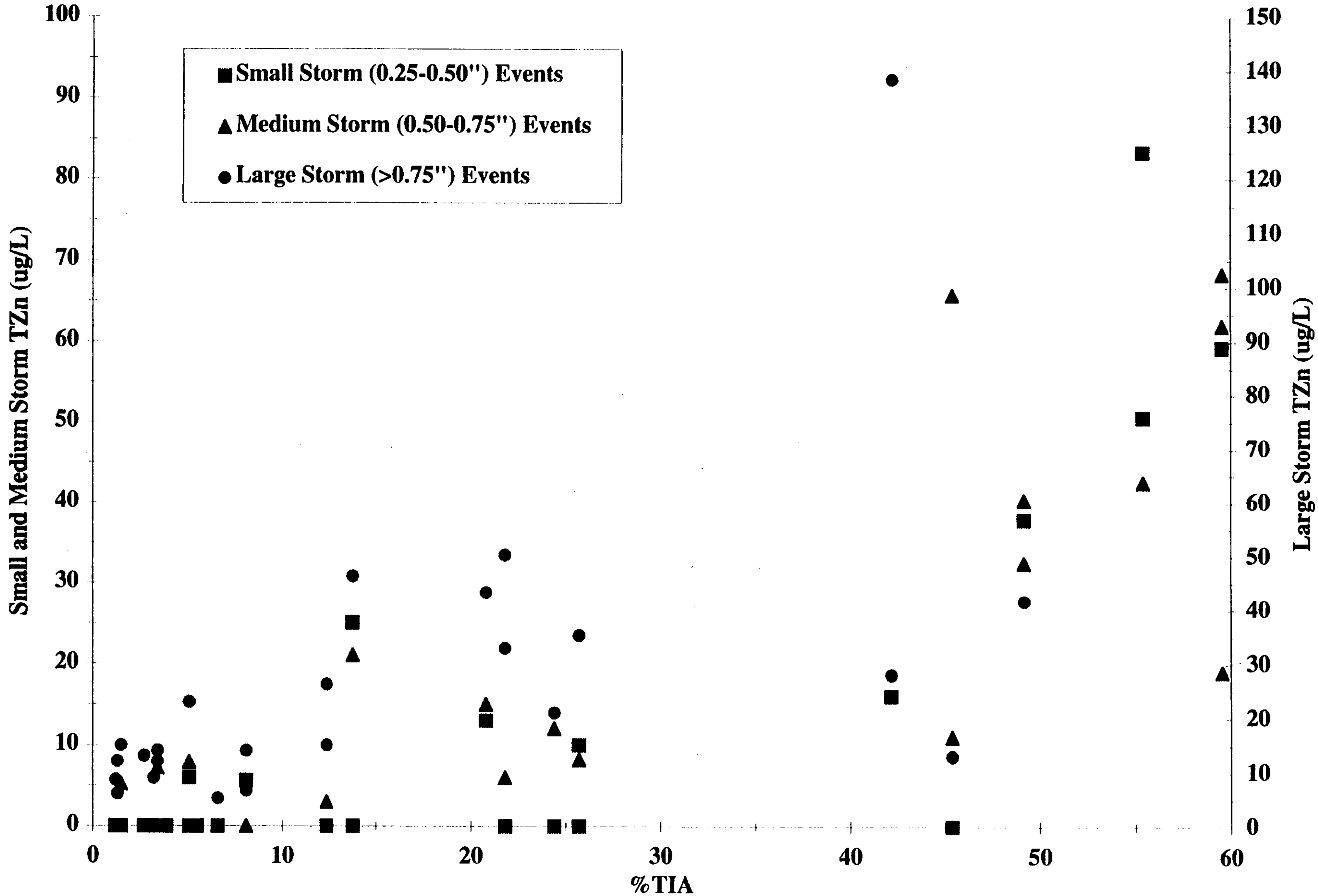

Figure 28: Storm event mean concentration (EMC) of total zinc (TZn) for streams in the Puget Sound lowlands for storms of different magnitudes (modified from Bryant, 1995).

In-stream Versus Stormwater Concentrations: EMCs for chemical water quality constituents in these streams were much lower than the typical concentrations reported for urban stormwater indicating an effect of instream dilution. The literature is replete with data on chemical constituents in stormwater, but little exists on the content in stream waters that receive the urban stormwater runoff. Chandler (1993) listed the typical concentrations of priority pollutants for the PS area, as well as those from the National Urban Runoff Program (NURP). Tables 17 (stormwater) and 18 (in-stream) show representative values for three constituents: TP, TSS and TZn. In-stream TP (EMCs) during storms was usually below the range observed in urban runoff. TP was higher than the regional mean for stormwater in only 5 out of 76 (7%) sampling events. In-stream storm EMCs for TSS were also well below the regional mean for urban runoff and in-stream TSS exceeded the regional mean for stormwater in only 14 out of 83 (17%) sampling events. Storm EMCs for TZn were an order of magnitude below the regional mean for urban runoff and no in-stream EMC for TZn exceeded the regional mean for stormwater. Thus, in-stream dilution of urban runoff is apparently significant.

TABLE 17: Typical urban runoff (stormwater) pollutant levels for the Puget Sound lowland (PSL) region (Chandler, 1993).

Constituent	PSL Mean	PSL Median	NURP Median
TP (μg/L)	320	240	420
TSS (mg/L)	93	81	183
TZn (μg/L)	210	180	210

TABLE 18: Stormflow event mean concentration (EMC) data for Puget Sound lowland streams (1994-96).

Constituent	Mean	Median	Maximum
TP (μg/L)	118	94	419
TSS (mg/L)	55	17	820
TZn (μg/L)	27	15	139

In-stream concentrations of chemical constituents during storms are generally much lower than those found in undiluted urban runoff. Values observed in these streams are comparable to the reported in-stream "wet-weather" concentrations reported previously for urban streams in King and Snohomish Counties. However, the City of Bellevue was the only cooperator to have collected flow-weighted composite samples during storms, similar to the method used here. Bellevue collected over 40 storm samples from both Coal and Kelsey Creeks during 1988-1992. Mean and median EMCs for TSS, TP, and TZn from the two data sets were within a factor of three (Table 19). Some difference could be expected, because the location, sample number, collection method, and basin development level (%TIA) for the events were not exactly the same. Bellevue's stormwater control/treatment systems have been steadily improving since the early 1990s, which may also partly explain their consistently higher values from 1988-1992, compared to those from this data set during 1994-1996. This may indicate that stream quality is actually improving, at least in Coal Creek.

Established maximum acceptable concentrations (MACs) for acute and chronic toxicity of TZn are 59 ug/L and 65 ug/L (for a hardness of 50 mg/L as $CaCO_3$; EPA 1986). Only 6 of the over 75 samples collected during this study had TZn concentrations that exceeded the chronic MAC and only 4 that exceeded the acute MAC and all excess

TABLE 19: Comparison of instream stormflow data for Coal Creek.

1988-92 (n=45) City of Bellevue Study	**TSS (mg/L)**	**TP (μg/L)**	**TZn (μg/L)**
Mean	266	270	58
Median	160	170	54
COV (%)	124	130	64
Maximum	1100	2000	160
Minimum	3	20	10
1994-96 (n=6) UW PSL Stream Study	**TSS (mg/L)**	**TP (μg/L)**	**TZn (μg/L)**
Mean	85	119	20
Median	76	60	15
COV (%)	104	71	79
Maximum	212	200	43
Minimum	2	20	7

concentrations were from streams draining the most highly urbanized basins (Thornton, Kelsey, Juanita, and Schneider Creeks), all with TIA > 40%. In-stream concentrations higher than the measured EMCs could have been present for a short period during storms, but not detected. However, about half the TZn, on average, was probably adsorbed onto particulate matter and, therefore not bioavailable (Bryant, 1995).

The obvious conclusion is that chemical water quality did not represent a major problem in most rural and suburban stream basins in the PSL, but was a potential problem in the most urbanized watersheds only (TIA > 40%). However, chemical quality may contribute, although in a minor way, to the total effect on biota in these streams. Nevertheless, other substances not determined here, such as pesticides, poly aromatic compounds and other organics, may be contributing to the overall effect.

Sediment Chemical Concentrations

Sediment samples were collected from each stream during the summer of 1994 and analyzed for trace metals and hydrocarbons derived from petroleum products. All samples were <MDL for hydrocarbons. Pb and Zn were the only sediment metal constituents that showed a significant correlation with basin %TIA. Sediment toxicity guidelines were exceeded for Pb or Zn in only the most highly urbanized streams, which was consistent with water content of Zn. Sediment from only Thornton Creek was above the lowest effect threshold limit for Pb (31 mg/kg). None of the stream sediments exceeded the lowest effect threshold established by the WA DOE (1991) for Zn (120 mg/kg) and none were above the severe effect threshold for either Pb (250 mg/kg) or Zn (820 mg/kg).

Riparian Corridors

The relationship between riparian buffer width and the level of development was examined on both a watershed and stream-segment scale. The minimum effective riparian buffer width was considered to be 10 m (30 ft) and a width of 30m (100 ft) would usually meet most functional requirements (Johnson and Ryba, 1992). Assessment was based on these two guidelines.

Riparian Buffer Integrity

A strong correlation was observed between riparian buffer width and basin imperviousness (%TIA) with both analysis scales. On a watershed scale, the fraction of riparian buffer >30 m wide was usually over 70% for the undeveloped (TIA < 5%) streams (Figure 29a). Both Green Cove and Big Bear Creeks also had substantial portions (>60%) of their riparian buffer >30 m wide. These streams are representative of the rural, low TIA (5-10%) basins. Only one urbanized stream (Schneider Creek), with an intact riparian zone located in a steeply-sided ravine, had greater than 70% of its riparian buffer wider than 30 m. Development in this basin extends to the ravine edge. If the riparian buffer width were measured from the slope-break of the ravine instead of the active channel, this stream would have <10% of its buffer width >30 m. Analysis at the stream-segment level shows similar results, although, as expected, there was more variation due to the finer scale of measurement (Figure 30a).

As an indication of encroachment, the fraction of riparian buffer < 10 m wide was also strongly correlated with urbanization on both the whole-watershed and stream-segment scales (see Figures 29b and 30b). In general, only undeveloped, reference (TIA <5%) streams had <10% of their riparian buffer in a nonfunctional condition (<10 m wide). Riparian encroachment increased proportionally with urbanization, except for three streams. Covington Creek (TIA <5%) had slightly greater than 15% riparian buffer width <10 m. A large fraction of this rural stream runs parallel to one of the major roads, along which most development was clustered. Green Cove and Schneider Creeks also did not conform (Figure 29b). Nevertheless, these criteria (% >30 m and % <10 m) provided an acceptable definition of lateral riparian integrity.

Each riparian buffer segment was also examined qualitatively considering the dominant land-use within 30 m of the stream. Mature forest, young forest, and wetlands were considered "natural" in contrast to a riparian zone with predominantly residential or

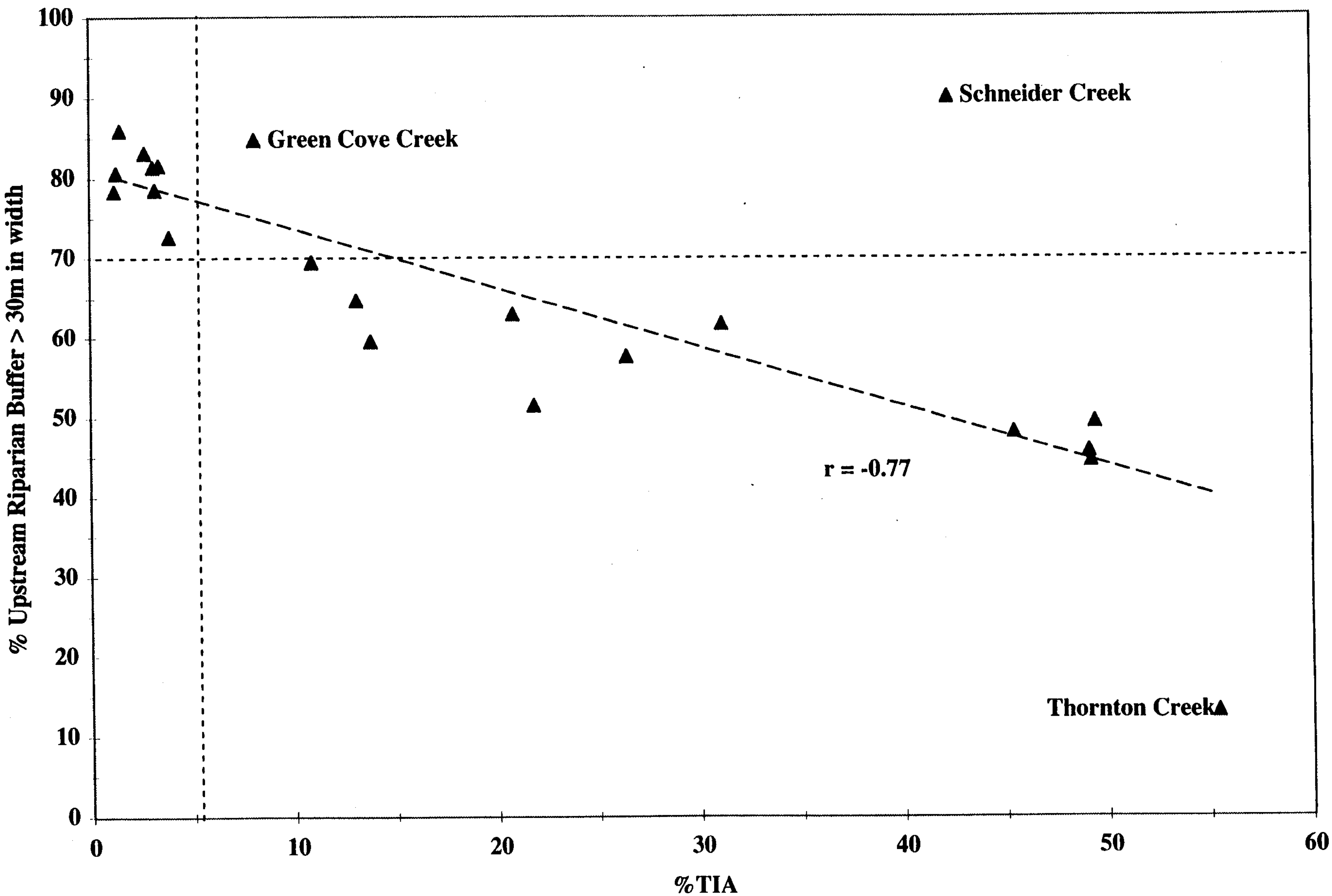

Figure 29a: Relationship between riparian integrity (% buffer width > 30 m) and basin urbanization (%TIA) on a watershed scale in Puget Sound lowland streams.

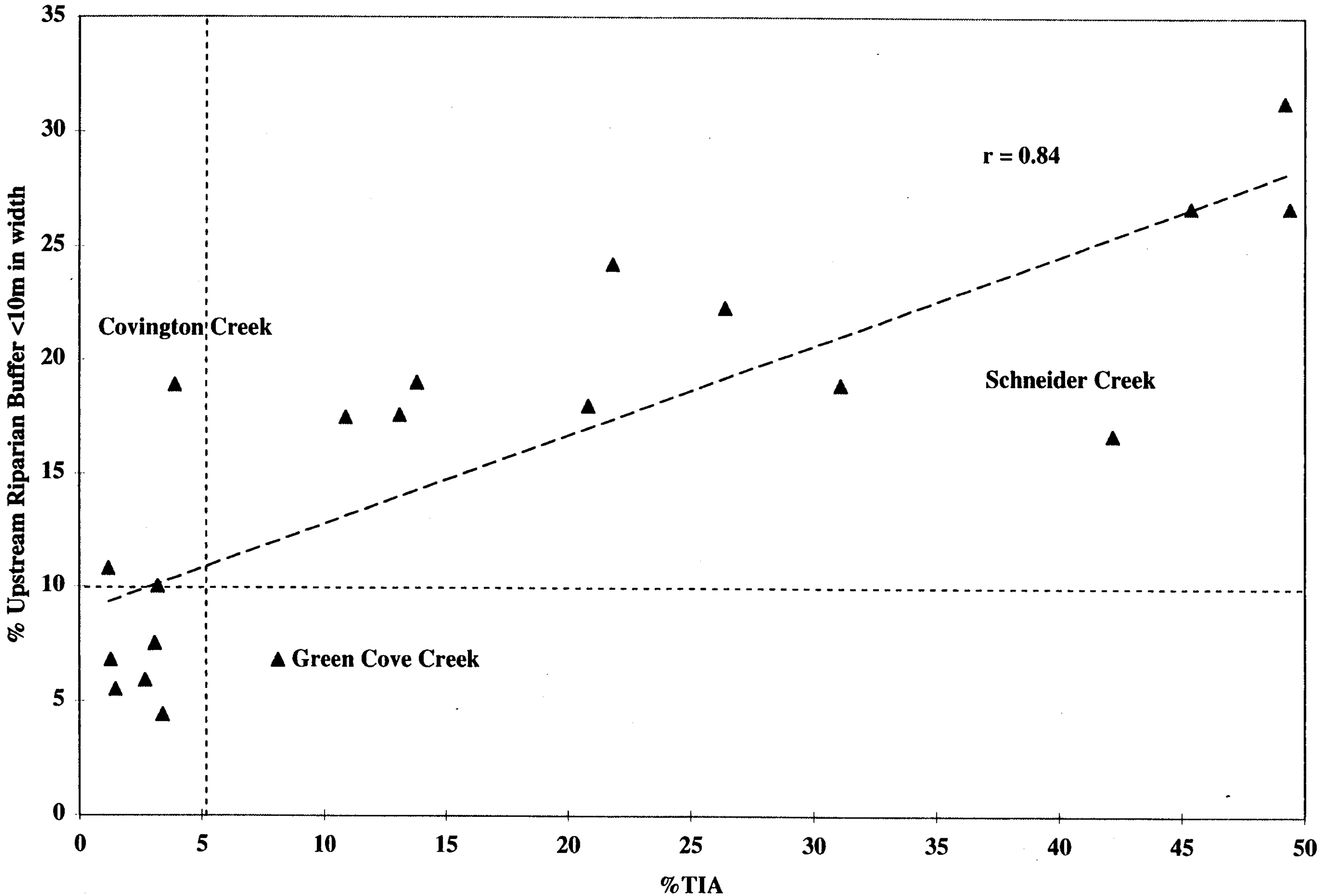

Figure 29b: Relationship between riparianencroachment (% buffer width < 10 m) and basin urbanization (%TIA) on a watershed scale in Puget Sound lowland streams.

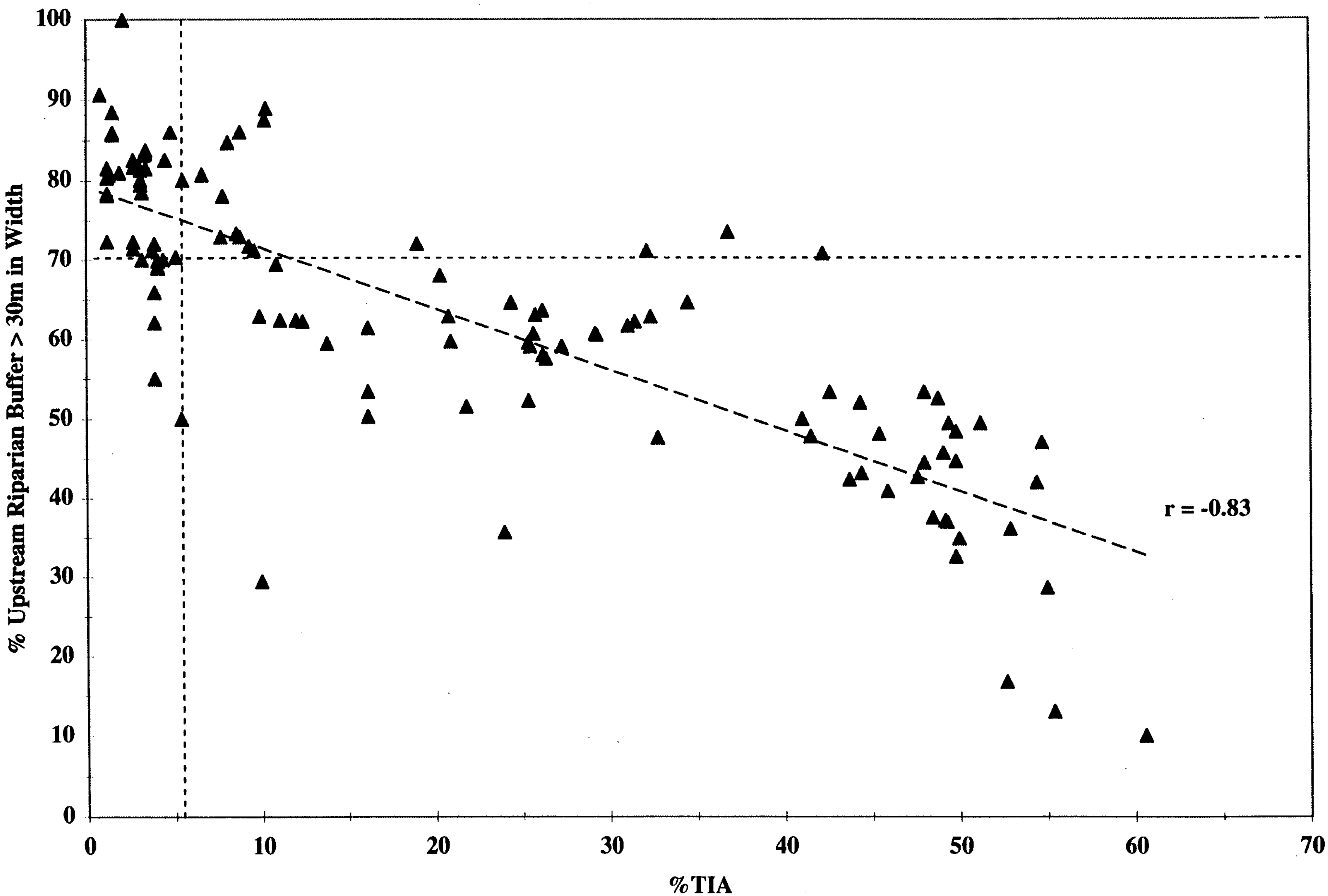

Figure 30a: Relationship between riparian integrity (% buffer width > 30 m) and basin urbanization (%TIA) on a stream segment scale for Puget Sound lowland streams.

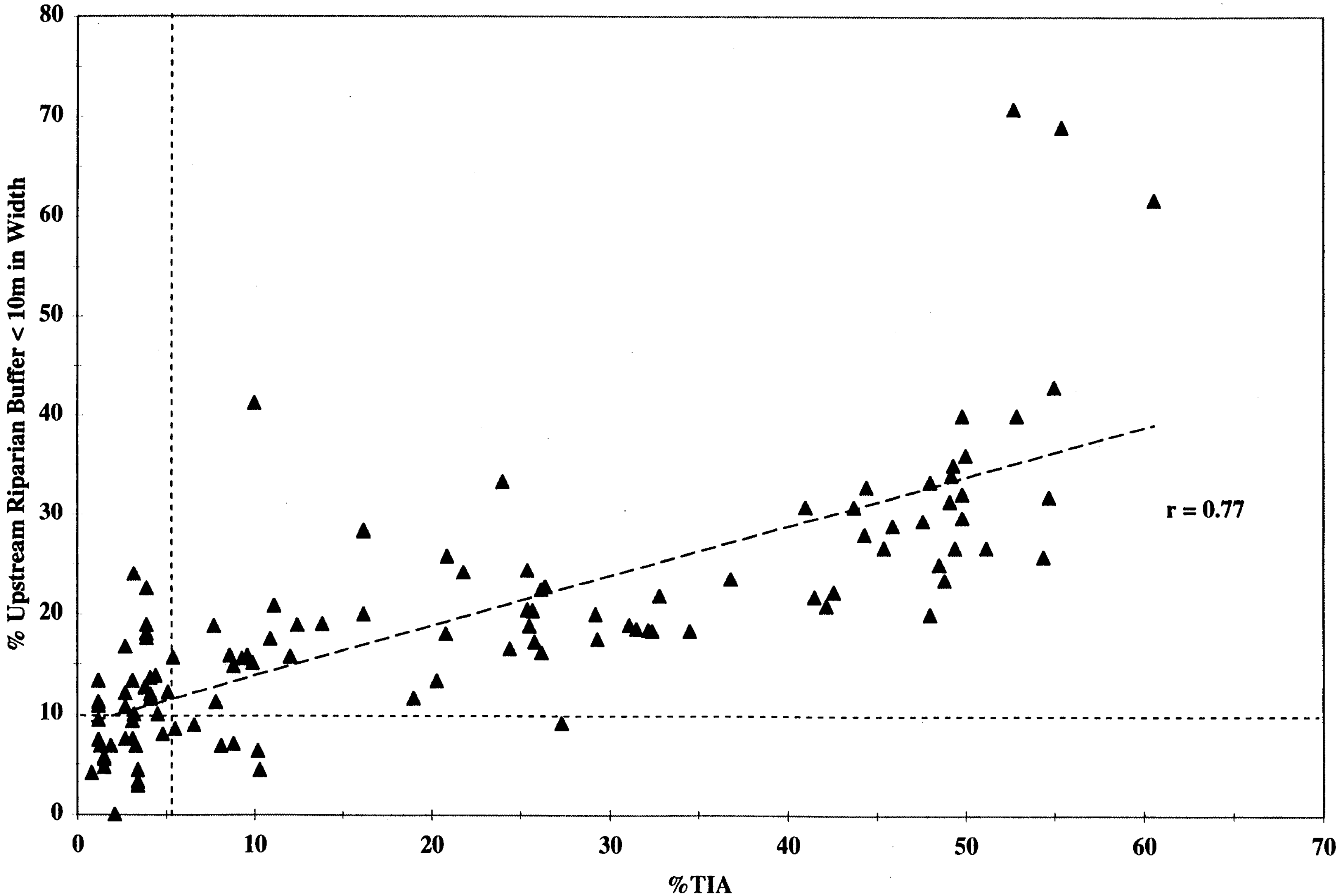

Figure 30b: Relationship between riparian encroachment (% buffer width < 10 m) and basin urbanization (%TIA) on a stream segment scale for Puget Sound lowland streams.

commercial development in the immediate vicinity. Mature forest and/or riparian wetlands are the two most desirable natural riparian conditions (Gregory et al, 1991). Figure 31 shows the percentage of riparian corridor in each watershed composed of mature forest and riparian wetland (%TIA is shown for comparison). Only the undeveloped, reference streams (TIA <5%) had a majority of their riparian corridor as mature forest (40% or greater), although the riparian wetland component varied considerably across the range of urbanization. Also shown on Figure 31 is the fraction of the floodplain that remained intact for each stream system. With the exception of Coal and Schneider Creeks, which have intact riparian zones, no urbanized streams retained more than 25% of their natural floodplains.

Riparian Continuity

Breaks in the riparian zone were tallied for each stream and survey segment as a measure of connectivity or "longitudinal integrity". Road-crossings, pipelines, and utility right-of-ways were enumerated as breaks. This value was normalized per km of stream or segment length. Breaks in the riparian corridor were directly related to urbanization on both the watershed and stream-segment scales (see Figure 11b for drainage density on a watershed scale).

The less urbanized streams tended to have less than one riparian-break (stream-crossing)/km. As would be expected, the number of riparian breaks (mostly road-crossings) were closely linked to basin road-density. Longitudinal riparian integrity was strongly correlated with urbanization (%TIA), as were lateral riparian integrity and riparian quality. Hence, riparian quantity and quality are excellent indicators of the effect of urbanization. However, the close association between riparian condition and urbanization makes the task of separating out riparian from other effects for mitigation purposes.

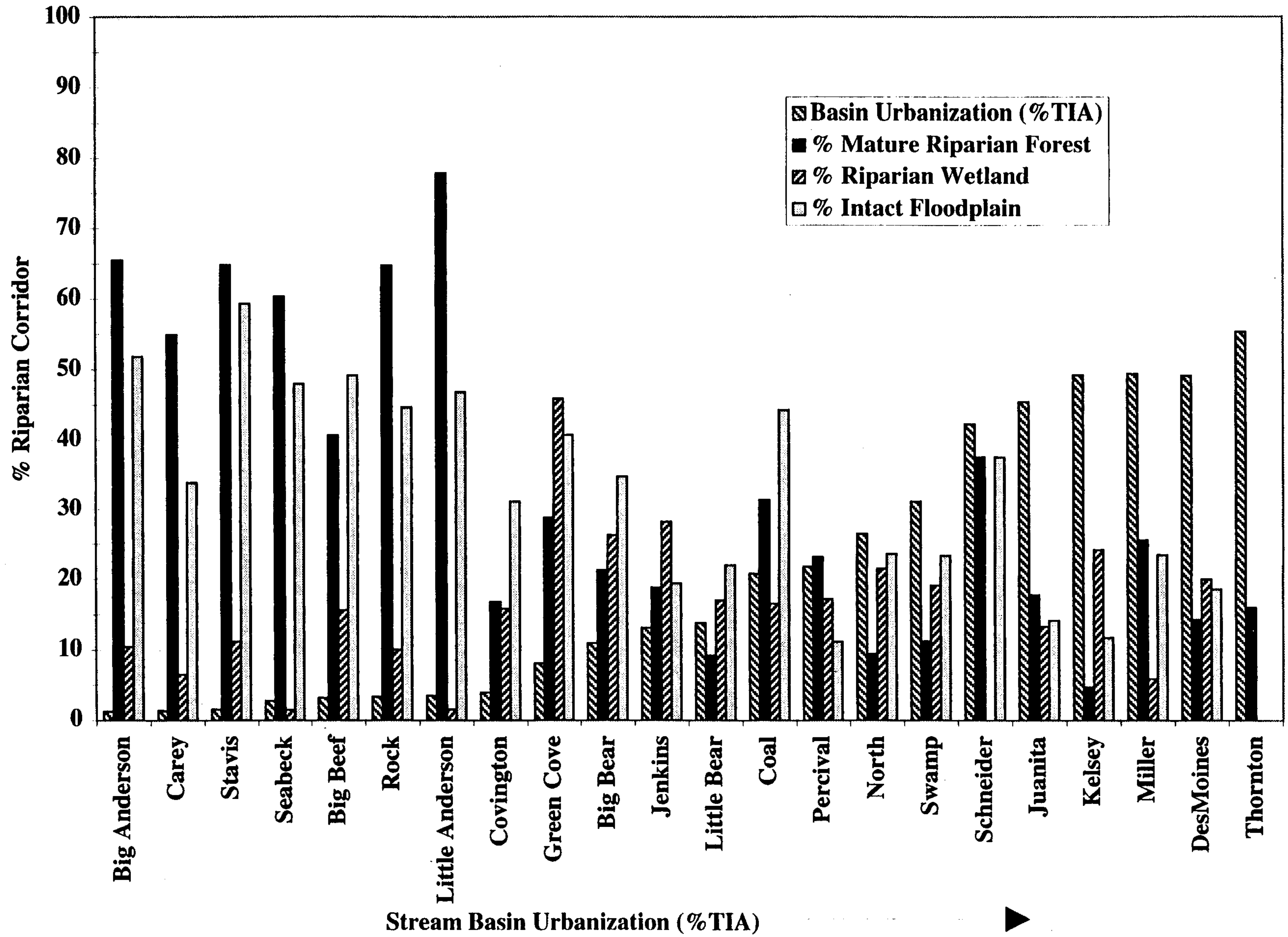

Figure 31: Riparian integrity (quality) for Puget Sound lowland streams (arranged by increasing %TIA).

Instream Salmonid Habitat

Habitat characteristics for salmonid spawning and rearing were surveyed and analyzed on the stream-segment scale only. Instream habitat quantity and quality were locally variable and, therefore, extrapolating these results to the stream or watershed level would be inappropriate. Instream habitat characteristics were also compared to basin-level characteristics to investigate possible links between urbanization and salmonid habitat.

Spawning Habitat

The wetted surface area of riffle sections of each stream-segment was measured as an index of salmon spawning habitat. That area was then recorded as a fraction of the total stream area. No significant relationship was observed between the level of urbanization and the fraction of stream area comprised of riffles, nor that of riparian integrity, and % riffle area. If those segments classed as wetland (low gradient areas are usually more suitable for rearing than spawning) were excluded, then only one reference stream segment (TIA <5%) had a riffle fraction less than 40%. A balance between pool and riffle habitat is usually considered optimum, so the nominal riffle fraction should be about 40-60% (Peterson et al, 1992). The quantity of spawning (riffle) habitat was usually adequate in all but the most urbanized (TIA >45%) PSL streams.

Fine Sediment: The quality of spawning and egg-incubating gravel is usually judged by the quantity of fine sediment. Percent fines in streambed gravels vary both spatially and temporally because of the dynamic nature of streams. The fraction of fine sediment (<0.85 mm) from a McNeil core-sampler is referred to as "% fines" and, if great enough, can be detrimental to salmonid survival (Chapman, 1988). A relationship between % fines and urbanization was evident (Figure 32; Olthof, 1994 and Wydzga, 1996). With only one exception, elevated fine sediment content (>15% fines) was not detected in reaches with a sub-basin imperviousness <20% TIA, which was more typical of undeveloped and rural watersheds. Local stream gradient also affects the deposition of fine sediment to the streambed. Average flows are apparently unable to consistently flush

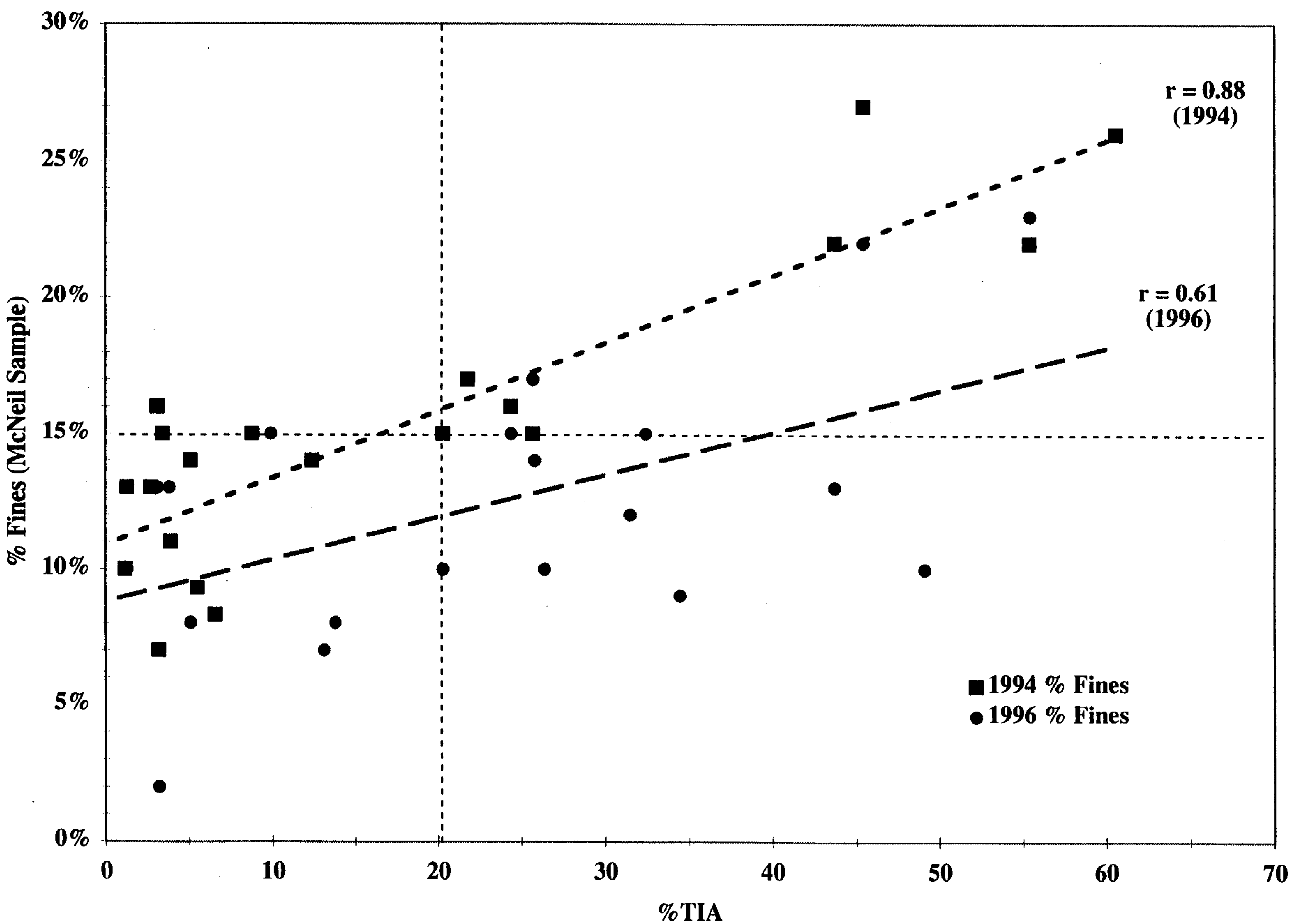

Figure 32: Relationship between urbanization (%TIA) and % fine sediment (% Fines) in Puget Sound lowland streams. Dashed lines indicate possible thresholds of adverse effects (Olthof, 1994 and Wydzga, 1997).

fines from the streambed in low-gradient reaches (<0.7%). All the sample sites with fines >20% were from low-gradient reaches (Wydzga, 1996). Cooper (1996) also demonstrated a relationship between fine sediment and hydrologic regime (Figure 18).

As an alternative to the labor-intensive sampling with a McNeil core, "zig-zag" pebble-counts were compared with % fines. As illustrated in Figure 26 and Table 10, the D10 pebble-count value was strongly related to % fines, indicating that this easier method could be substituted for the McNeil procedure to indicate riffle spawning quality.

Substrata Embeddedness: This was another potential indicator of riffle quality, although embeddedness was poorly related with urbanization. As would be expected, however, substrata embeddedness was related to the amount of fine sediment. Apparently, some combination of stream gradient, flow, and sediment supply determines the degree of substrata embeddedness. Cooper (1996) found that at high stormflow/baseflow ratios (>20), embeddedness was consistently greater than 20% and as high as 60% in streams which also had high % fines (Figure 18). Embeddedness was also by him to affect benthic biota, as would be expected.

Intragravel dissolved oxygen (IGDO): IGDO was determined periodically in artificially-constructed salmonid (coho) redds. Instream DO was measured coincidentally with IGDO monitoring. There was no significant difference among instream DO concentrations across the spectrum of urbanization. However, mean IGDO concentrations showed a negative trend as urbanization increased, presumably due to the influence of fine sediment. IGDO and % fine sediment were inversely related, each responding to the effects of urbanization. IGDO was not related to any of the measures of riparian integrity, nor did riparian characteristics explain the variability in the relationship between IGDO and % fines. Like scour and fine sediment measurements, IGDO may be significantly affected by local conditions as well as watershed and stream-segment factors. Nevertheless, the IGDO/DO exchange ratio reflected a general negative trend with increasing sub-basin urbanization (Figure 33). Only one reference reach had an IGDO/DO interchange ratio less than 80%. Above a TIA of 5%, there was considerable scatter in the IGDO/DO data, but most were less than 80%.

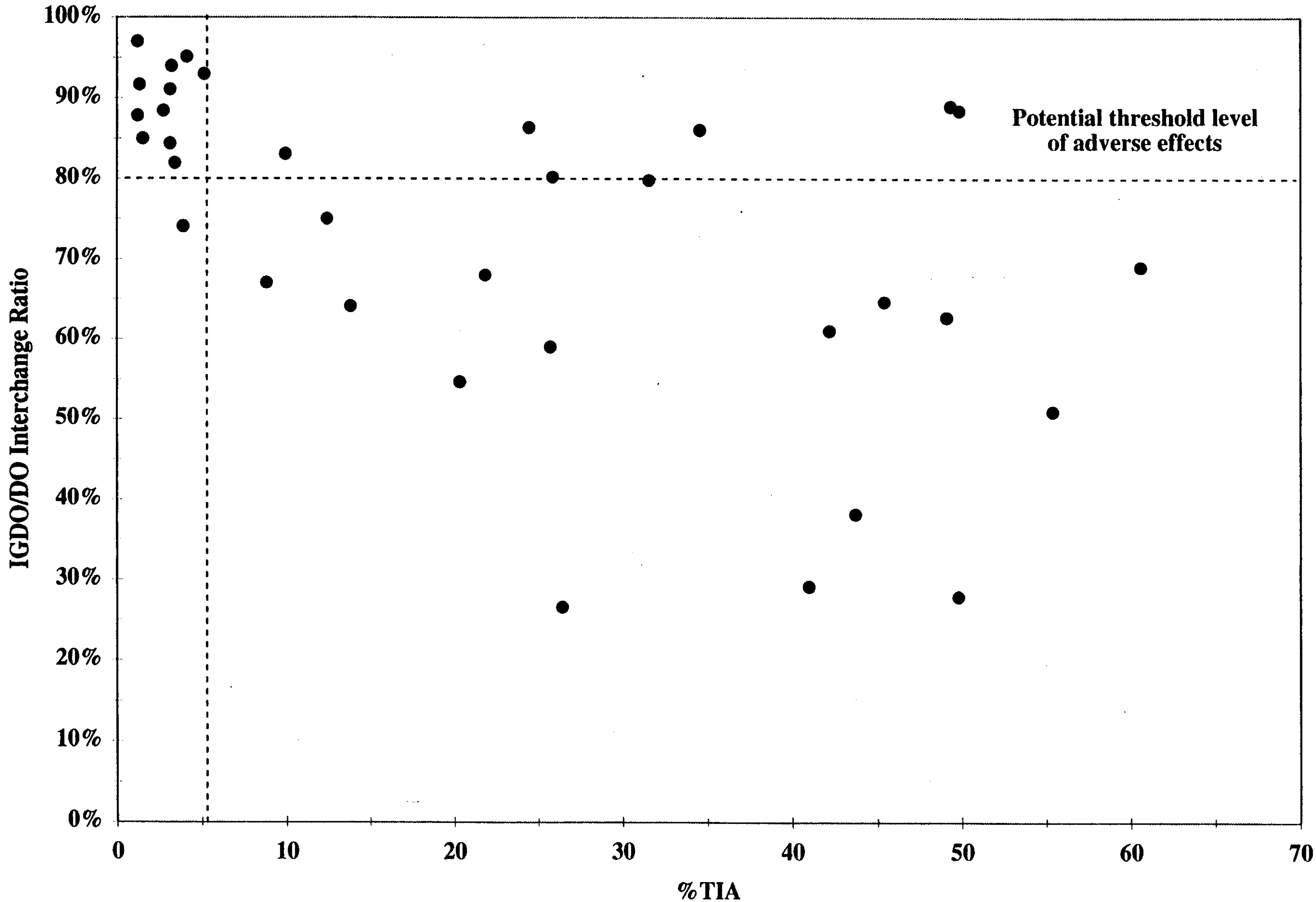

Figure 33: Relationship between sub-basin urbanization (%TIA) and the mean intragravel dissolved oxygen interchange ratio (IGDO/DO) for Puget Sound streams during salmonid incubation period.

Undeveloped, reference streams (TIA <5%) typically had IGDO levels very close to instream DO, which was consistently at or near saturation. Sample reaches on reference streams also showed little variability in DO or IGDO measurements on either a spatial (within riffles) or temporal scale (see Figure 34). Apparently, fine sediments were often flushed from riffles, which improved the DO exchange. The relative abundance of LWD in undeveloped streams also influences fine sediment production, storage, transport, and deposition.

On the other hand, reaches located in more urbanized sub-basins typically had IGDO levels well below the level of instream DO (see Figure 35). Local variability between monitors in the same riffle was quite common in urbanized stream reaches. Temporal variability in IGDO was also noted in most urbanized streams, presumably due to some variability in flow and/or fine sediment levels. In most cases, the minimum IGDO measured, even in the most urbanized streams, was not low enough to adversely effect salmon-to-egg survival (<5 mg/L; US EPA, 1986), although they were low enough (<saturation) to adversely affect fry size (Shumway et al., 1964). Table 20 contains a summary of IGDO and instream DO data (see May, 1996, Appendix G for a complete set.

Rearing Habitat

Pools: The relative amount of pools is considered an ecologically meaningful indicator of instream rearing habitat (Bisson et al, 1982). The pool quantity was determined as that fraction of stream (wetted) surface (slow, relatively deep water with appropriate cover). A balance between spawning and rearing habitat is ideal. Thus, as with riffle habitat, 50% as pool habitat is considered optimum (Peterson et al, 1992).

Cumulative upstream development (%TIA) significantly decreased the fraction of pool area in streams (Figure 36). The relationship, although not strong, is significant and shows that no stream segment with TIA >5% had a pool area fraction greater than 50%. More surprising was the observation that the majority of reference stream segments also had <50% pool habitat. This indicates that in addition to the negative effects of

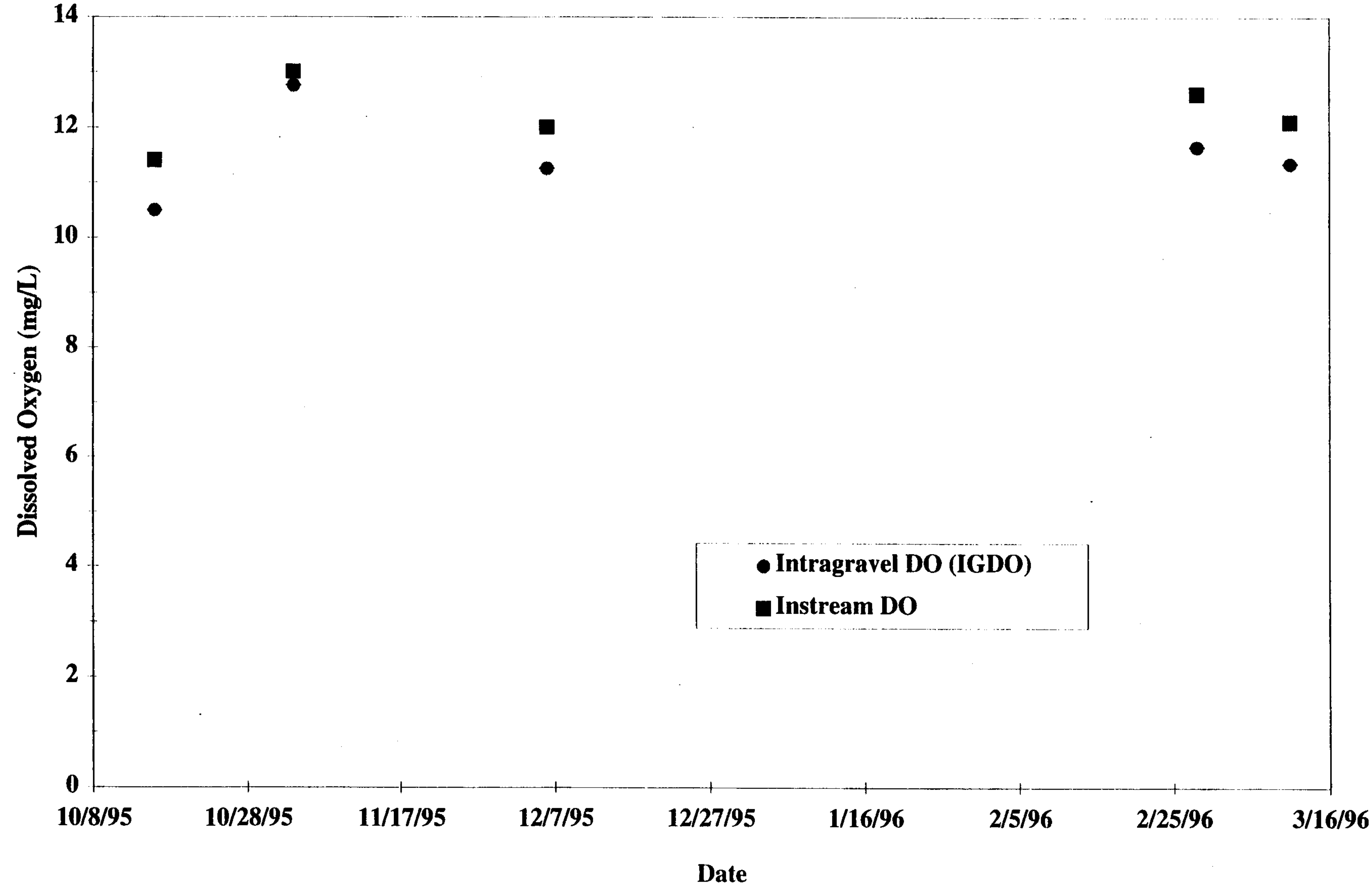

Figure 34a: Intragravel dissolved oxygen (IGDO) and instream DO in lower Rock Creek .

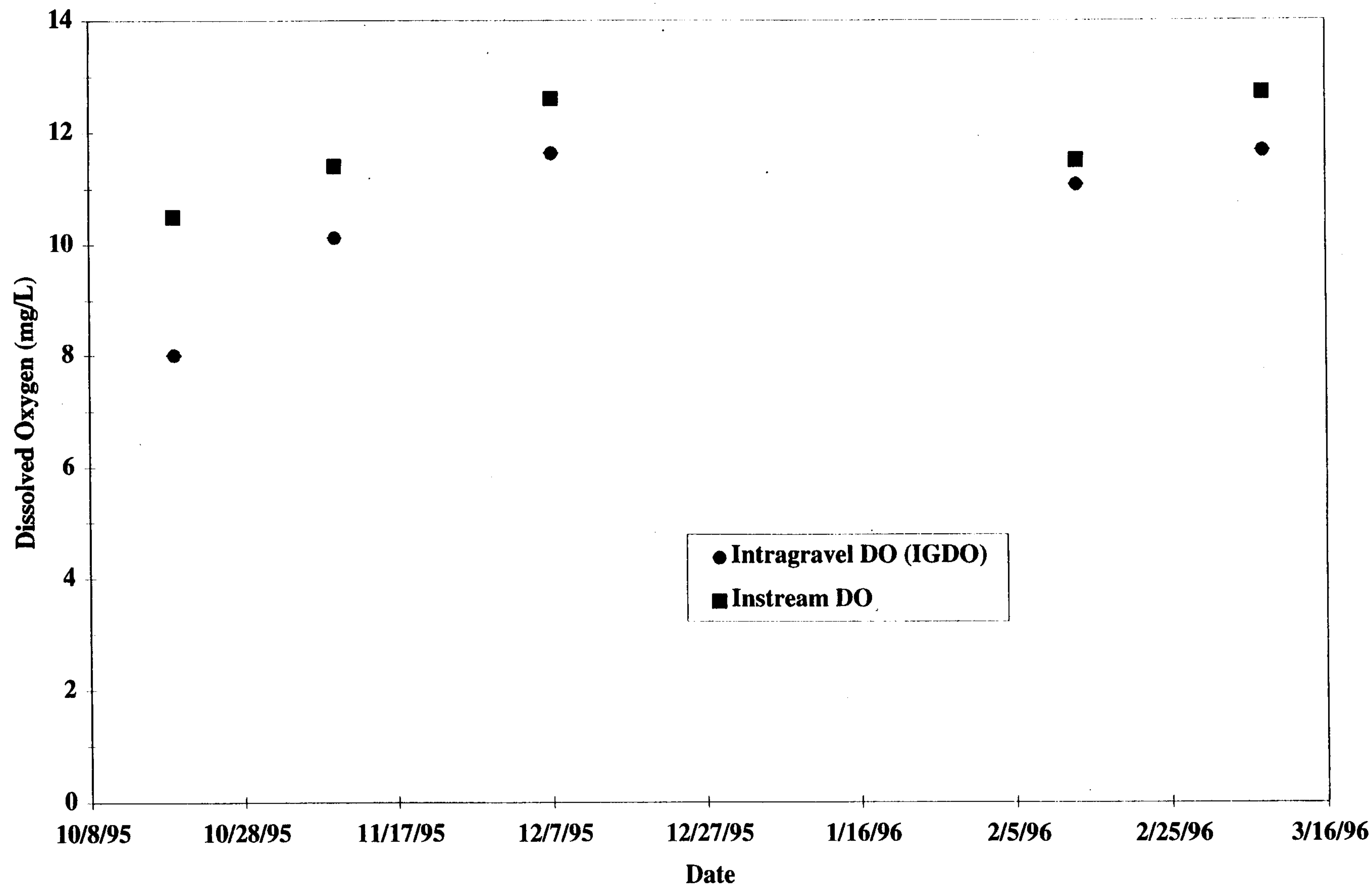

Figure 34b: Intragravel dissolved oxygen (IGDO) and instream DO in lower Seabeck Creek.

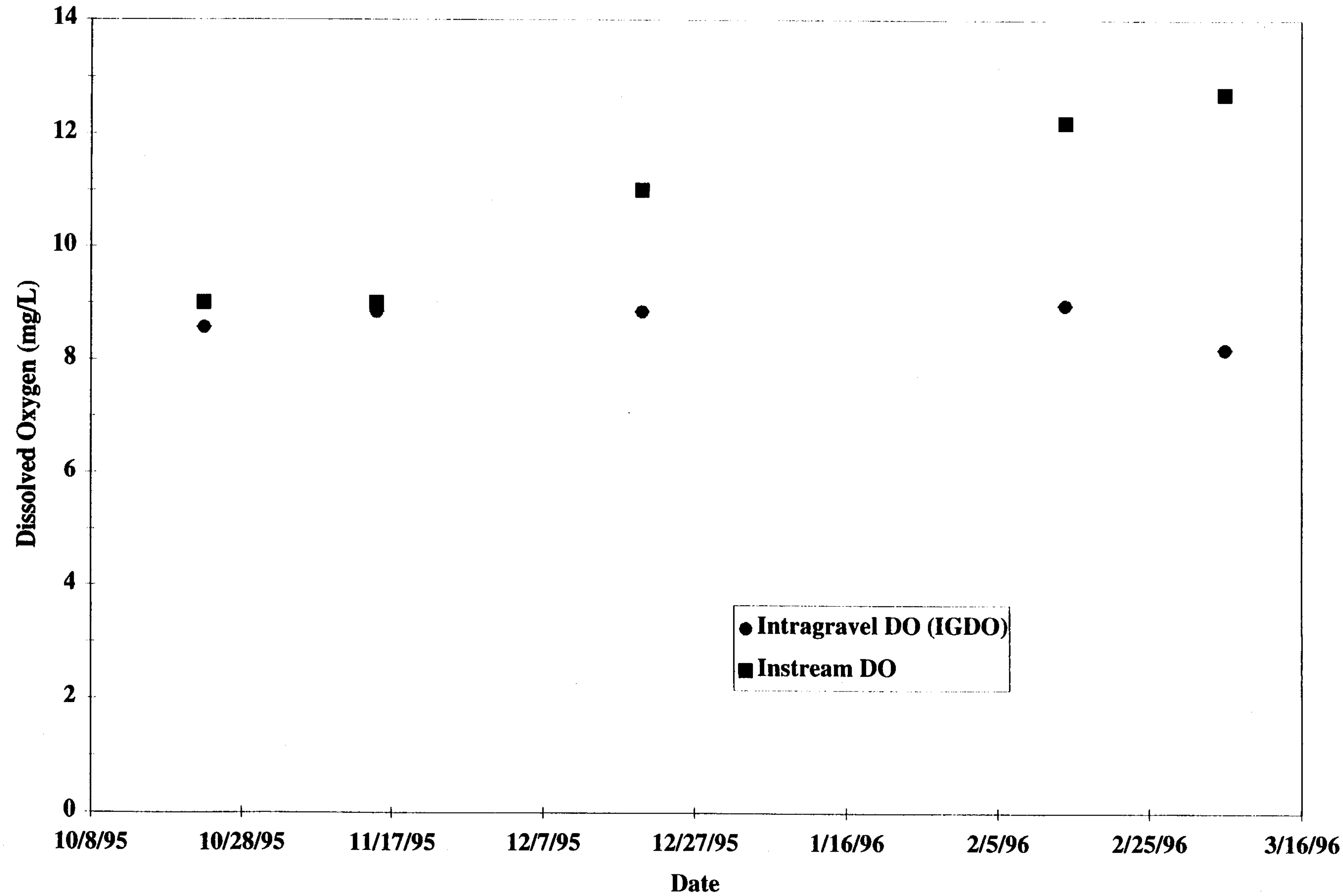

Figure 35a: Intragravel dissolved oxygen (IGDO) and instream DO in upper Swamp Creek.

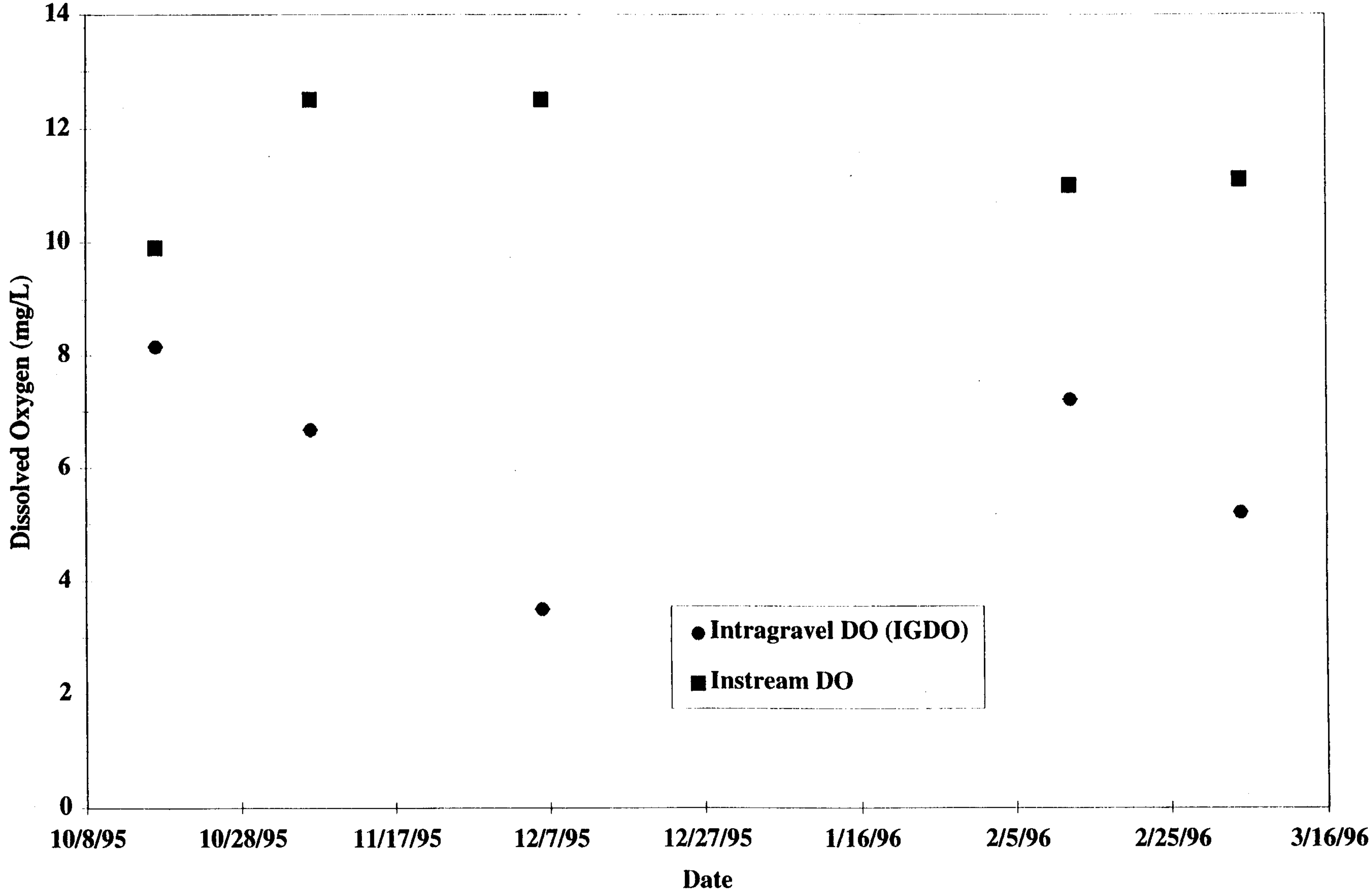

Figure 35b: Intragravel dissolved oxygen (IGDO) and instream DO in middle Coal Creek.

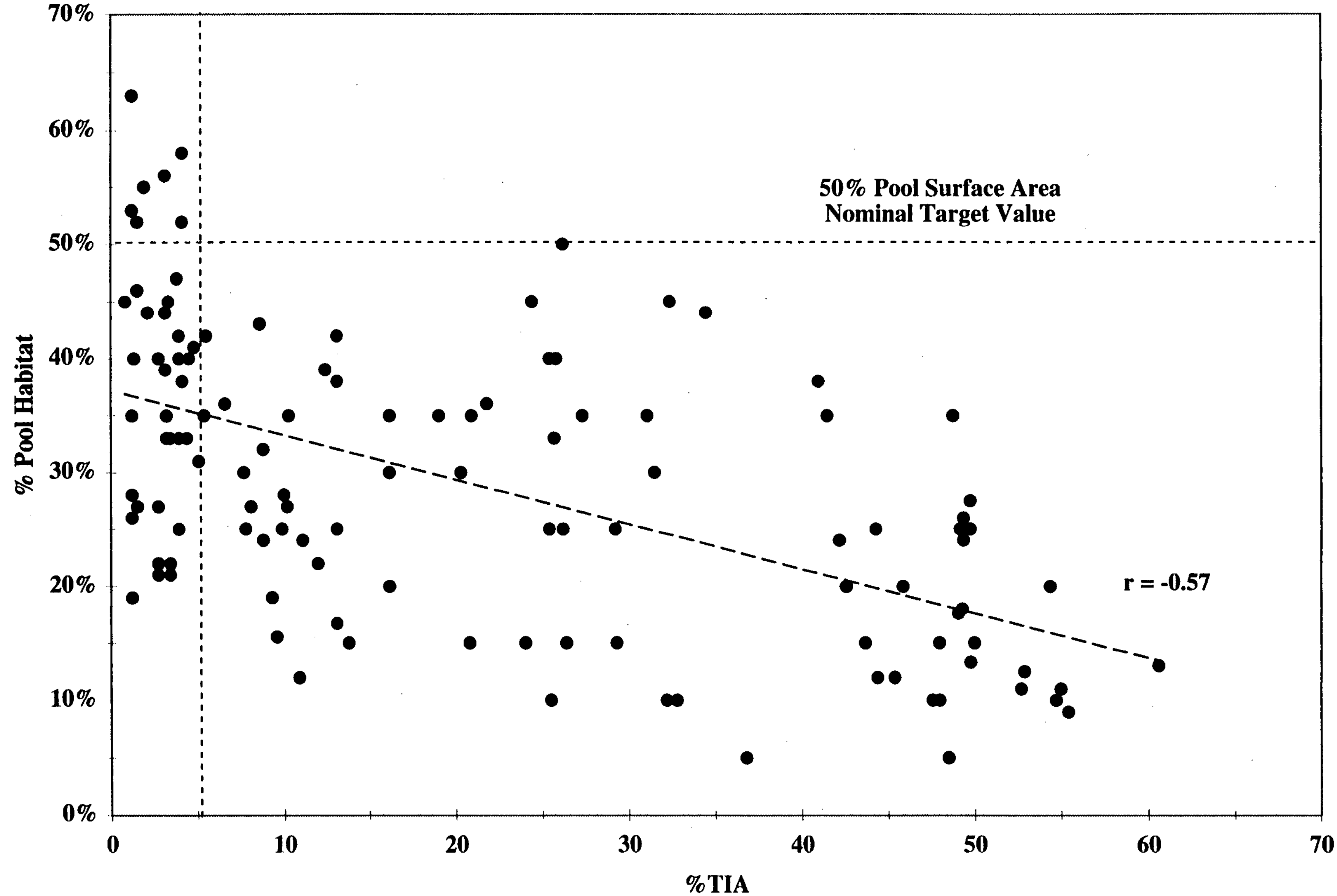

Figure 36: Relationship between sub-basin urbanization (%TIA) and salmonid rearing habitat (% pool surface area) in Puget Sound lowland streams.

Table 20 : Summary of instream and intragravel dissolved oxygen (IGDO) data for Puget Sound lowland streams.

STREAM SEGMENT	Sub-Basin %TIA	94-95 Seasonal DO % Saturation			Minimum IGDO (mg/L)	95-96 Seasonal DO % Saturation			Minimum IGDO (mg/L)
		Instream Mean	IGDO Mean	IGDO/DO		Instream Mean	IGDO Mean	IGDO/DO	
Big Anderson (Lower)	1.2	93.8%	91.0%	97.0%	9.1	100.2%	88.0%	87.8%	8.6
Carey (Middle)	1.3	92.0%	69.0%	75.0%	2.1	98.0%	89.9%	91.7%	8.8
Stavis (Lower)	1.5	93.3%	84.0%	90.0%	8.8	97.4%	82.8%	85.0%	8.5
Seabeck (Lower)	2.7	93.5%	87.0%	93.0%	8.0	97.4%	86.1%	88.4%	6.8
Big Beef (Below Lake Symington)	3.1	91.8%	78.0%	85.0%	6.3	101.2%	92.2%	91.1%	9.5
Rock (Upper)	3.1					99.7%	84.1%	84.4%	9.1
Rock (Lower)	3.2	94.9%	93.0%	98.0%	10.5	102.2%	96.1%	94.0%	10.2
Little Anderson (Middle)	3.4	94.7%	89.0%	94.0%	9.4	98.4%	80.7%	82.0%	6.6
Covington (Lower)	3.9	94.6%	70.0%	74.0%	2.0				
Big Beef (Above Lake Symington)	4.1					91.2%	86.7%	95.1%	9.3
Little Bear (Upper/Rural)	5.1	92.5%	74.0%	80.0%	5.5	96.5%	89.7%	93.0%	9.6
Big Bear (Upper)	5.5	90.2%	46.0%	51.0%	2.2				
Big Bear (Lower)	6.6	90.9%	50.0%	55.0%	1.9				
Green Cove (Middle)	8.8	85.1%	57.0%	67.0%	4.5				
Little Bear (SR 9/Middle)	9.9					101.0%	83.9%	83.1%	7.4
Percival (Upper)	12.4	93.3%	70.0%	75.0%	7.1				
Little Bear (Lower/Woodinville)	13.8	93.9%	77.0%	82.0%	8.3	99.2%	63.6%	64.1%	6.0
Coal (Middle)	20.3	95.1%	78.0%	82.0%	6.4	98.9%	54.1%	54.7%	2.5
Percival (Lower)	21.8	95.6%	65.0%	68.0%	8.1				
Swamp (Middle/Cypress)	24.4	90.3%	28.0%	31.0%	0.6	96.5%	14.9%	15.4%	1.3
North (Middle/UW Site)	25.7	91.5%	54.0%	59.0%	1.2				
Swamp (Upper/Butternut)	25.8					90.3%	72.4%	80.2%	6.1
North (Lower/County Line)	26.4					98.9%	26.3%	26.6%	1.8
Swamp (Lower/Wallace Park)	31.5					98.3%	78.4%	79.8%	8.2
Swamp (Middle/Larch)	32.4					96.0%	82.9%	86.4%	7.8
North (Middle/Mill Creek)	34.5					98.9%	85.2%	86.1%	7.9
North (Lower/McCollum Park)	41.0					103.1%	30.1%	29.2%	2.0
Schneider (Lower)	42.2	93.4%	57.0%	61.0%	6.6				
Juanita (Lower)	45.4	90.6%	48.0%	53.0%	2.5	95.4%	61.6%	64.6%	3.2
DesMoines (Lower)	49.1					88.4%	55.4%	62.7%	4.9
Kelsey (Lower/Park)	49.2	97.1%	66.0%	68.0%	6.6	98.7%	37.7%	38.2%	3.9
Miller (Middle)	49.3					104.2%	92.7%	89.0%	9.4
DesMoines (Middle)	49.8					98.2%	27.4%	27.9%	0.4
DesMoines (Upper)	49.8					90.1%	79.7%	88.5%	9.6
Thornton (Mainstem)	55.4	92.5%	62.0%	67.0%	3.9	97.8%	50.7%	51.8%	1.5
Thornton (South Branch)	60.6	94.2%	65.0%	69.0%	5.6				

urbanization on rearing habitat, the background condition of rearing habitat is below optimum in PSL streams. This may be due to the residual effects of past land-use practices such as timber-harvest and agriculture.

This characteristic was also related to the level of sub-basin development. Pool frequency is often normalized for stream size using BFW spacing rather than pools/km. Based on requirements for salmonid migration (holding pools) and rearing (refuges), a target of 2 BFW-spacings or less between pools has been proposed for forested, low-gradient streams in the PNW (Peterson et al, 1992). The only PSL stream segments with a pool-spacing of <2 BFWs were in undeveloped sub-basins (Figure 37). For undeveloped streams (TIA <5%), BFW-pool-spacings varied from 0.5 to 6.5 BFWs between pools. In contrast, the BFW pool spacing varied from a bit more than 2 to nearly 25 BFWs between pools in the more urbanized streams(TIA >30%).

The average size of pools (mean pool area) and % of cover on pools was apparently reduced by the cumulative effects development. However, no significant relationship was found between mean residual pool depth and cumulative upstream development. In fact, deep-scour pools were common on many urbanized streams. These pools were often associated with culverts and other unnatural structures. While providing depth refuge for salmonids during baseflow conditions, these pools often lacked cover and seemed to provide little refuge from high flows.

The relationship between rearing habitat and riparian integrity also reflected the effects of urbanization. Pool quantity (surface area fraction and frequency) was positively affected by the maintenance of wide riparian buffers (Figure 38). Stream segments with >50% of their surface area as pools were found only in reaches with 60% or more of riparian buffer wider than 30 m. However, a wide, intact riparian buffer was not a guarantee of adequate rearing habitat. Several stream segments with between 60-90% of their buffer wider than 30 m, had <30% pools, again demonstrating that instream habitat is a function of multiple external variables. In addition, a pool frequency of 2 BFW spacings or less was also found only in stream-segments with >60% riparian buffer wider than 30 m. Pool quality (% cover) was also positively influenced by riparian

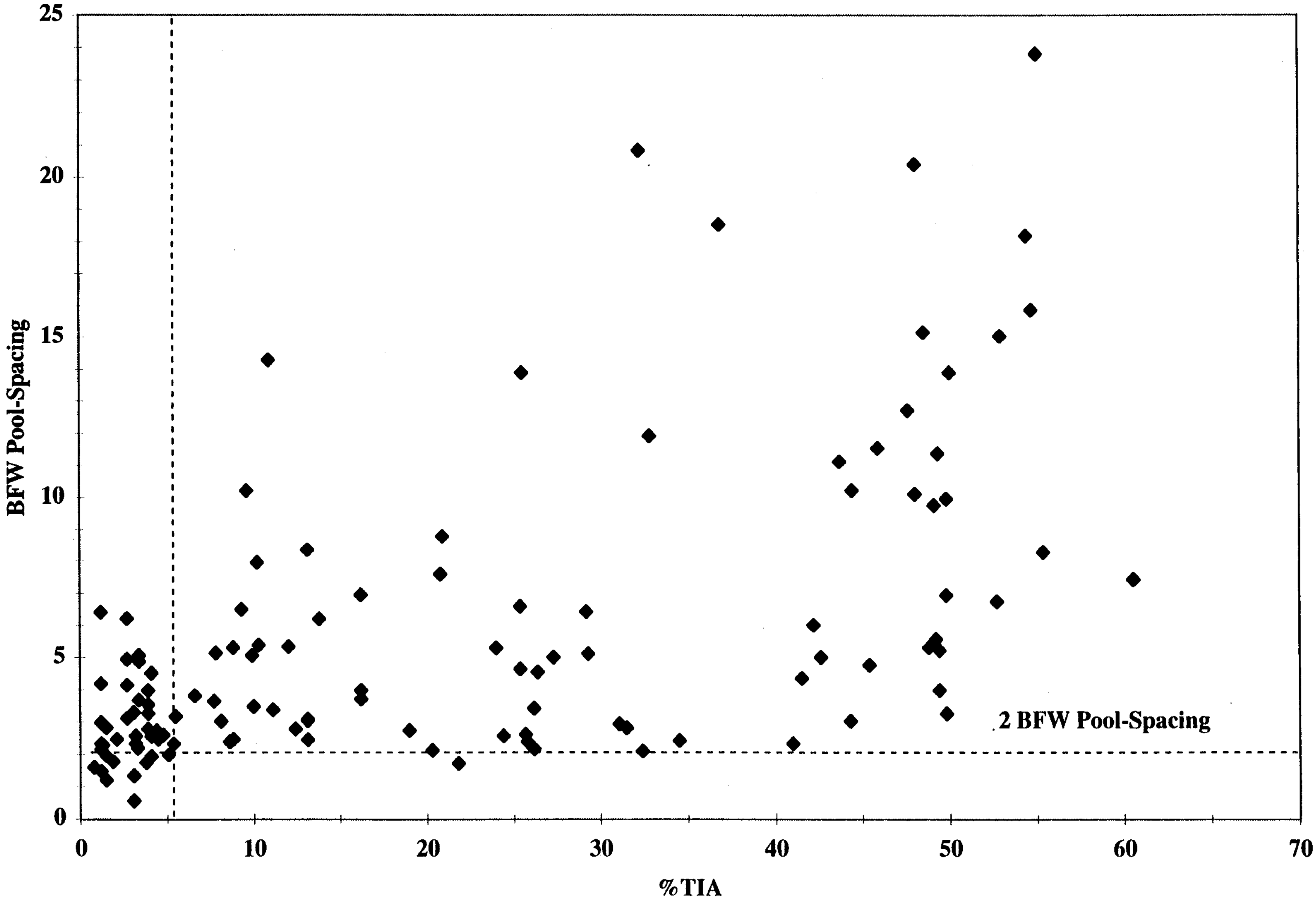

Figure 37: Relationship between sub-basin urbanization (%TIA) and pool habitat frequency (bankfull width spacings) in Puget Sound lowland streams.

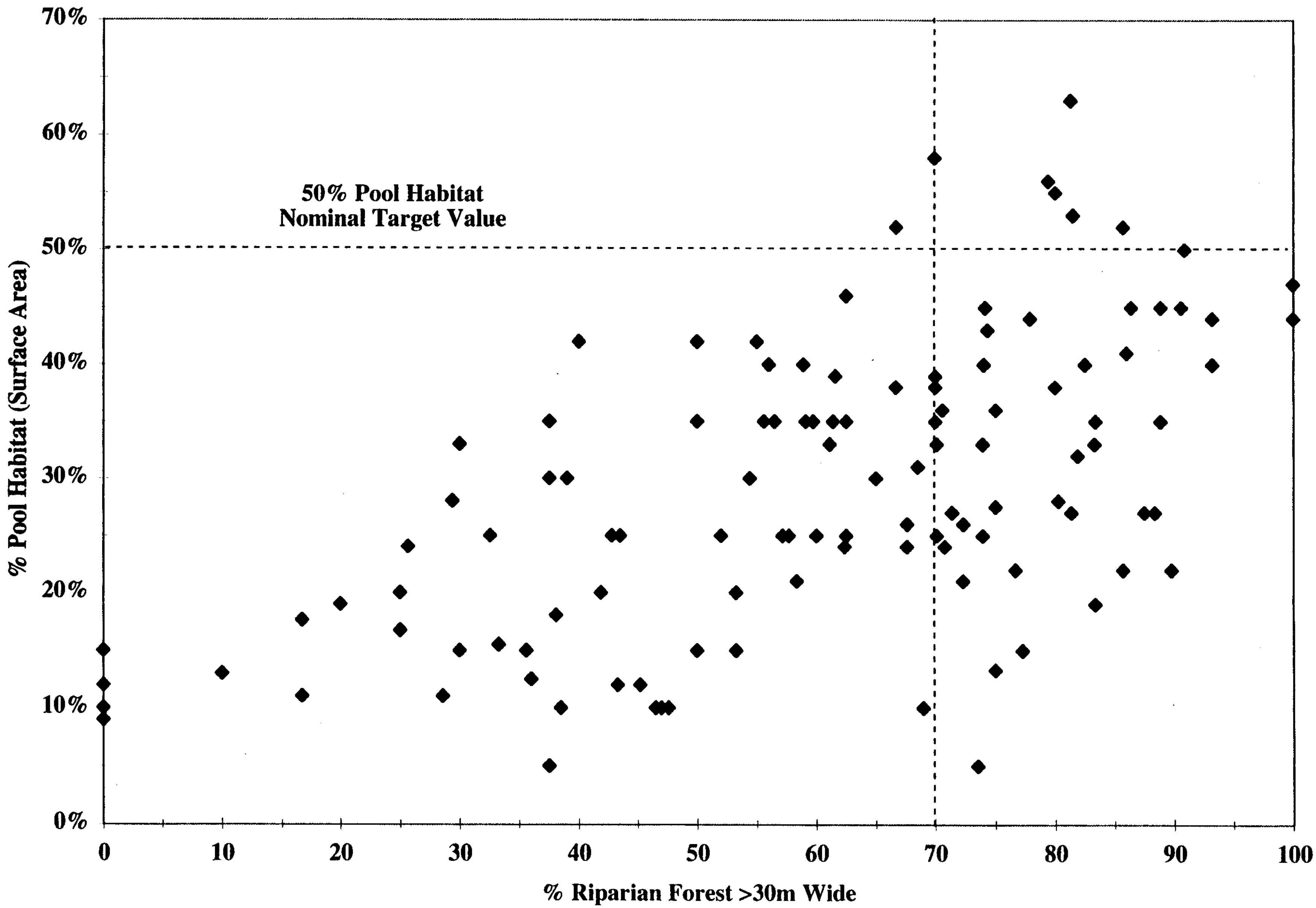

Figure 38: Relationship between riparian buffer width and salmonid rearing habitat in Puget Sound lowland streams. Dashed lines indicate possible thresholds of adverse effect.

integrity (buffer width and quality). Riparian encroachment, as measured by the fraction of buffer less than 10 m in width, also had a significant negative impact on instream rearing habitat. Stream segments with greater than 50% pools were found only if no more than 15% of the riparian buffer was <10 m in width. The same was true for segments with pool frequency <2 BFW spacings. Again, maintaining a large fraction of the riparian buffer functionally intact (>10 m) does not guarantee adequate rearing habitat, but it usually does benefit instream habitat quantity and quality. The linkage between riparian conditions and instream habitat depends on many variables, the most important being LWD recruitment.

Large Woody Debris: LWD frequency (#/km) decreased substantially as cumulative upstream development increased (Figure 39). Total LWD volume (m^3/km) also decreased with increasing upstream development. The latter relation showed less variability with fewer extreme values, indicating that there were fewer pieces of LWD in urbanized streams (Figure 39). The few extreme values (high LWD highly urbanized sub-basins) that did exist could be partly explained by the presence of mature riparian forest. LWD has been installed in two such segments in an effort to restore the stream channel. One is the Adopt-a-Stream Foundation's site on North Creek at McCollum Park and the other is the Trout Unlimited site on lower Miller Creek (Figure 39).

There was also a strong relationship between LWD quantity and the 2-year stormflow/baseflow ratio, indicating that instream LWD may be dependent on washout (Cooper, 1996).

LWD was also strongly related to riparian buffers, both quantitatively and qualitatively. LWD frequency (#/km) and total volume (m^3/km) were substantially influenced by riparian buffer width. LWD is apparently largely absent in the stream channel unless >70% of the riparian buffer is at least 30 m (Figure 40). The inverse is true for riparian encroachment. If >10% of the riparian buffer is effectively eliminated (<10 m wide), then LWD will probably be largely absent in the stream (Figure 41). Riparian quality (i.e., the fraction that is natural) also strongly affected LWD quantity. These results indicate that substantial frequency and volume of LWD will probably be

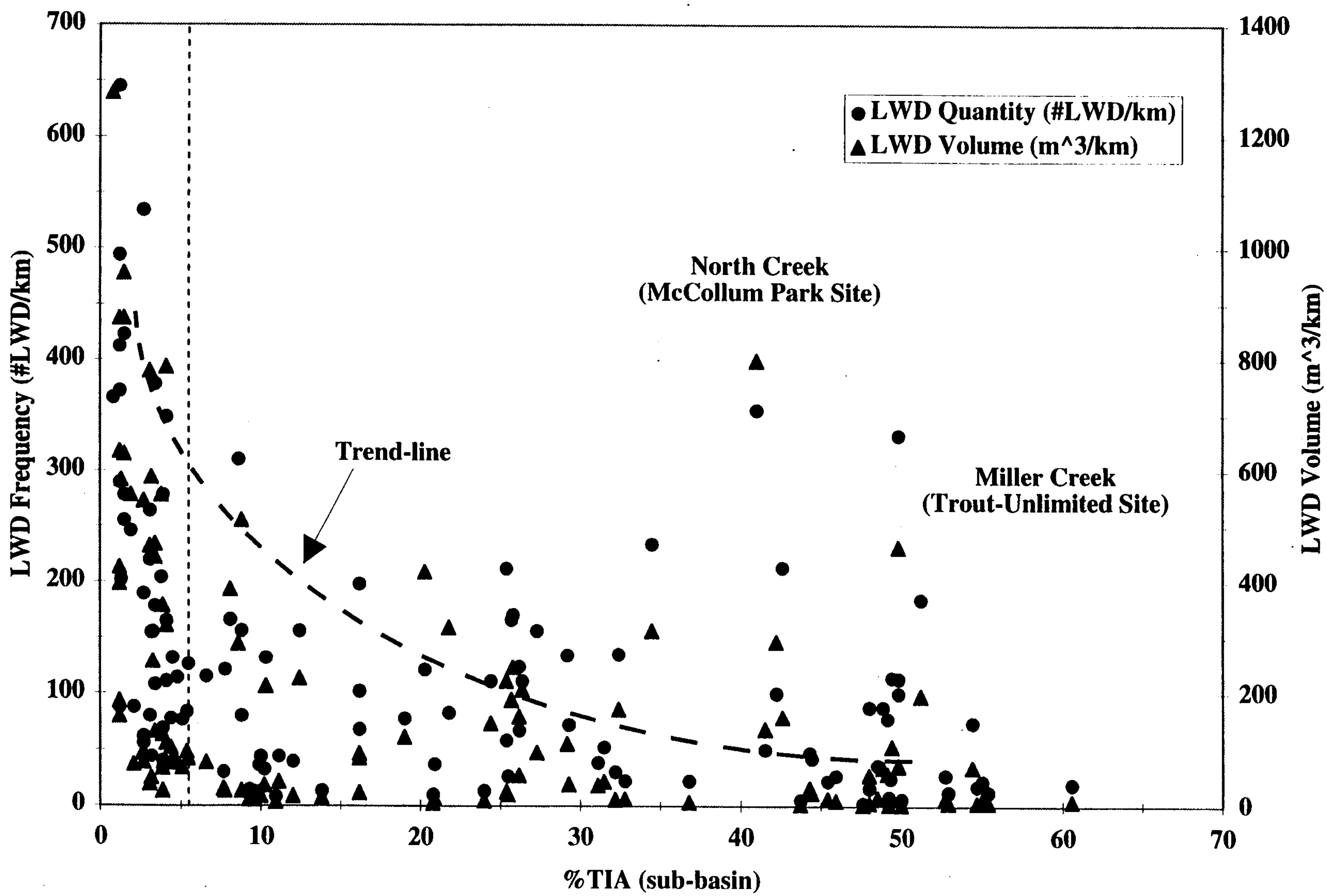

Figure 39: Relationship between sub-basin urbanization (%TIA) and large woody debris (LWD) in Puget Sound lowland streams. Vertical dashed-line indicates possible threshold of adverse effect.

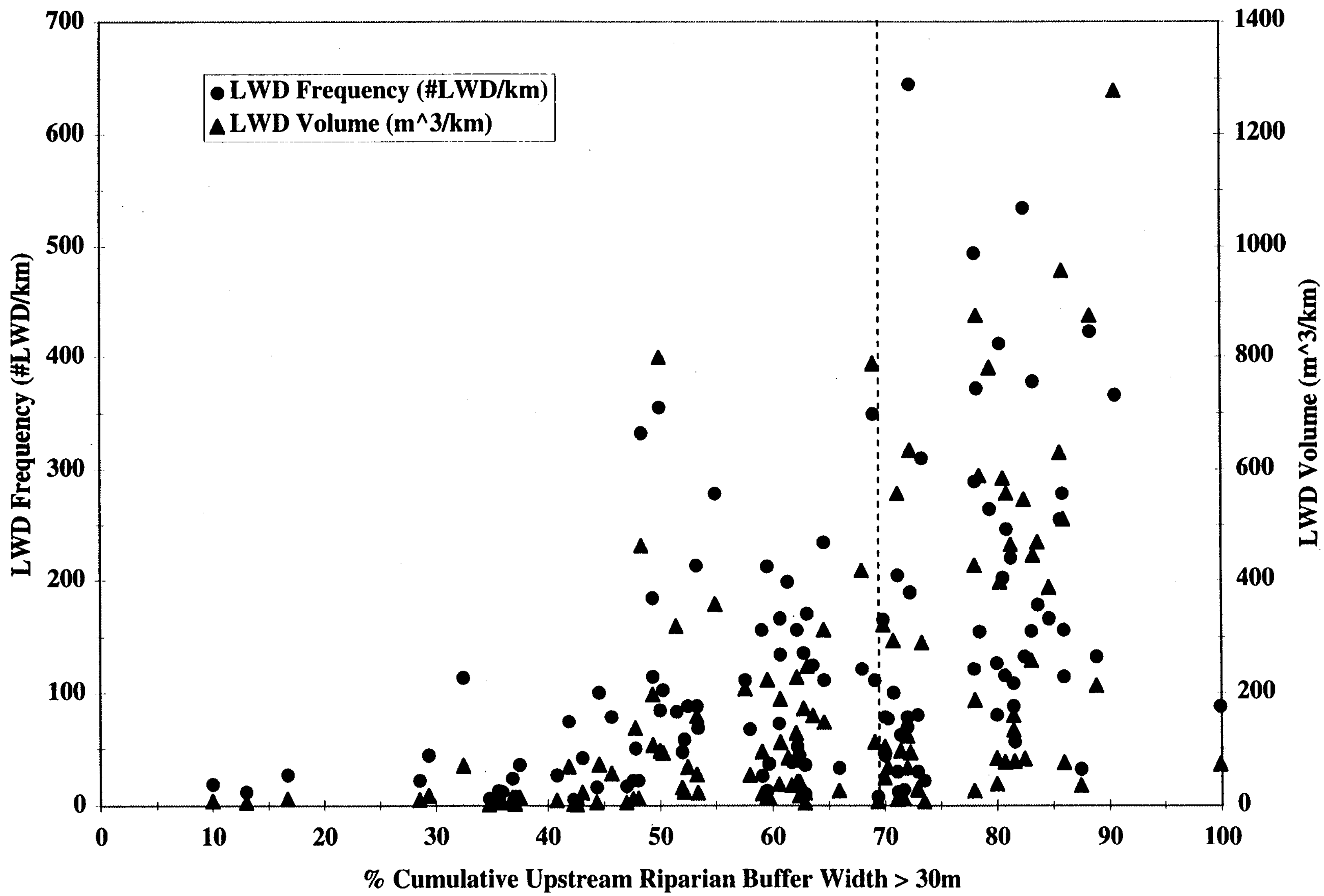

Figure 40: Relationship between riparian integrity (buffer width > 30 m) and large woody debris (LWD) in Puget Sound lowland streams. Vertical dashed-line indicates possible threshold of adverse effects.

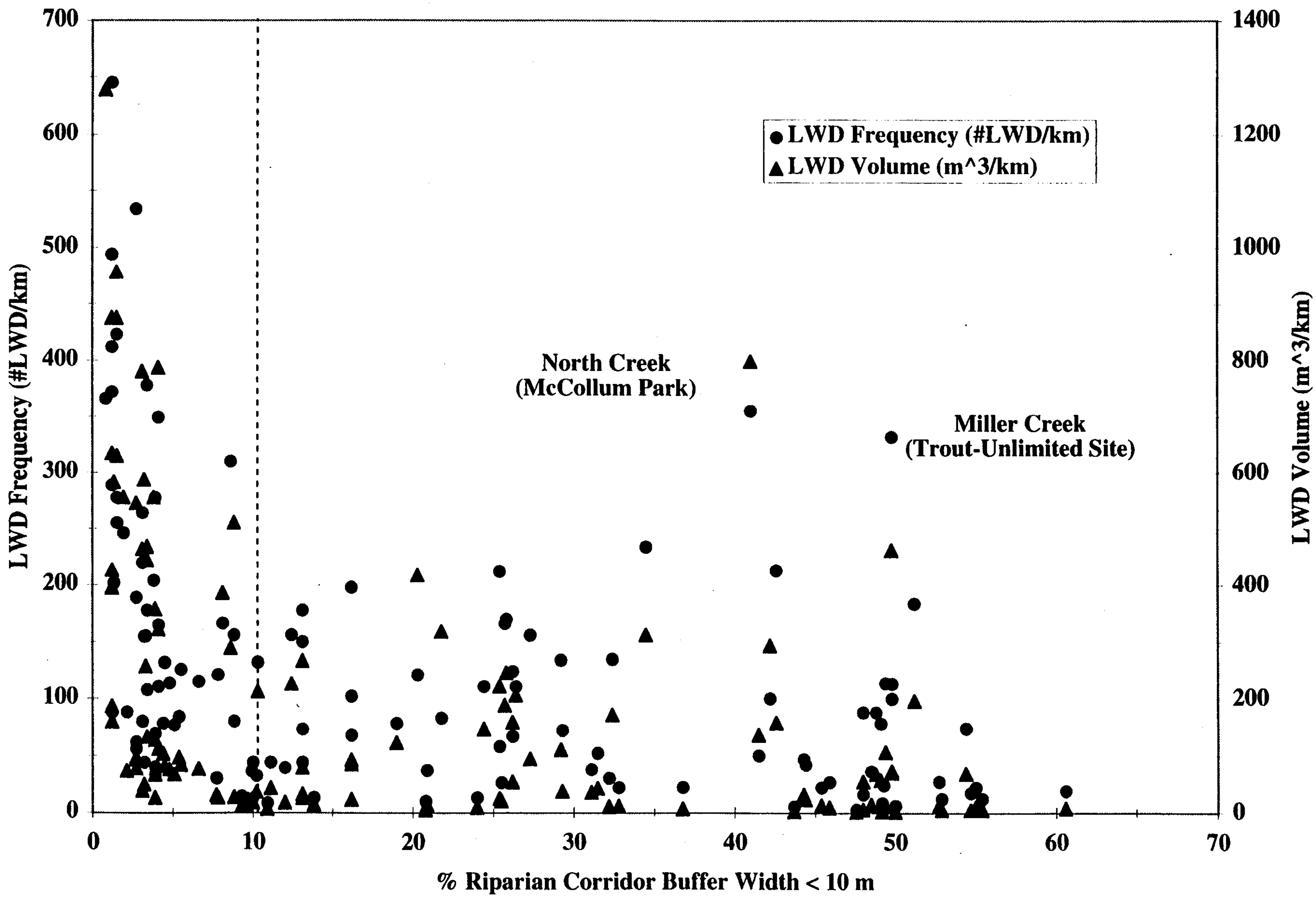

Figure 41: Relationship between riparian encroachment (buffer width < 10 m) and large woody debris (LWD) in Puget Sound lowland streams. Vertical dashed-line indicates possible threshold of adverse effects.

absent from streams unless the riparian corridor is maintained in a predominantly natural condition.

LWD frequency was normalized for stream size based on BFW, similar to pools. A target of two pieces of LWD/BFW of stream length has been proposed as optimal for forested streams in the PNW (Peterson et al, 1992). Recognizing that debris size is as important as frequency, an target for optimal conditions was also proposed for mean LWD volume/BFW. For typical PSL streams (5-10 m BFW), the target for mean LWD size (volume) is approximately 1 m^3 (Peterson et al, 1992). With few exceptions, LWD frequency in these urban streams was well short of that optimum (Figure 42). While several urbanized streams had greater than optimum LWD frequency or volume, no streams in sub-basins with TIA >5% exceeded optima for both characteristics.

These results were similar to those riparian buffer integrity and encroachment. With very few exceptions, only those stream segments with more than 70% of their riparian buffers >30 m wide and under 10% of their buffers <10 m wide, consistently met or exceeded the optimum levels for LWD frequency and volume. Exceptions to this general trend were typically those segments with a few exceptionally large LWD pieces.

As an alternative to the time-consuming task of measuring LWD volume, the number of LWD pieces > 0.5 m in diameter were recorded for each reach. There were significantly less large-sized LWD pieces in segments draining more urbanized sub-basins. Location of LWD within the stream channel was also analyzed. Initially, higher flows were expected to mobilize and transport LWD out of the active (BFW) channel. However, that was not consistently observed; LWD position was unrelated to sub-basin development. However, there were significantly more debris jams in streams with a sufficient source of LWD (intact riparian corridor), such as Miller, DesMoines, Kelsey, and Coal Creeks. The lower than expected fraction of LWD located outside the BFW channel may have been due to human intervention.

LWD quality also decreased in relation to basin development. A qualitative scoring based on size distribution, species composition, extent of decay, channel location, and recruitment potential, was related to sub-basin development (%TIA) as well as riparian

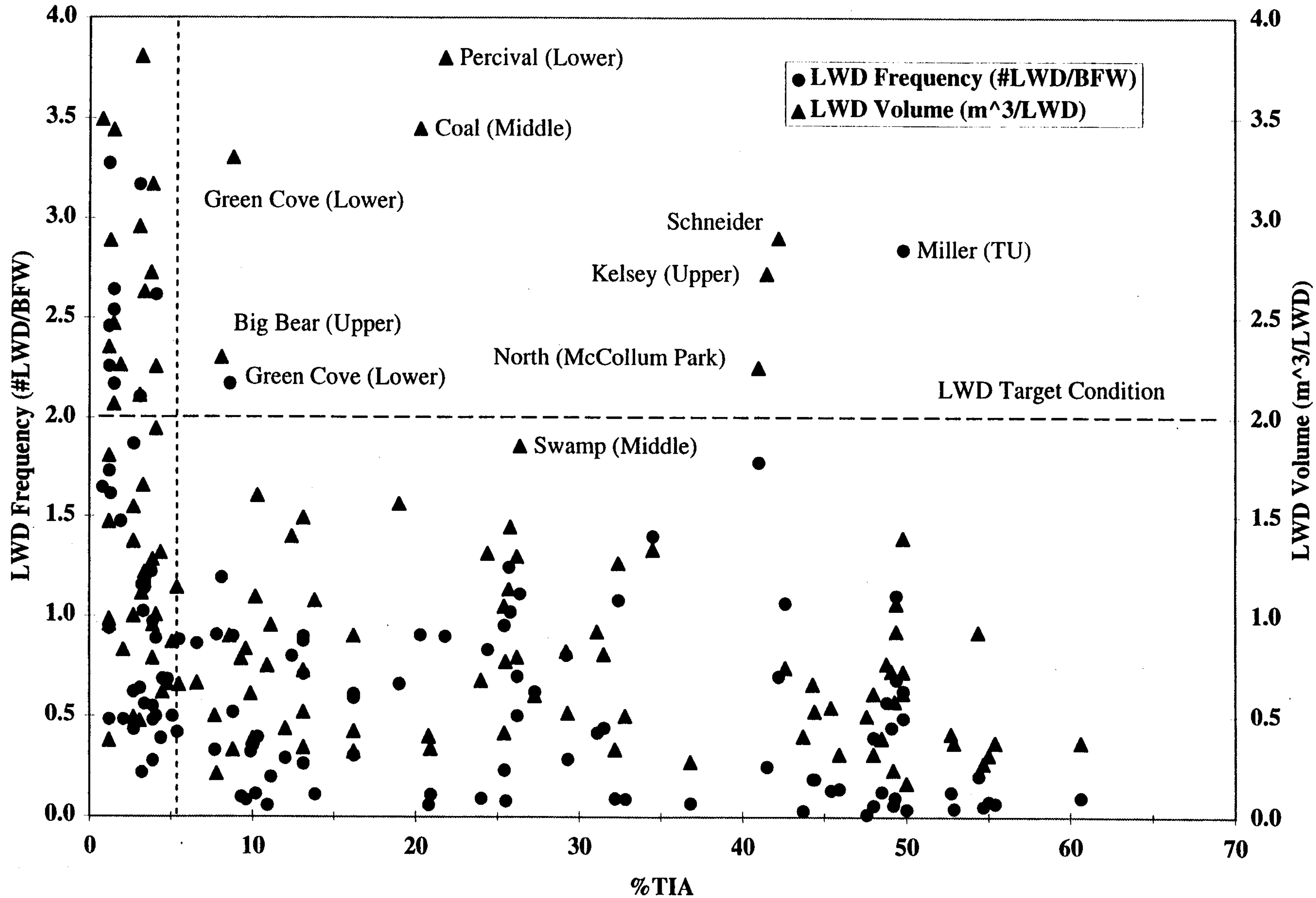

Figure 42: Large woody debris (LWD) in Puget Sound lowland streams compared to the proposed target value based on forested streams . Vertical dashed-line indicates possible threshold of adverse effects.

buffer conditions. An interesting aspects of LWD quality was that species composition was closely linked to LWD size. That was also shown for streams in forested basins. Only large coniferous LWD has longevity in PNW streams (Bilby and Ward, 1991). Most of the larger pieces of LWD found in urban streams were coniferous (predominantly cedar) and were typically in an intermediate or advanced state of decay. This suggests that the small amount of large (>0.5 m in diameter) LWD in PSL streams is probably residual remaining from past timber harvesting. Thus, maintaining natural (mature-coniferous) riparian forests as a source of LWD along streams is obviusly important.

Another critical instream function of LWD is to provide cover on pools. The % pool-cover was strongly related to LWD frequency and volume. In addition, the quality of cover on rearing habitat was directly related to LWD for forming and maintaining pools.

The relationship between LWD and pools is well-documented for streams in forested areas of the PNW (Bisson et al, 1987; Bilby and Ward, 1989; Peterson et al, 1992). The quantity and quality of salmonid rearing habitat (pools) is generally directly related to the quantity and quality of instream LWD. The results here confirm that r for urban streams in the PSL. Pool frequency (#/km) was positively related to both LWD frequency (#/km) and volume (m^3/km). Likewise, total pool area (m^2/km) increased in proportion to the quantity of LWD present (Figure 43). The final, most conclusive indication of the inseparable link between rearing habitat and LWD is demonstrated by the relationship between LWD BFW-spacing and pool BFW-spacing. The target of 2 pieces of LWD per BFW-spacing corresponds directly to a pool BFW-spacing of 2, which is also the optimum for forested streams in the PNW (Peterson et al, 1992).

Qualitative Habitat Assessment

The qualitative habitat assessment protocol, together with the quantitative habitat surveys, was evaluated for PSL streams. A similar approach has been used elsewhere as a low-cost alternative to complete habitat assessment (Plafkin et al, 1989; Rankin, 1989; Hayslip, 1993; Plotnikoff, 1993). Habitat quality was subjectively assessed in each

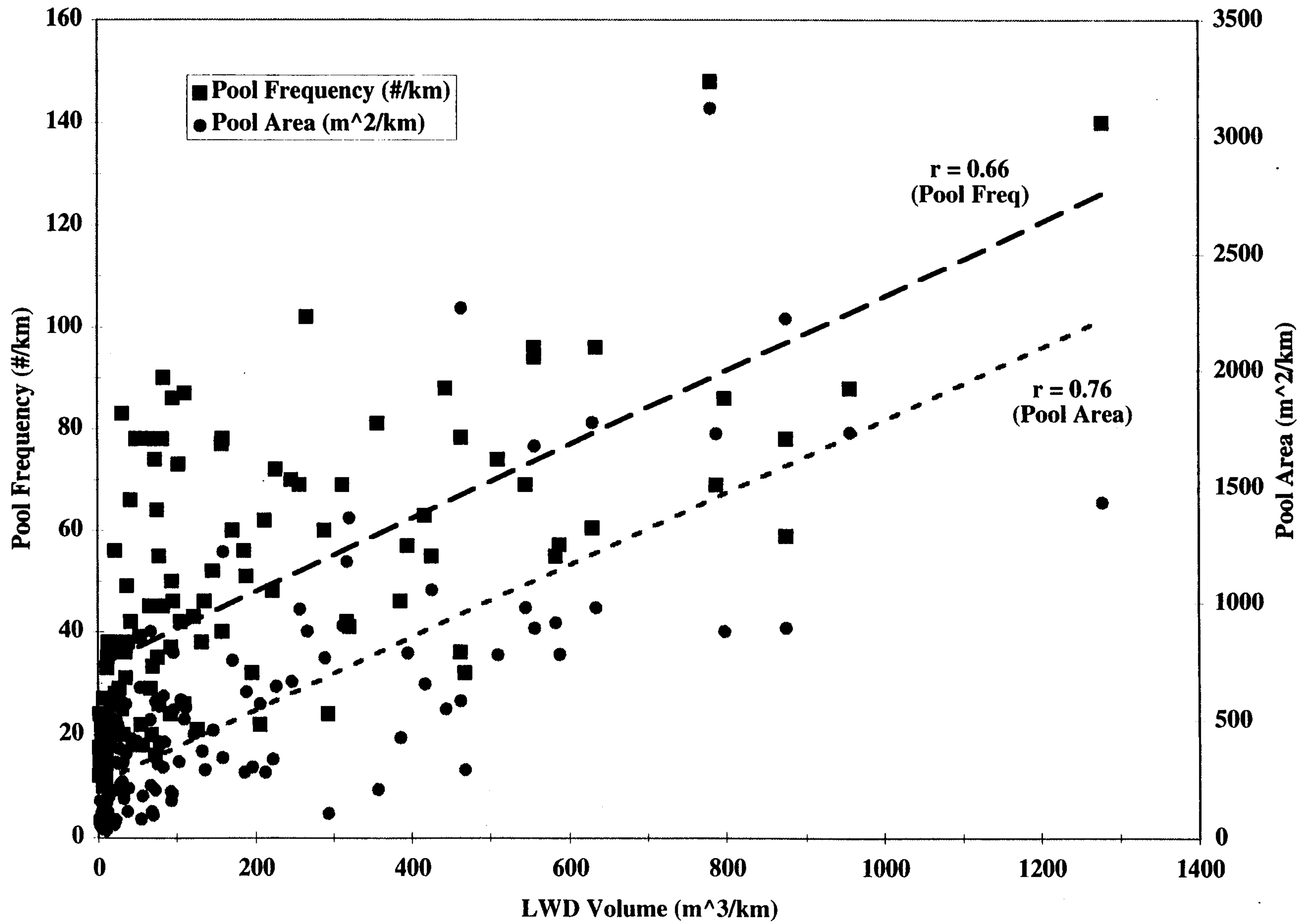

Figure 43: Relationship between large woody debris (LWD) and salmonid rearing habitat (pools) in Puget Sound lowland streams.

stream-segment surveyed and included 15 characteristics, with 60 being the maximum possible score and 15 the minimum (4=optimal, 3=sub-optimal, 2=marginal, and 1=poor). The composite score will be referred to as the Qualitative Habitat Index (QHI).

The QHI was responsive to the cumulative effects of urbanization and was sensitive enough to effectively differentiate between the best and worst stream-segments. However, the variability was quite high for streams in the rural-suburban, mid-range of urbanization (Figure 44). QHI was more strongly related to urbanization assuming a non-linear ($r = -0.79$), rather than a linear fit ($r = -0.62$), indicating that habitat quality may degrade rapidly with the on-set of development, then decrease more gradually thereafter. The QHI scores for all undeveloped, reference streams ranged from 35 to 60. This large range may reflect the degree of "natural" variability in PSL streams or the residual effects of past land-use practices (agriculture and/or timber-harvest). However, there were no QHI scores above 50 for any segments with TIA >5%.

QHI also correlated well with riparian buffer conditions and quantitative measures of instream habitat (Table 21). Again, the rate of decrease was steeper initially as riparian integrity declined, similar to that shown for imperviousness (%TIA). These results showed that qualitative measures (QHI) could be used to indicate quantitative measures of habitat (Table 21).

Biological Indicators

Biological indicators were the final compartment of the conceptual linkage between urbanization, physical habitat, and aquatic biota (see Figure 9). They included periphyton, salmonids, and benthic macroinvertebrates. Benthic macroinvertebrates were emphasized, because data acquisition could be more complete and urbanization effects on those organisms should parallel, in many respects, those on salmonids, which are the resource in these streams of ultimate concern.

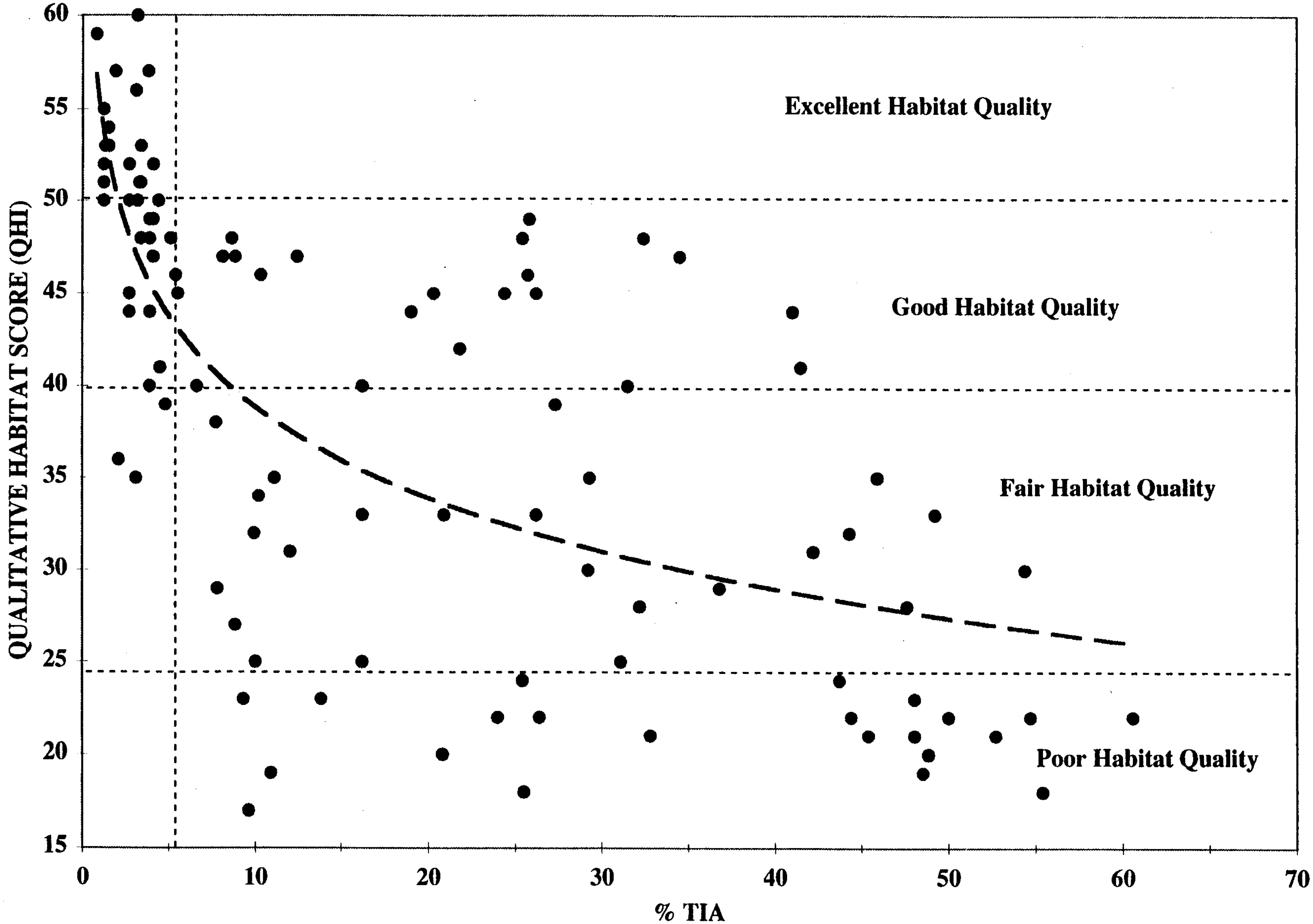

Figure 44: Relationship between sub-basin urbanization (%TIA) and the Qualitative Habitat Index (QHI) for Puget Sound lowland streams (dashed lines indicate possible habitat quality categories).

Table 21: Comparison between qualitative habitat index (QHI) scores and selected quantitative measures of habitat assessment and riparian integrity (stream-segment scale) for Puget Sound lowland streams.

Quantitative Measure	**Linear Correlation Coefficient (r)***
Pool Frequency (#/km)	0.79
Pool Area (m^2/km)	0.77
Pool Cover (Mean %)	0.65
% Pool Habitat	0.74
% Riffle Habitat	0.47
% Fines (1994 and 1996 combined)	-0.82
% Embeddedness	-0.78
Pebble-Count D10	-0.41
IGDO/DO Interchange Ratio	0.78
LWD Frequency (#/km)	0.78
LWD Volume (m^3/km)	0.86
% Riparian Buffer > 30 m	0.71
% Riparian Buffer < 10 m	-0.67
% Riparian Buffer in Natural Condition	0.72
Riparian Breaks per km	-0.73

*Spearman rank correlation; all r-values significant at the 0.05 level

Periphyton

Periphyton was considered a possible sensitive indicator of urbanization, because nutrients are recognized as a major cause for the impairment of lakes, streams and estuaries with nonpoint source runoff (Baker, 1992). Periphytic biomass was determined at each site and compared with light availability (canopy cover) and nutrient content during summer, 1994. Biomass was well below (<50 mg/m^2 chl *a*) levels recognized as causing nuisance conditions (150 mg/m^2; Dodds et al., in press) at all sites (Purcell, 1994). Nutrients, especially SRP, were sufficiently high at most sites to have produced biomass levels well above those observed, but substantial canopy cover restricted light availability sufficiently to have probably limited growth. Biomass levels of periphyton well above the nuisance level were observed earlier in some of these same PSL streams, but at sites without canopy (Welch et al., 1988). This observation also supports the maintenance of adequate riparian buffers with mature forest in order to minimize the effect of nutrient increase with urbanization.

Coho:Cutthroat Ratio

Coho are the only species of salmon that over-winter and spend more than one year rearing in small PSL streams. Therefore, this species and the resident cutthroat trout were chosen as the primary fish of interest. A significant downward trend in numbers of coho spawners has been observed in the Lake Washington stream system over the past 20 years (WDF, 1995). A similar trend has been noted in Hood Canal basin streams (Big Anderson, Stavis, Seabeck, Big Beef, and Little Anderson) over the period of recent urban development (Bahls, 1994). Juvenile coho numbers (smolt production) have also decreased over the same period, indicating that instream rearing survival and productivity have been adversely affected by urbanization (Seiler et al, 1995).

Dominance shifts from coho to cutthroat in PSL streams with increased urbanization (Scott et al, 1986; Booth and Reinelt, 1993; Lucchetti and Fuerstenberg, 1993). However, data are limited to document that trend. The only consistently, reliable data available on juvenile coho and cutthroat abundance comes from surveys done in 1983 (Muto and Shefler) and in 1994 (UW Fisheries unpublished data). Nevertheless, the trend is still obvious, even with this minimal data set. The most current (1994) juvenile coho to cutthroat abundance ratio from 11 of the PSL streams shows a significant difference between undeveloped and urbanized streams (Figure 36). Coho:cutthroat ratios were >5 in undeveloped streams (TIA <5%), but that the natural dominance of coho juveniles over cutthroat (coho:cutthroat <2) was lost at a very low level of urbanization (TIA <5%) (Figure 36). While a specific threshold of urbanization is not entirely clear, these data indicate that suitable habitat conditions also decline rapidly with increased development above a TIA of 5-10%. The relationships between the coho:cutthroat ratio and other measures of urban impact (riparian integrity, road-density, LWD, etc.) support that proposition; i.e., coho and their habitat decline rapidly beyond even a low level of basin development.

QHI was also related with the coho:cutthroat ratio (Figure 37). Only those stream reaches with very high QHI scores (>50), indicating excellent habitat quality, had coho:cutthroratios at high, near natural, levels (>5). These results were similar to those for quantitative habitat measures. In highly urbanized streams (TIA >45%), the coho:cutthroat ratio was consistently <1, indicating a dominance by cutthroat.

Benthic Macroinvertebrates

Macroinvertebrate data for 1994 and 1995 were converted to a multiparameter benthic index of biological integrity (B-IBI), which was used as the primary biological index to judge effects of urbanization in PSL streams (Kleindl, 1995). B-IBI was strongly and inversely related to %TIA (Figure 36), showing a simple linear decline in B-IBI with increase in %TIA. Stream segments with the highest biological integrity are all within largely undeveloped, reference sub-basins. The only reaches with B-IBI >33 were located in sub-basins with a TIA <5%.

As with many of the quantitative measures of physical habitat, variability in B-IBI among the sample sites was high in the low to moderate development range (Figure 45). Biological condition varied most in the middle range of development (TIA + 10-30%) but was also high in the low range (TIA <10%) as well. The high variation in B-IBIs at TIA <10% (from 25-43) deserves further study. Discovery of the subtle differences among watersheds that is not captured in the TIA measure would provide important insight about how to minimize the influence of the early stages of development. B-IBI scores ranged from 18-32 in the 10 to 30% TIA range; they were high in streams, such as Swamp and North creeks, where substantial riparian corridor and wetlands remained and were lowest in Coal Creek, a stream influenced by runoff from an old coal mine. That range of urbanization contained most of the suburban development, which was highly diverse in land-use pattern and distribution. That range of development also contained

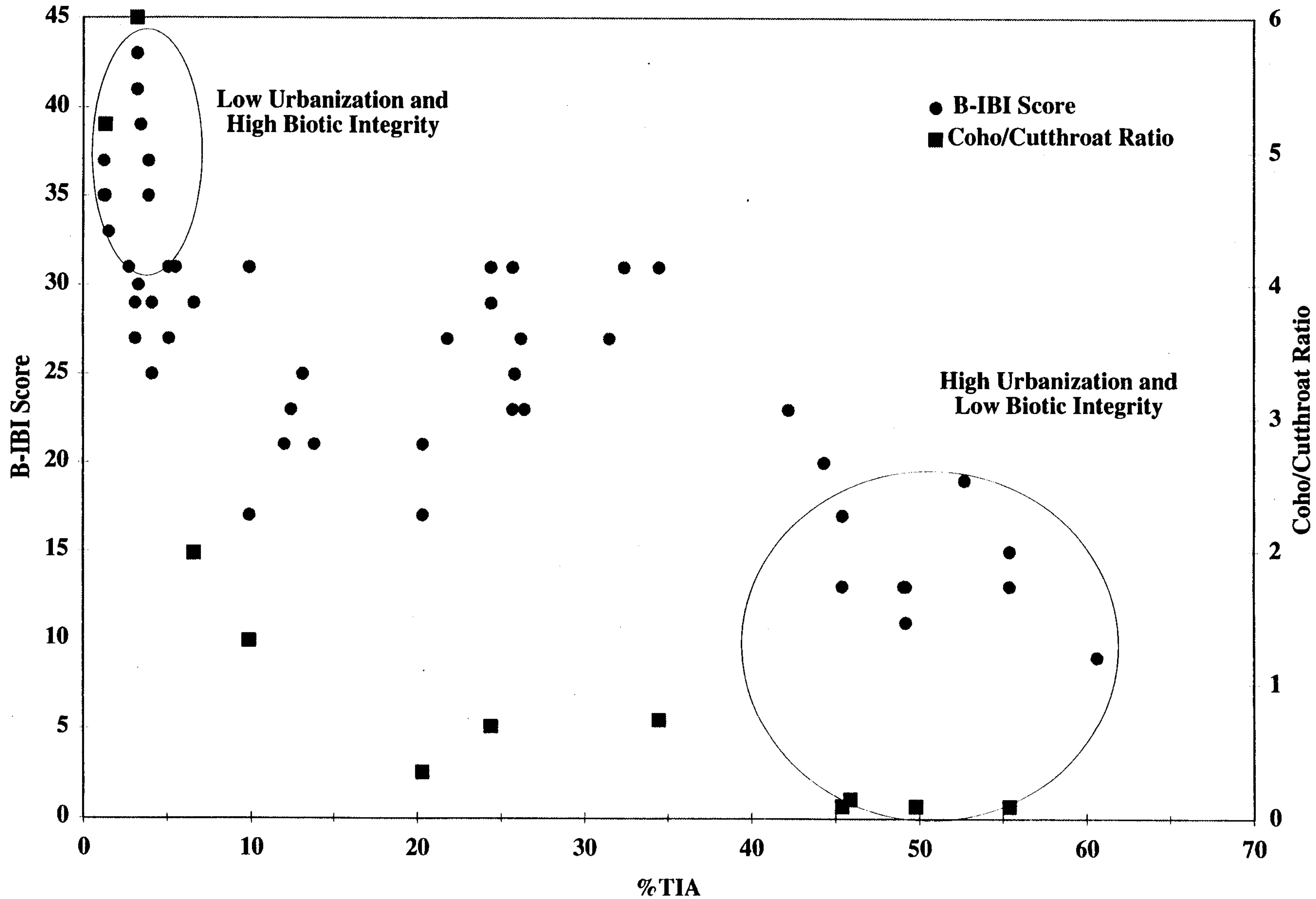

Figure 45: Linkage between watershed urbanization (%TIA) and biological integrity in Puget Sound lowland streams. Indices of biological integrity include the B-IBI and the abundance ratio of juvenile coho to cutthroat.

the most variability in riparian corridor integrity and buffer width and quality. The quantity and quality of riparian buffer may well have had a significant effect on in-stream physical habitat over that range of moderate to high development and, consequently an effect on the benthic macroinvertebrate community. Biological integrity (along with physical habitat) was generally poor at TIA >45%. All stream segments greater than that level, except two, had a B-IBI score <15. In highly urbanized stream basins, the cumulative effects of development appeared to have overwhelmed the resistance and resilience of the stream biota.

The relationships between the other watershed measures of urbanization and B-IBI score are similar (Table 22). Except for the discharge index, these physical measures could all be argued to have a largely indirect effect on macroinvertebrates living in stream substrata. The discharge index, 2-year stormflow to winter baseflow ratio, however, could be expected to directly affect these organisms as physical disturbance of the substratua increased with urbanization (Cooper, 1996). The increased disturbance results from greater flow variability and peak stream power (and basal shear stress) that occur in

Table 22: Correlation of benthic index of biotic integrity (B-IBI) scores with measures of basin development in Puget Sound lowland streams.

Development Measure	Linear Correlation Coefficient (r)*	
	1994	1995
Cumulative Basin %TIA	-0.79	-0.73
Road-Density (km/km^2)	-0.71	-0.65
Urban/Natural DD Ratio	0.74	0.74
2-YR Stormflow/Baseflow Discharge Ratio	-0.53	-0.61
Riparian Corridor Breaks (per km)	-0.81	-0.68
% Riparian Corridor in Natural Condition	0.76	0.59
% Riparian Buffer Width > 30 m	0.77	0.72
% Riparian Buffer Width < 10 m	-0.69	-0.63

*Spearman rank correlation; all r-values significant at the 0.05 level urbanized streams. However, riparian corridor integrity can also be reasoned to benefit macroinvertebrates, so that the relative importance of one versus the other of these variables is difficult to separate considering their relatively similar correlation coefficients against B-IBI.

Instream habitat quality (QHI) was also related to B-IB (Figure 46). The distribution of B-IBI scores against QHI was very similar to that for QHI and the coho:cutthroat ratio, especially for low and high values (Figure 46). B-IBI scores were related similarly with several physical in-stream habitat characteristics (Table 23). Establishing the link between instream habitat quality and biological integrity was an important goal of this study (see Figure 9), as well as showing the relative importance of potentially causitive factors of urbanization. The following in-stream characteristics were expected to be especially important factors affecting macroinvertebrates: % fines, pebble-count D10, IGDO, and substrata embeddedness (Table 23). The fraction of fines in the

Table 23: Correlation of benthic index of biotic integrity (B-IBI) scores with physical and chemical variables in Puget Sound lowland streams.

Instream Variable	Linear Correlation Coefficient (r)*	
	1994	1995
% Fines (1994)	-0.87	-0.79
% Fines (1996)	-0.68	-0.48
% Embeddedness	-0.73	-0.64
Pebble-Count D10	0.70	0.60
IGDO/DO Interchange Ratio	0.62	0.71
BFW/BFD Ratio	0.68	0.63
Pool Frequency (BFW-Spacing)	-0.61	-0.62
LWD Frequency (BFW-Spacing)	0.60	0.50
LWD Volume (m^3/km)	0.57	0.53
Baseflow Conductivity (uS/cm)	-0.57	-0.56
Sediment Zn (ug/L)	-0.61	-0.37
Sediment Pb (ug/L)	-0.61	-0.75
Storm EMC TSS (mg/L)	-0.27	-0.62
Storm EMC TP (ug/L)	-0.73	-0.84
Storm EMC TZn (ug/L)	-0.79	-0.82

*Spearman rank correlation; all r-values significant at the 0.05 level

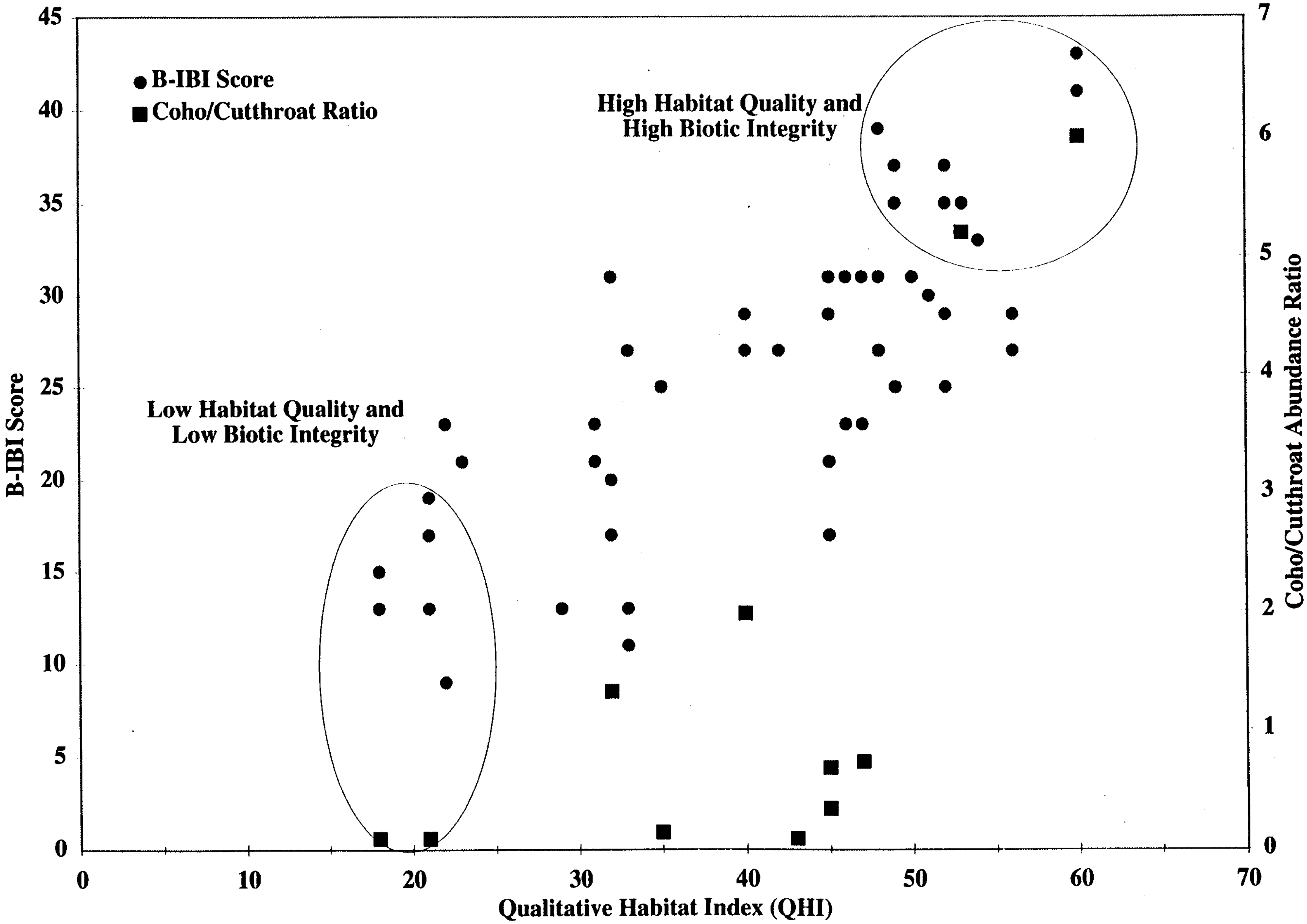

Figure 46: Linkage between habitat quality and biological integrity in Puget Sound lowland streams. Indices of biological integrity include the B-IBI and the abundance ratio of juvenile coho to cutthroat.

substratum has been shown to be especially detrimental to macroinvertebrates, because many of the deep-dwelling fauna characteristic of eroding substrata are adapted to substrata free of fines (Hynes, 1960). Not surprisingly, B-IBI was highly related to % fines, especially with the 1994 data (Table 23; Wydzga). Moreover, % fines were <15 in streams with high B-IBI (>25) and draining basins that had TIA <5%, while fines were >20% in streams draining highly urbanized land (TIA >45%) where B-IBI scores were <20 (Figure 47). Pool and LWD-related would not be expected to directly affect macroinvertebrate, because they were sampled from riffle areas only. However, the fact that these characteristics are also related to urbanization in general results in generally close fits with B-IBI scores as well (Table 23). That is especially evident with several of the chemical water quality variables. As discussed previously, metal concentrations were high enough in streams draining the most urbanized watersheds to have possibly caused even chronic effects to macroinvertebrates, yet B-IBI scores decreased rather quickly as urbanization increased beyond a TIA of only 5%. Thus, the relatively high correlations between B-IBI and Zn definitely do not signify cause and effect (Table 23). Moreover, variables such as TP and conductivity (at levels in these streams) should not adversely affect macroinvertebrates, yet the relationships of these variables with B-IBI scores show fits as close as with variables known to have adverse effects at the levels determined in these streams (e.g., % fines). Thus, attempts to determine which factors were relatively more important than others relied more on comparisons of the magnitude of variables/characteristics relative to urbanization levels and experimental results from elsewhere than on correlation analysis.

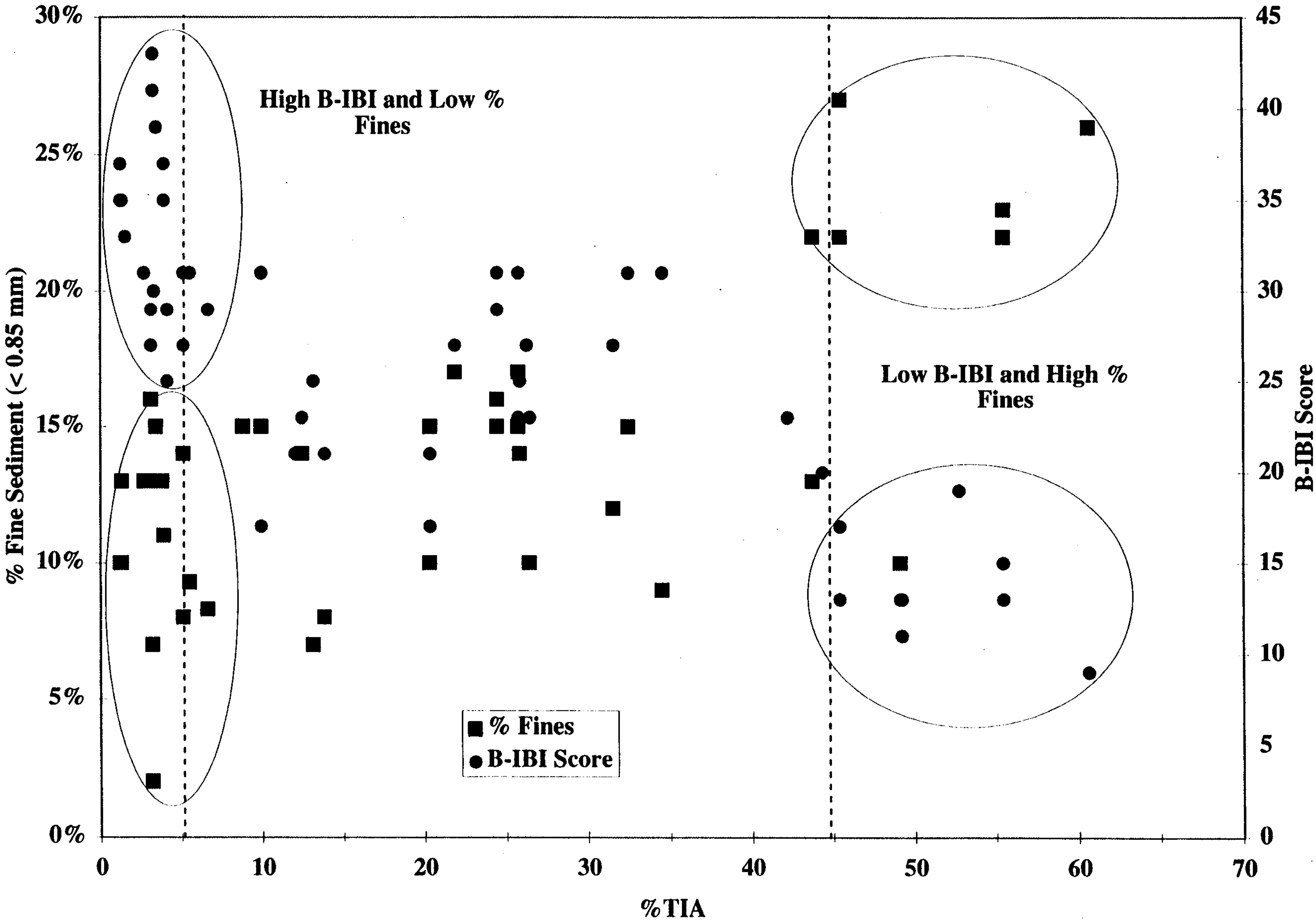

Figure 47: Linkage between streambed fine sediment content and the benthic index of biological integrity (B-IBI) for Puget Sound lowland streams. Circled data points indicate relatively large and small effects of urbanization (%TIA).

DISCUSSION

Measures of Urbanization

Imperviousness

Imperviousness has recently emerged as a unifying quantitative measure of urbanization in an effort to protect watersheds (Schueler, 1994). Impervious cover is one of the few watershed-level measures that can be quantified and managed during each phase of development. Impervious surfaces have characterized urbanizing areas for decades, but only recently have been used to indicate environmental effect. In addition to being quantitative and integrative, imperviousness is relatively simple and cost-effective to determine. Impervious surfaces contribute to hydrologic change and NPS pollution that accompany urbanization, i.e., loss of soil infiltration capacity and enhanced conveyance. Impervious surface is a major component of all urban land-use, whether residential, commercial, or industrial, and has been effectively linked to the degradation of stream quality (Leopold, 1968; Klein, 1979; Griffin, 1980; Arnold et al, 1982; Steward, 1983; Garie and McIntosh, 1986; Steedman, 1988; Booth, 1991; Booth and Reinelt, 1993; Lucchetti and Fuerstenberg, 1993; Taylor, 1993; Schueler, 1994).

Imperviousness as %TIA was an excellent basin-scale indicator of urbanization in PSL (stream) watersheds. Although a threshold of adverse effect was not clearly determined, ranges in %TIA were definitely associated with specific changes in stream quality. Most physical, chemical, and biological characteristics of stream quality changed measurably with increasing development in a continuous rather than step-function manner. However, physical and biological quality indices degraded rapidly from undeveloped, reference stream segments (<5%TIA) to more urbanized levels. These indices continued to show degradation as urbanization increased, but at a more gradual rate. Altered hydrologic regime and physical habitat were the most plausible explanation for the initial decline. As urbanization approached build-out level (TIA >45%), the decline in most stream quality indices became steeper. That was especially true for some chemical water quality constituents (e.g., Zn), which showed a potential adverse effect on biota only in the most urbanized watersheds. Even then, instream water and sediment content remained below accepted criteria for aquatic life.

Imperviousness was used as the primary measure of watershed and sub-basin development, in part because of its universal acceptance, but mainly because results showed it to be a valid indicator of urbanization effects. Specific ranges of %TIA were used to identify links with habitat and biotic effects (Table 24). These discretized (groups of) values of imperviousness were associated with ranges of effects rather than having specific thresholds.

Imperviousness Alternatives

There are actually two measures of imperviousness; total (%TIA) and effective (%EIA). Effective impervious area is more appropriate for some purposes (hydrologic modelling), because it is the surface that is hydrologically-connected to the stream (Beyerlein, 1996). However, %EIA is more difficult and time-consuming to quantify than %TIA. TIA is usually used, because the two measures are proportional, except for purposes of hydrologic modeling (Alley and Veenhuis, 1983; Taylor, 1993; Beyerlein, 1996). Total imperviousness was used here, but EIA could be calculated using accepted conversions:

$$\mathbf{\%EIA = 0.89 * (\%TIA) - 1.33 \quad (r^2 = 0.99)}$$

Table 24: Development categories and the range of %TIA

Development Category	%TIA Range
Undeveloped/Reference	< 5
Rural	5-10
Low-Density Suburban	10-20
Medium-Density Suburban	20-30
High-Density Suburban	30-45
Urban	> 45

Computed values for impervious surface area may vary, due to the wide range of impervious cover observed for similar zoning categories and the wide variety of development type. TIA associated with medium-density, single-family residential zoning varied from 25-60% depending on street and home dimension and layout (Schueler,1994). Estimates can also vary with the method of computation. For estimates used here, land-use data from different municipalities was found in a variety of formats, including LANDSAT, GIS, aerial photo analysis, and basin development records. The age and accuracy of each of these data types varied considerably (see Table 1). Nevertheless, impervious area was still the most representative measure of urbanization.

Alternatives to imperviousness were evaluated to search for specific cause-effect relationships. The largest element of imperviousness is usually transportation. The City of Olympia (1995) found that transportation-related imperviousness comprised from 63-70% of TIA for residential, commercial, and industrial areas. Roads, parking-lots, driveways, and sidewalks are included in the transportation component. Roads for timber-harvest have also been identified as major contributors to watershed disturbance (Eaglin and Hubert, 1993) and the effect has been quantified (Wemple, 1994). Road-density, as the total length of roads per basin area). was used here, rather than a more complicated imperviousness factor. The standard two-lane road was used as a baseline and highways were treated as multiples of the basic road. Road-density was strongly related with TIA, as well as being easy to measure (only an up-to-date road map required).

Roads serve as important conduits of runoff water and pollutants from impervious areas to streams in urban as well as in logged areas. Road-stream crossings form direct connections between roads and stream channel networks via roadside ditches and/or storm-drains. These intersections are major paths for stormwater and NPS pollutants to streams.

Because roads and parking lots cover substantial areas in the urban landscape, they offer an excellent opportunity to reduce impervious cover. Reducing the width of residential access roads from the required 32 to 20 feet in the City of Olympia (1995) has the potential to reduce total imperviousness by 6% for a typical 100-acre suburban subdivision. Road surface reduction is one of the main goals of "cluster" developments. These residential and commercial developments concentrate building around a central core while preserving open-space and naturally vegetated (forest) areas. Large-lot residential subdivisions may have lower impervious cover per lot , but longer roads and sidewalks result in more imperviousness than in cluster sites. A reduction of about 10-50% in impervious cover is feasible, depending on lot-size, buildings, and roads (Arnold and Gibbons, 1996). Clustering can also be used in commercial areas where the largest reduction in imperviousness can be achieved by reducing parking area size. Parking lots in the PNW are typically over-sized by as much as 50% (City of Olympia, 1995). Over-building of parking space is a trend common to retailers and wholesale or warehouse stores and is designed to meet peak demand during weekends and holidays (Arnold and Gibbons, 1996). Techniques for reducing impervious cover associated with parking include multi-level parking garages, porous pavement, bio-infiltration swales, and other BMPs.

Perviousness

There is usually more emphasis on impervious than pervious surfaces in urban watersheds, with the latter being a significant portion of low to moderately developed catchments. Although pervious areas are generally vegetated (lawns, parks, golf courses, etc.), they are not environmentally benign. Areas with disturbed or removed natural forest or wetlands will contribute some imperviousness to the basin. This is especially true of the PNW where precipitation storage capacity depends so much on interception by the forest canopy and soil ("duff" layer) infiltration. Disturbance or loss of the native soil structure is probably the most distinguishing difference between natural and urban pervious areas. Urban soils are typically compacted and have low permeability (Schueler, 1995). Pervious areas are also typically graded with natural depressions filled to encourage runoff toward impervious areas or stormwater conveyance systems. PSL urban "green" areas can have nearly ten-times the runoff from a typical storm event than a naturally forested area of similar size and slope (Wigmosta et al, 1994). Pervious urban areas are very diverse in vegetative cover and management intensity. Private lawns can comprise over 50% of the pervious area in urban watersheds, while "public" turf (parks, golf courses, ball-fields, etc.) areas represented about 25% on average (Schueler,1995). The remainder was composed of landscaped areas and urban forests. This is probably representative of the PSL, although the data here indicated that urban forests were generally more abundant in rural and suburban watersheds than in other parts of the country.

The type of pervious cover is determined to a large extent by adjacent impervious surfaces. Lawns and landscaped "borders" were typically the only pervious areas associated with urban land-use categories (TIA > 45%) found in commercial, industrial, and high-density residential areas of the PSL. Schueler (1995) described the urban landscape as a complex mosaic of pervious and impervious areas, interacting hydrologically. Depending on the design (slope and structure), pervious areas can act as additional sources of runoff or can serve to attenuate and infiltrate runoff. Current BMP encourage the use of vegetated swales and filter strips to infiltrate stormwater and reduce NPS pollution (Horner et al, 1994). Using pervious areas to "disconnect" impervious surfaces from the drainage network has the potential of reducing effective imperviousness by 20-50% (Schueler, 1995). This would imply that pervious areas must be designed to behave more like natural areas, with adequate infiltration and storage capacity. For in-depth discussions of current stormwater management techniques see Horner et al. (1994) and Konrad et al. (1995).

Naturally forested and wetland areas within a watershed are important from both an ecological and hydrological perspective. Their importance is greatest in riparian areas, but applies to surrounding areas of the basin as well (Steedman, 1988). The fraction of watershed in the PSL as natural forest or wetland varied from about 5-85%. Not surprisingly, the fraction of forested area decreased proportionately with the increase in urbanization. Impervious surface area exceeded natural coverage in a transition zone between a TIA of 20 and 40% (Figure 48). There is also a transition in basin hydrologic regime and certain some stream quality characteristics in this zone where watershed forested area and TIA are both about 30%. Above this point, the urban/natural DD and 2-year stormflow/winter baseflow ratios were generally >1.50 (50% increase in DD) and >20, respectively. Taylor (1993) observed a similar transition in wetland water-level fluctuation at this level of basin development in the PSL. Natural vegetated area is obviously an important element of watershed and stream integrity, especially in the riparian corridor.

Riparian Corridor Function

Riparian forests and wetlands form the interface between terrestrial and aquatic ecosystems and are important to maintaining the natural functioning of stream ecosystems. The natural riparian corridor is a contiguous mosaic of stream channels, wetlands, and floodplains located within the valley floor (Naiman, 1992). The riparian zone is composed of the active stream channel, floodplain area, and the hillslope forest. Typically, floodplains in the PNW can be forested and/or wetland-dominated. The dimensions of the riparian zone vary based on these spatial, geomorphic characteristics, as well as temporal ecosystem dynamics (Gregory et al, 1991). The functions of riparian forests and wetlands include, regulation of water temperature through shading, wildlife habitat, LWD recruitment, supply of particulate organic matter, streambank stability, sediment control, and filtering NPS pollutants.

Riparian Buffers

Riparian zones are increasingly encroached upon as development expands. The riparian corridor is fragmented due to road-crossings, penetration by stormwater outfalls, and by human land-use activities. Riparian zones may be protected as sensitive areas by buffers along streams and wetlands. While the need for buffer to protect such sensitive areas is rarely contested, there is much disagreement on the size and type of buffer needed to achieve the desired level of protection. Undersized buffers will probably not

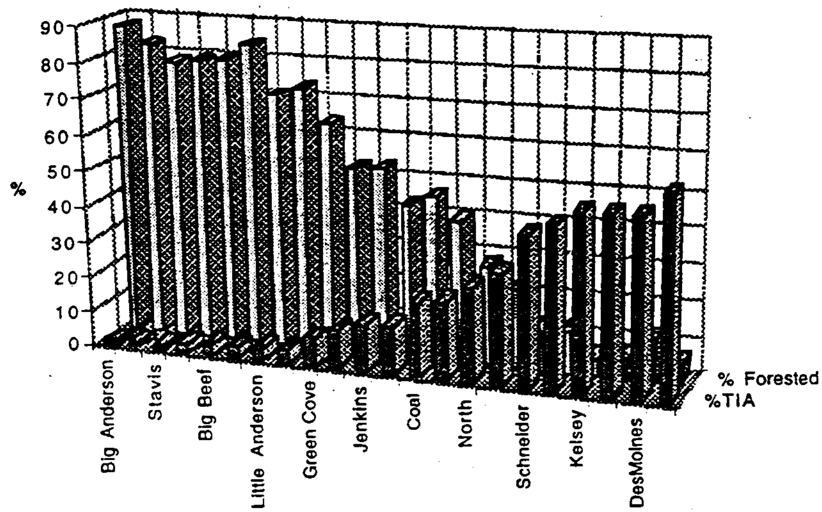

Figure 48 : **Relationship between watershed urbanization (%TIA) and natural forest area in Puget Sound lowland streams.**

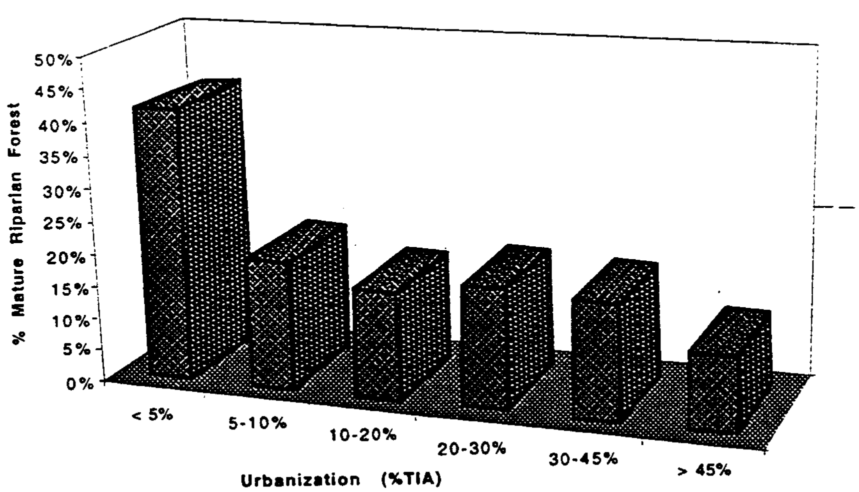

Figure 49: **Mature riparian forest area in relation to watershed development category (%TIA) in Puget Sound lowland streams.**

provide long-term protection, while oversized buffers will provide adequate protection, but may lead to conflicts with land-owners on what is absolutely necessary. Therefore, "right-sized" buffers should be established based on scientific data, rather than political acceptability. The criteria used to determine adequate buffer size are (Castelle et al, 1994):

1) ecological value and sensitivity of the resource
2) intensity and pattern of adjacent land-use
3) existing buffer quality and characteristics
4) functional requirements of the buffer
5) upstream buffer connectivity and condition

In general, smaller width may be adequate if the existing buffer is high quality, mature forest, with few penetrations or if the stream or wetland is of low quality and/or the surrounding land-use has a low impact.

As inferred by the above criteria, buffer sizing is difficult and, while fixed-size buffers are easy to regulate, scientific evidence for a "one-size-fits-all" solution is scarce. Most scientific basis for riparian buffer size requirements comes from timber-harvest activities, although they have application to urban streams. Riparian buffer width here is defined from the BFW outward on each side of the stream.

Control of streambank erosion and hillslope sediment production (mass-wasting) is a major function of riparian buffers and a width of about 30-60 m was found to be adequate in that regard(Castelle et al, 1994). NPS pollutant (nutrients, sediment, and metals) removal is another valuable function of buffers in urbanized basins, although range recommended widths vary greatly depending on the constituent of interest, existing land-use, and type of buffer. Suspended sediment is the most difficult NPS pollutant to remove and, hence, those pollutants that readily sorb to sediment particles. Sediment removal efficiencies vary from 50-90% depending on the situation, but 30-50 m of forest buffer is often cited (Johnson and Ryba, 1992). The dissolved fraction of nutrients is most easily removed and N more readily than P. Most investigators have recommended widths of 30-100 m for pollutant removal, although recent results show that 50% removal efficiency was attained in the first 10-30 m of forest buffer (Daniels and Gilliam, 1996).

Buffer-strip width of 30 m or more of mature trees generally provides the same level of shading for temperature control as that of an old-growth stand (Beschta et al, 1987). Recommended buffer widths range from 30 m to protect instream habitat for fish, aquatic insects, and amphibians, 30-50 m for LWD recruitment, and 100-200 m for birds and mammals (Johnson and Ryba, 1992). Riparian buffers appear to be at least partially effective at mitigating some adverse effects of urbanization. Specific examples will be discussed later.

Less than 10 m of buffer is generally considered functionally ineffective (Johnson and Ryba, 1992). Therefore, riparian buffer <10 m was used here as an indication of riparian encroachment and, based on above cited evidence, a minimum buffer width considered effective was 30 m. Therefore, the fraction >30 m was used as the primary indicator of riparian buffer integrity. The cumulative fraction of buffer >30 m was generally inversely proportional to the level of basin development in PSL streams (Figures 29 and 30). Only one stream with a TIA above 5% had greater than 70% of its riparian buffer wider than 30 m, in contrast to all undeveloped reference streams, which had high riparian integrity (>70% buffer >30 m). Also, riparian encroachment (cumulative fraction <10 m) was directly proportional to urbanization (Figures 29 and 30). These two findings indicate that development has indeed adversely affected riparian zones along PSL streams, sensitive area regulations or prescribed buffers notwithstanding. On the other hand, sensitive area ordinances and buffer requirements have only recently been adopted.

Encroachment

Human disturbance of the riparian buffer zone along PSL streams is quite common. Disturbance was, in all cases, greatest in riparian zones adjacent to higher intensity land-uses (i.e. commercial, industrial, and high-density residential). A key problem with buffers in urban watersheds is that boundaries are usually poorly delineated and often are unmarked. Regulators and contractors typically insure that buffers are well delineated during the construction phase, both on plans and in the field. However, boundaries often become invisible to property owners after development is established. The problem is compounded by the majority of stream buffers being in private ownership.

All urbanized (TIA >5%) streams had >10% of their riparian buffer severely reduced (<10 m). Conversion of native vegetation to lawns was the most common form of encroachment, as well as the removal of trees for firewood, prevention of windfall damage or yard aesthetics. These observations agree with those for wetland buffers in the PSL (Cooke, 1991). Other forms of buffer encroachment include; dumping of yard wastes, invasive/exotic plant species, footpaths, and stormwater short-circuiting via drainage ditches or down-spout piping, which was commonly observed in residential areas. Clearly, there is much room for improvement in delineating buffer boundaries, educating property owners to the benefits of buffers, buffer encroachment and monitoring buffer conditions.

Fragmentation

The continuity of the riparian corridor is as functionally important as riparian buffer width. A near-continuous riparian zone is usually the natural state in the PNW (Naiman, 1992). Fragmentation of the riparian corridor results most commonly from road-crossings. Roads and bridges not only cause discontinuities in the riparian corridor, but they also provide an avenues for stormwater runoff and NPS pollutants, as well as invasive or exotic plants. Multiple roadside drainage ditches typically accompany each road-crossing. These ditches, along with the road surface, significantly increase the drainage network causing major changes in the stream's hydrologic regime. The fraction of the forested riparian corridor was strongly related to the stability of stream discharge in forested lowland streams in Ontario, Canada (Barton et al, 1985). The fact that buffers tend to be correlated with the fraction of basin forested may also influence hydrologic variables. This is further justification for preserving forested areas, both in the riparian corridor and in the adjacent landscape.

NPS pollutants, such as nutrients, metals and other organics tend to increase with greater stormwater volume. Suspended sediment decreased with greater riparian corridor continuity in the same Ontario study (Barton et al, 1985). However, no significant relation was observed between storm TSS and any of the measures of riparian integrity in the PSL, although there was an indication that a greater riparian buffer width and continuity had a slight effect on TP ($r = -0.53$), but a greater effect on TZn ($r = -0.77$). These results indicate that, while the potential for riparian filtering capacity is significant, NPS pollutant removal capacity is limited due to inadequate stream buffers in urbanized areas. The main reason for this appears to be the concentration of runoff and lack of sheet-flow in urban watersheds.

Stream crossings can also result in fish migration barriers due to improperly installed culverts or culverts that become "perched" due to erosive flows. Culverts can create a flow constriction and a potential barrier to fish passage, as well as block the natural downstream movement of organic and inorganic material. This is especially significant in the PNW where LWD is so important to fish habitat. Culverts are usually sized to pass a specified level of stormflow, depending on the age of the culvert and the jurisdiction. However, passage of LWD is usually not included in the design. Buildup of LWD and human-related debris often results in local flooding. Also, gravel and sediment are often prevented from moving downstream from source areas causing excessive deposition on the upstream side of the culvert and scour on the downstream side, providing the only deep-pool habitat in many urban streams. The debris buildup usually requires frequent maintenance and removal. The problem normally worsens as urbanization increases and the culvert becomes under-sized for the increased runoff. Bridges or arched culverts are preferable, but still result in discontinuities in the riparian corridor.

Not surprisingly, the number of stream-crossings (roads, trails, and utilities) increased in proportion to urbanization (Figure 11b). All but one reference stream (TIA

<5%) averaged less than one riparian break/ km of stream. Of the highly urbanized streams, all but one had greater than two breaks/ km. The trend was also clear at the stream-segment scale with the imperviousness discretized into land-use categories (Table 25). Breaks in the riparian corridor <2/km were consistently maintained in rural land-use only. However, the potential for <2/km exists in suburban development if road patterns and development layouts are redesigned, which is the principal of "cluster" development plans (Arnold and Gibbons, 1996).

Riparian Quality

Riparian quality was determined by the dominant land-cover near the stream channel. Forests (mature or young) and wetlands are part of natural riparian areas, although ideal riparian conditions are rarely encountered along urban streams. Historically, most PSL watersheds were intensively logged, including riparian zones. This, together with disturbance from agriculture and urbanization, explains the lack of mature, conifers along steams in urbanized watersheds that are commonly found along pristine forest-streams elsewhere in the PNW (Naiman, 1992). The importance placed on riparian forests in the functioning of streams in the PNW suggests that the goal for riparian zones in urbanized areas should be mature, native forests (Gregory et al, 1991). Native riparian forests are characterized by a mixture of coniferous and deciduous trees and a variety of understory vegetation. Depending on the soil conditions and topography, the dominant trees are western red cedar, western hemlock, and red alder (Rot, 1995). Repeated cycles of logging have substantially reduced large conifers in most PSL riparian corridors where alders usually comprise a larger fraction of the overstory. This has directly affected stream ecosystems. LWD derived from alders tends to be smaller and less resistant to decomposition than conifers, especially cedar (Grette, 1985). The altered composition of organic particulate material may also affect the invertebrate community.

Table 25: Riparian connectivity related to urbanization in Puget Sound lowland streams (COV = coefficient of variation).

Land-use Category	%TIA Range	Mean Riparian Breaks/km	COV (%)	n
Undeveloped	< 5	0.74	74	37
Rural	5-10	1.53	46	16
LD-Suburban	10-20	2.02	45	13
MD-Suburban	20-30	2.17	58	17
HD-Suburban	30-45	2.37	40	14
Urban	>45	3.52	35	23

In general, riparian zone quality, measured as the fraction of corridor that was natural (forest or wetland), decreased linearly with urbanization (Figure 31). The mature forest component along undeveloped, reference streams, using discretized data, was significantly different from all categories of urban streams (Figure 49). The mean fraction of mature riparian forest for reference streams (>40%) was double that of urban streams (<20%). The fraction as mature forest should have been greater along reference streams, except for residual effects of historic logging. Thus, there is an obvious need to increase the amount of mature forest in PSL riparian corridors. Designating a mixed, mature riparian forest as the vegetative goal for riparian buffers would be a valid long-term goal to mitigate adverse effects of urbanization on PSL streams.

Riparian Wetlands

Streams and wetlands are often closely linked in PSL. Wetlands can serve many important functions for water quality including stormwater storage, sediment filtering, and NPS pollutant removal (Reinelt and Horner, 1995). The location of wetlands is often more important than the actual quantity of wetland area in a watershed (Johnston et al, 1990). This is especially true of riparian wetlands that are an integral part of the stream system.

Historical aerial photographs and topographic maps indicated that headwater wetlands were a common feature along most lowland streams in the PSL. However, only remnants of extensive wetland areas remain in most urban watersheds. Headwater wetlands are typically located on the low gradient till-plains where many PSL streams originate. An example of this is the Snohomish "bench" located in the Everett area north of Seattle. This region once contained an extensive, interconnected network of wetlands at the headwaters of Swamp and North Creeks. Most of these wetlands have been lost to developments, such as, Paine Airfield and Everett Mall, but some still remain, e.g., Kasch Park at the headwaters of Swamp Creek and the wetlands along Penny Creek, a tributary to North Creek. Both Little Bear and Big Bear Creeks, and its tributary Cottage creek, also have extensive and mostly intact headwater wetlands, although development pressure is beginning to impact these areas. The Paradise Valley in the headwaters of Big Bear Creek is an excellent example of an intact headwater wetland. Nevertheless, most urbanized streams have lost nearly all riparian wetlands, of which Thornton Creek is a prime example (Lucchetti and Fuerstenberg, 1993). That stream lost all riparian wetlands and over 60% of first-order stream channels to development between 1893 and 1977. PSL reference streams, such as Rock, Big Anderson, Stavis, and Big Beef Creeks have extensive headwater wetlands remaining. The rural and low-density suburban streams, Big and Little Bear, Green Cove, Covington, and Jenkins Creeks, all have a large fraction of their riparian and headwater wetlands still intact. In the medium-density suburban category, Percival, Swamp, and North Creeks, still retain a good portion of their riparian wetlands although the headwater ones are significantly degraded. More urbanized streams generally have few riparian or headwater wetlands and those that do exist are of low quality. Wetland protection should obviously be a high priority.

Design of Urban Riparian Buffer Zones

Protection of sensitive areas (streams and wetlands) with fixed width riparian buffers has become standard practice in most PSL jurisdictions. However, site-specific conditions are often not considered with fixed-width buffers, so stream/wetlands may not be adequately protected. Buffer size should maintain system function and be based on previously cited criteria. Simple prescriptive management, such as fixed-width riparian buffers, is generally less effective than adaptive management based on the unique local conditions and disturbance regime (Naiman, 1992). An excellent list of benefits of riparian buffer zones was given by Schueler (1995a).

From this study and current literature, the following guidelines are suggested for the design of functionally appropriate urban stream buffers:

1) A minimum buffer width of 30 m over 70% of the total riparian corridor should provide adequate protection of most streams if buffer quality is good (mature forest) and adjacent land-use is low intensity (rural or suburban).

2) No more than 10% of the riparian corridor should functionally lost (i.e., <10 m width). To achieve that, buffer boundaries should be delineated and monitored frequently for human encroachment and land the benefits of buffers and their stewardship should be stressed to land owners.

3) Mature riparian forest should a long-term goal to ensure adequate LWD recruitment. That will probably require active management including tree plantings, LWD enhancement, and long-term monitoring.

4) Buffer sizing should be based on aquatic resource value adjusted for current buffer conditions, adjacent land-use intensity, preserving active floodplain (100-year) and riparian wetlands, and steep slopes. Buffers of up to 100 m should be considered for especially sensitive or valuable areas and will require judgment and flexibility.

5) Buffer sizing should attain a near-continuous riparian corridor. Riparian breaks should average <2/km and stormwater runoff inputs, which negate riparian filtering effect, should be avoided. Quantity and quality of runoff should be controlled with structural BMPs, the location and sizing of which is given by (Schueler, 1995a).

Riparian zone management should focus not only on buffer width, but also on corridor connectivity, maturity of vegetation, and species diversity. Management should be approached on a broad watershed level with due consideration for other factors, such as hydrologic disturbance.

Hydrologic Regime

Urbanization in the PSL region has resulted in significant changes in the hydrologic regime of small watersheds. Streams in developed basins tend to exhibit the classic "flashy" hydrograph response to storm events described by Leopold (1968). Loss of infiltration capacity, due to loss of natural vegetation, increase in impervious surface area, and modification of the natural drainage system, results in higher peak (storm) flows, longer duration of peak flows and a greater runoff volume per precipitation event. The hydrologic regime in PSL streams has shifted from subsurface to surface-runoff dominated discharge.

Changes in Streamflow

The importance of disturbance and environmental variability as major factors structuring stream ecosystems is well-documented (Ward and Stanford, 1983; Pickett and White, 1985; Resh et al, 1988; Reice et al, 1990; Yount and Niemi, 1990; Detenbeck et al, 1992). Disturbance variability is often considered the primary regulator of the relative contribution of abiotic and biotic processes (Poff and Ward, 1989). Streamflow is the primary disturbance mechanism in streams and determines many instream physical attributes, such as ambient water velocity, habitat volume, streambed shear stress, and morphological characteristics. Flow fluctuations and extreme events (floods) are, respectively, the primary sources of environmental variability and disturbance, which are so important in determining patterns of community diversity and structure in stream ecosystems(Poff and Ward, 1989). The indicators of those attributes in this study are benthic macroinvertebrates and salmonids. Different combinations of streamflow variation and flooding (frequency and intensity) resulted in varying degrees of control over biotic community structure. Highly variable flow regimes tend to result in abiotic control of instream processes and ecological patterns, whereas more predictable and moderate flow regimes typically produce strong biotic interactions and balanced ecosystem function (Poff and Ward, 1989).

Flood Frequency

Continuous discharge data for selected streams in the PSL study indicated that urbanization increased both flood frequency (disturbance) and streamflow variability. Small flood-events (3X mean daily discharge, MDD) were about twice as frequent in highly urbanized catchments (TIA >45%) as in relatively undeveloped basins (TIA <10%). Medium-sized floods (5XMDD), as well as large flooding events (10XMDD), were more than 3-times as frequent in the highly urbanized streams. Lack of long-term data from basins in the intermediate development range (suburban) prevented analysis of this important land-use category. This agreed with previous findings that showed that urbanization can result in dramatic increases in flood frequency and magnitude (Hollis, 1975). Booth (1991) demonstrated that even relatively low levels of PNW development (TIA >10%) can increase the magnitude of small, previously insignificant flooding events, and increase flood frequency nearly two orders of magnitude from once every 5-10 years to several times each storm season.

Flooding events are nevertheless a natural part of the hydrologic (disturbance) regime in all streams. However, the frequency and magnitude of flooding events are the critical disturbance factors. Floods are widely viewed as natural and necessary events that "reset" stream processes (Resh et al, 1988). Suppression of floods on regulated (dammed) rivers has recently been shown to alter the predator-grazer balance of macroinvertebrates, which adversely affected steelhead trout (Wootton et al, 1996). However, severe flooding can adversely affect stream biota due to streambed scour and/or washout (Seegrist and Gard, 1972). Normally, aquatic organisms adapt to the natural disturbance regime and instream refugia allow biota to recover from extreme events (Sedell et al, 1990; Reice et al, 1990; Detenbeck et al, 1992). As flood frequency increases, however, organisms with adaptive behavior and life-history characteristics are favored (Poff and Ward, 1989). Benthic community structure is especially sensitive to increased frequency of hydrologic disturbance (Robinson and Minshall, 1986). Macroinvertebrates typically seek refuge in the substratum, migrate to off-channel areas, or "drift" in search of more hospitable conditions. This behavior may become ineffective as urban flood frequency and magnitude increase.

Fish community structure may also change as a result of increased flooding. Loss of instream structure (LWD), off-channel areas, and habitat complexity, typically accompany urbanization and reduce refuge space for fish during flooding events (Pearsons et al, 1992). Resistance and resilience (recovery ability) to disturbance, and life-history (spawning and rearing) characteristics, determine how each population reacts to the alteration in hydrologic regime. Coho are especially sensitive to the simultaneous loss of habitat complexity (McMahon and Hartman, 1989) and changes in hydrologic regime. Coho spawn in the fall, making their eggs and embryos vulnerable to scour during winter storms. Cutthroat spawn in the spring and emerge later, possibly allowing them to avoid damaging winter spates. Coho overwinter as juveniles, which are exposed to development-induced high flows. Juvenile coho have a body form designed for transient burst swimming and rapid maneuvering within pools formed by LWD while juvenile steelhead have a cylindrical shape that adapts them to faster water found in the riffle habitat where they dominate (Bisson et al, 1988). Juvenile cutthroat, on the other hand, are apparently not specifically adapted to either slow or fast water. This intermediate morphology may give cutthroat a competitive advantage in glide-dominated urban streams. Cutthroat are recognized as stream "habitat-generalists", which may give then an advantage over coho in urbanized streams. The lower coho:cutthroat ratios common to urbanized PSL streams support the hypothesis that cutthroat are more adapted to the urban streams than are coho (Wchetti and Firstenberg, 1981).

Streamflow Variability

Analysis of long-term records also indicated that streamflow variability (maximum:mean or maximum:minimum MDD) was significantly greater in urbanized streams (see Table 7). Modeling of the nominal 2-year stormflow by Cooper (1996) also confirmed this phenomenon. The calculated discharge ratio of modeled 2-year stormflow to winter (storm-season) baseflow was strongly related to watershed urbanization.

Analysis using urbanization categories indicates that a significant change in streamflow variability may occur at a ratio of about 20 (Figure 12). This corresponds to the level of urbanization where the watershed shifts from predominantly rural to suburban land-use (TIA >20%). At that point, the natural level of fluctuation apparently shifts to one characterized by frequently changing and extreme flow events. Taylor (1993) observed a similar threshold for wetland water-level fluctuation (WLF).

Significant increases in peak flows even occur at lower levels of TIA (5-10%). Hollis (1975) observed that peak discharge increased by a factor of 5-10 for storms with a recurrence interval of one year or less over this same range of TIA. Booth (1990) used modeling and streamflow records to show a significant increase in the occurrence of small storms due to small increases in basin urbanization. These storms did not register as significant flow events prior to development. Taylor (1993) also observed a possible wetland WLF threshold around 3.5% TIA. The ability of instream biota to recover is severely compromised with increasing disturbance frequency (Reice et al, 1990).

Drainage Density

Urbanization adds numerous artificial channels, which are part of the reason for the change in hydrologic regime in natural streams. The most common of these are road-crossings (along with roadside drainage ditches) and stormwater outfalls. These new channels are the main conduits by which stormwater is routed to the stream. These artificial channels are typically more direct than natural ephemeral channels or swales they replaced and significantly reduce the time between precipitation and streamflow response. Infiltration and storage is greatly reduced with artificial stormwater routing systems, so the volume of runoff is dramatically increased. Many of the natural swales and ephemeral channels (and even many first-order channels) are simultaneously lost to grading and construction, further reducing the natural DD.

Below a TIA of 5%, the DD ratio was generally <1.25 (a DD ratio of unity represents a completely natural drainage network; see Figure 11b). At this low TIA, road-crossings were limited to less than one/km of stream channel in nearly all cases, but a "piped" stormwater drainage system to handle runoff was still necessary. Roadside ditches are still the norm in rural areas. As urbanization increases (TIA >10%), roads increase, as do stormwater networks, with a "need" for more "efficient" routing of stormwater runoff. The 2-year stormflow:baseflow ratio exceeded 20 at a DD ratio of approximately 1.50, which corresponded to a TIA in the 20-30% range, which is suburban.

The developed drainage network, therefore, has an equally significant impact on basin hydrologic regime and streamflow variability as does imperviousness. This was discussed first by Graf (1977) in the mid-1970s. Road-related drainage-density changes with timber-harvest activities have also caused significant alterations in hydrologic and erosional processes (Montgomery and Dietrich, 1989; Eaglin and Hubert, 1993; Montgomery, 1994; Wemple, 1994). Wemple (1994) found that typical logging-road systems could increase DD by 36-60%. DD increased by 50% along PSL streams when predominant land-use shifted from rural to suburban, while highly urbanized watersheds had nearly a 100% increase. This represents a major change in hydrologic functioning of a stream regime and is an important underlying cause for many of the observed effects of urbanization.

Notwithstanding these problems, stream channels are still routinely used as conduits for urban runoff in urbanizing watersheds. Thus, land-use planning should include controls on the drainage network as well as impervious surfaces. The number of road-crossings should be limited and direct stormwater connections eliminated, along with efforts to reduce impervious surfaces. Results from the PSL streams suggest that targets should be; 1) road-crossings <2/km (average) of stream channel, 2) no outfalls with untreated stormwater, 3) drainage density within 25% of natural conditions, and 4) streamflow 2-year stormflow:baseflow ratio <20.

Chemical Water Quality

Stormwater runoff is the most important NPS pollution to urban streams and typically contains a wide variety of chemical constituents at concentrations often exceeding water quality criteria/standards (Chandler, 1995). However, constituents rarely exceeded such criteria/standards in the PSL streams although numerous results elsewhere have documented adverse effects of stormwater on stream quality (Griffin et al, 1980; Pitt and Bozeman, 1982; Field and Pitt, 1990; Bannerman et al, 1993; Charbonneau and Kondolf, 1993; Pitt et al, 1995). The lack of a significant effect in PSL streams is probably due to several factors; 1) a sizable fraction is in a particulate for and unavailable for uptake, 2) exposure period is short for EMCs, 3) stormwater enters in winter when temperature and organism activity is low, 4) stormwater is diluted because it enters in winter when normal stream flow is high, 5) variations in constituents due to geographic location, climate, land-use patterns, and treatment (BMPs). An excellent summary of urban stormwater quality was reported by Makepeace et al. (1995).

Chemical water quality in the PSL urban streams generally declined as urbanization increased. Baseflow COND, stormflow EMCs for TSS, TP, and TZn, as well as sediment Pb and Zn, were all strongly related with urbanization. Although fecal coliform bacteria was not determined here, due to the emphasis being on aquatic macroorganisms, which are not affected by coliforms, it an excellent indicator of urbanization (Makepeace et al, 1995). Conductivity was strongly related to urbanization during baseflow, but not during stormflows (Olthoff, 1996; Bryant, 1995; May, 1996). This was probably due to dilution of groundwater, containing highly mobile ions leached from materials added to urbanized areas (e.g., concrete), with rain water. Nitrate-N, which is highly soluble, was also strongly related to urbanization during baseflow probably for the same reason. Conductivity, therefore, was thought to represent a possible surrogate for soluble forms of constituents in urbanized streams.

Nevertheless, large storms, and the resulting high flows in urbanized watersheds, created short-term elevated TSS, turbidity, and nutrient (TP) concentrations. This is probably a result of streambank erosion, localized mass-wasting, and transport of constituents accumulated on impervious surfaces. Construction sites are also potential sources for sediment and sorbed nutrients. The sediment load can adversely affect the stream as flows decrease and sediment is deposited in the stream substrata (Guy and

Jones, 1972). TSS is probably the most damaging stormwater constituent in PSL streams, because of its adverse effect on salmonid reproduction and behavior (Bruton, 1985; Servizi and Martens, 1992) and benthic invertebrate habitat (Lemly, 1982).

Stormflow TSS EMCs were significantly higher in PSL streams during large (>0.75 inches/24 hours) storm events than during small-to-moderate events. The largest TSS concentrations (>250 mg/L) were detected during extremely large rainfall events (>2.50 inches/24 hours) that resulted in major flooding. While the largest TSS concentrations were from urbanized streams (Schneider, Coal, and Percival Creeks), several of the larger TSS levels also occurred in undeveloped (Carey, Stavis, and Seabeck Creeks) streams. Stormflow TSS was only loosely correlated with precipitation magnitudes, durations, or between-storm intervals (antecedent dry periods). Stormflow TSS was also poorly related with measures of riparian integrity and road-density or road-crossings. Therefore, TSS was apparently determined by a combination of factors, such as storm magnitude, basin development, and other more localized events. Nevertheless, instream TSS content was definitely related with stormwater TSS, which is also highly variable (Chandler, 1995).

Stream channel conditions directly upstream from the sample site also affect TSS. Streambank stability ratings were poor for sites with the highest TSS concentrations, due to bank erosion and mass-wasting. This was particularly true for Schneider Creek, where severe streambank erosion and tributary channel incision occurred. High storm TSS in Seabeck and Carey Creeks may have been at least partly due to runoff from recent construction, agricultural and logging operations, while Stavis Creek high TSS was largely linked to an old logging road that has washed out in recent years and continued to affect the stream.

The poor relation between riparian buffer variables and TSS also points to instream sources of sediment for much of those solids. The lack of LWD in urban streams substantially limits instream sediment storage and reduces protection from bank erosion.

TP was the only nutrient that was significantly related with stormflow and urbanization. That is partly because TP is associated with suspended solids (Welch, 1992). Although nutrient concentrations in PSL streams were higher during storm events and increased with urbanization, there was no evidence of nuisance periphyton problems associated with those high nutrient levels (Purcell, 1994). As indicated earlier, biomass levels determined at the established sites did not approach levels defined as nuisance (Welch et al, 1988; Welch et al, 1989; Horner et al, 1990). That was probably due to the shaded conditions at most PSL stream sites (Purcell, 1994). Excessive periphyton, while not common, was observed in open sections of more urbanized streams and nuisance levels were determined earlier in some PSL streams (Welch et al., 1988). Even if riparian canopy is preserved and high TP does not cause nuisance algal biomass levels in urbanized streams, local lakes, such as Sammamish (Perkins etal., 1997), are highly sensitive to the runoff from urbanized watersheds.

Similar to stormflow TSS, TP was highly variable and not associated with any measures of riparian buffer integrity (buffer width, stream-crossings, and buffer quality). However, TP was related closely with precipitation and %TIA. This again indicates that much of the TP is associated with TSS, probably originated from instream sources such as streambank erosion.

Zinc is common in urban runoff and is associated with construction and transportation, e.g., galvanized metals and tires. Similar to P, Zn is associated with TSS, although the soluble fraction may be associated with dissolved solids, which was estimated here by conductivity. Thus, TZn was strongly related with %TIA (Bryant, 1995). However, instream TZn concentrations during storms were often nearly an order of magnitude less than average stormwater concentrations per se, indicating that dilution of stormwater is important. Instream variability in TZn was as great as that found in stormwater runoff (Chandler, 1995), so organisms in PSL streams may have been exposed to higher concentrations than reported here. Nevertheless, metal (Zn, Pb, Cu, Cd and Cr) concentrations in PSL streams were relatively insignificant, even during large storms, unless TIA exceeded 45%, with the only violations of water quality criteria occurring in the most urbanized streams due to Zn (Thornton, Juanita, Schneider, and DesMoines Creeks). Zn is not quite as toxic to aquatic life as Cu, Cd or Cr, but is just as damaging to organism tissues and respiratory activity and can bioaccumulate to some extent (Makepeace et al, 1995). Recent mesocosm experiments show that macroinvertebrates were adversely affected by instream metal concentrations at the very low established chronic criteria (Hoiland and Rabe, 1992; Kiffney and Clements, 1993; Clements, 1994). Adverse effects to macroinvertebrates occurs through loss of the most sensitive species or avoidance by "drift".

The analysis of Zn data showed an interesting contrast with that of TP and TSS. The multiple regression of several variables on stormflow TZn identified riparian buffer width (% upstream corridor >30 m), in addition to basin imperviousness (%TIA) and precipitation, as most significant. However, TSS and TP were significantly related to imperviousness and precipitation only. This suggests that riparian buffers may have been effective at filtering Zn, which probably originated largely from impervious surfaces rather than from the stream channel itself, such as suggested for much of stream TSS and TP being due to streambank erosion. Riparian buffers may also have affected other urban-originating substances that were not measured here. Of course, there may be other explanations for the apparent reduction in TZn with increased riparian buffer width, such as BMPs.

Sediment Zn (and Pb) can also adversely affect benthic organisms (Garie and McIntosh, 1986; Hall and Anderson, 1988; Pitt et al, 1995) and may actually be more detrimental in stream water during high flow, due to the longer exposure. The highest sediment metal concentrations occurred in the most urbanized streams, with the only criteria violations in Thornton Creek, the most urbanized stream. Similar metals in the overlying water, sediment levels did not increase significantly until development levels exceeded 40-50% TIA. Although "chemical quality", as indicated by metals, begins to decline even at low levels of urbanization, it apparently was not a major cause for stream degradation at least until urbanization reached a TIA of >45%.

Channel Morphological Characteristics

Streambed and streambank instability were the most significant problems observed in urbanizing streams in the PSL. Streambank erosion affected a large portion of the channels of urban streams at TIA >5%. Artificial streambank "protection" was also a common feature, especially in highly-urbanized streams (TIA >45%). Urban streambank reinforcement (rip-rap) can adversely affect salmonid habitat and abundance, especially in small, habitat-limited streams (Knudsen and Dilley, 1987). Streambed scour and fill are natural processes in PNW streams, but excessive scour, aggradation, and streambed instability can adversely affect instream biota and their habitat, especially benthic macroinvertebrates and developing salmonid embryos. The increase in fine sediment in the bottom substrata also reduces substratum habitat and biota.

Streambank Erosion

The combination of altered hydrologic regime, loss of instream structure (LWD), and reduction in riparian integrity, resulting from urbanization, increased streambank erosion, as measured by the streambank stability rating. Stable, vegetated streambanks were a dominant feature of natural, reference (TIA <5%) stream-segments. LWD and roots of riparian trees provide bank protection in natural stream channels (Bilby and Likens, 1980). Gradually sloping, vegetated banks and adjacent floodplain areas also provide stability to the active channel margins. In low to moderately urbanized (TIA = 10-20%) sub-basins, stable streambanks were still common; only a few reaches showed localized effects (loss of riparian cover, encroachment of development, residual effects of agriculture, and construction). As sub-basin development increased to the suburban (20-30%) range, streambank stability decreased and bank erosion became more common. That level corresponds closely to the shift in basin hydrologic regime, as indicated by the stormflow:baseflow ratio (see Figure 12). That is logical because available stream power is proportional to discharge. As streamflows continued to increase with urbanization, so did the frequency and magnitude of streambank erosion. The loss of instream structure (LWD) increased erosion potential, which resulted in channelization, which in turn created locally severe erosive flows. Booth and Reinelt (1993) noted a threshold of channel stability at about 10% impervious area in comparing a 10-year flow for a forested basin to a 2-year flow for the same urbanized basin. While their chosen hydrologic variable was different than the one used here, the trend was similar. They also noted a difference in BFW and bank condition between vegetated and "modified" streambanks (Booth and Reinelt, 1993). Obviously, riparian integrity has a significant effect on streambank condition, as indicated by the consistently strong relations between streambank stability and riparian conditions in PSL streams (Table 26).

Scour and Aggradation

Streambed particles are normally mobile in gravel-bed streams, which is the case in the PNW. However, excessive substratum shifting can be detrimental to benthic organisms and incubating salmonid embryos, even in natural streams (Nawa and Frissell, 1993), but can cause an even greater effect in urban streams (Booth, 1990). Moderate scour and aggradation pose little threat to salmonid redds or benthic animals if adequate

Table 26: Comparison of streambank stability rating with imperviousness and riparian integrity indices in Puget Sound lowland stream-segments (n=120).

Quantitative Indices	**Linear Correlation Coefficient (r)***
Sub-Basin Imperviousness (%TIA)	-0.64
Segment Imperviousness (%TIA)	-0.74
Cumulative Riparian Buffer >30 m	0.64
Segment Riparian Buffer >30 m	0.64
Cumulative Riparian Buffer <10 m	-0.59
Segment Riparian Buffer <10 m	-0.60
Cumulative Riparian Breaks per km	-0.53
Segment Riparian Breaks per km	-0.54

* Spearman rank correlation; all r-values significant at the 0.05 level

refuge space exists in the substrata. However, scour events on the order of 10-20 cm in depth can damage or destroy salmonid redds. Aggradation in high sediment areas can smother bottom-dwelling organisms and salmonid eggs via intrusion of fines into intragravel living-spaces.

Scour measurements were not significant related to imperviousness and the magnitude of scour (see Figure 24). Nor was the magnitude of scour or aggradation strongly related with storm magnitude (see Figure 25). However, this is not sufficient evidence to recommend against the use of the scour monitor. The magnitude and frequency of scour is determined by complex interactions among physical factors such as stormflow-induced shear-stress, streambed particle size distribution, and instream structure (LWD). Scour and aggradation are heterogeneous processes that are highly variable. Considerable variability has been observed within stream reaches in response to the same storm event (Hassan, 1990; Quinn and Peterson, 1994). Of course, variability has also been observed within a stream reach due to peak flow events of different magnitude (Duncan and Ward, 1985; Reid et al, 1985; Sidle, 1988, Quinn and Peterson, 1994). Between-stream variability in response to similar storm events was also high, which was similar elsewhere (Tripp and Poulin, 1986; Neller, 1988; Lisle, 1989; Nawa and Frissell, 1993).

Scour and aggradation are largely controlled locally, due to hydraulic conditions that determine local shear stress. Degree and direction of streambed change varies with location within the stream and stream channel gradient and sediment load also affect adjustment of channel morphology. The size distribution of substrata, relative roughness of the streambed, and degree of streambed consolidation also contribute to channel stability (or instability). Thus, shear stress, which is responsible for bedload movement, is highly variable throughout a stream channel, so portions of the streambed move independent of each other as the threshold of incipient motion is reached (Leopold et al, 1964).

Instream structure (LWD) also affects localized scour and fill. LWD can deflect flows and cause different flow patterns within a reach at varying discharge levels. Scour pools caused by LWD were common features in PSL streams. As LWD shifts during peak-flow events, the pattern of local scour and fill also changes. Very large, stable LWD establishes stable patterns of scour and fill within a reach and may promote a relatively stable pool-riffle sequence favorable to aquatic biota (Lisle, 1982; Bilby, 1984; Beschta and Platts, 1986; Nawa et al, 1990). Loss of these large, stable pieces of LWD can result in rapid and pervasive streambed destabilization. Land-use activities that reduce recruitment of or directly remove instream LWD can lead to more severe scour. An increase in coarse sediment load due to upstream streambank erosion, mass-wasting events, or construction site inputs may also change the magnitude and frequency of scour or aggradation.

Greater scour and fill were expected if the above factors combined with increased stormflow due to increased urbanization. Insufficient sample size probably explains the weak relationship observed between urbanization and scour. Only one or two reaches on selected stream segments were monitored due to logistic constraints. In addition, only one or two scour monitors were installed at each transect (reach). In retrospect, that is not enough to adequately represent the process of scour and fill. During each peak-flow event, some salmonid egg-pockets within a given reach are typically scoured away and some are not. Consequently, monitoring at a single location would probably produce highly variable data. Future monitoring should be done at several designated transects (Olson-Rutz and Marlow, 1992). Transects should be located downstream of specific areas of interest such as construction sites, BMPs, or major stream crossings, to assess their effect on stream substratum. Each transect should have multiple scour monitors installed no more than 1 m apart. Scour monitors should be checked and reset at least monthly (or after each major storm event).

Interestingly enough, some culverts produced pool habitat. The few scour-monitors located directly below culverts showed significant scour with little post-storm aggradation. Although there was little or no other cover besides depth, they were often the only deep ("holding") pools found within urbanized streams and frequently contained relatively large cutthroat trout. However, many culverts also were "perched" and presented potential barriers to migrating fish.

Fine Sediment

Pebble-counts are a well-accepted method of characterizing streambed particle size distribution (Wolman, 1954). The original "Wolman pebble-count" was recently modified to examine the effects of sediment from forest management practices (logging) on salmonid spawning habitat (Shirazi and Seim, 1981; Young et al, 1991; Kondolf and Li. 1992). The use of pebble-count techniques to determine the effect of upstream disturbance has considerable promise because particle size distribution is relatively sensitive to changes in sediment load. The particle size cumulative frequency curve can indicate a significant change in streambed fine sediment (Potyondy and Hardy, 1994). Too much fine sediment can degrade instream habitat and adversely affect biota (Chapman, 1988; Meehan, 1991; MacDonald et al, 1991; Ryan, 1991).

The "zig-zag" pebble-count (Bevenger and King, 1995) results indicated a significant shift in fine sediment portion (D10) of the cumulative distribution for urbanized streams compared to reference (undeveloped) streams (see Table 10). A total of 44 reaches were sampled, including 12 reference reaches (TIA <5%). A D10 particle size of 2 mm or less indicated that at least 10% of the representative particles had an intermediate diameter less than 2 mm, which was the minimum that could be hand-measured. Only one reference stream, lower Big Beef Creek in Kitsap County, had a D10 <2 mm and was not used for comparison. That segment is downstream from an impoundment (Lake Symington) and has a history of sediment problems, partly due to streambank erosion and hillslope mass-wasting (Madej, 1978). Of the 24 sample reaches with a basin TIA >20%, 14 had a D10 <2 mm indicating a significant fraction of fine sediment. Fines were even more frequent in the highly urbanized reaches (TIA >45%), where 8 of 12 reaches had a D10 <2 mm. Only the middle portion of DesMoines and upper Kelsey Creek had D10 values >2 mm and those segments were surrounded by intact riparian zones and had mostly stable streambanks. D10 values in the two outwash basins (Covington and Jenkins Creeks) were similar to reference streams. However, both of these creeks also contained reaches with some of the largest D50 (median particle size) values, indicating the dominating effect of their outwash soil structure.

The increased input of fine sediment into urbanizing streams is probably due to; 1) increased particulate runoff due to greater basin impervious surface area, 2) more exposed or disturbed soils (construction), 3) loss of riparian corridor, and 4) streambank erosion. The importance streambank erosion was shown by 15 of the 19 reaches with D10 <2 mm with poor streambank stability ratings of 1 or 2. Streambank erosion is the most likely source of sediment to urbanized streams basins with minimal construction activity or adequate riparian buffer. Erosion and scour are potentially greater with more frequent and higher magnitude flows in urban streams, even with riparian buffers. There may also be more opportunity for flushing fines from bottom substratum, but the net effect will depend mainly on the availability of sediment material and its routing to the channel. Fines accumulation is also partly due to the low gradient of PSL streams (most were <2%), limits effective flushing to very large storms that also produce the most erosion. Lack of structure (LWD) to store sediment may also account for more fines in substratum appropriate for spawning (May, 1996).

Clearly, the pebble-count as a simple, low-cost, and repeatable technique was validated for monitoring extent of fine sediment deposition from urbanization. It is also appropriate to evaluate the effectiveness of BMPs or instream rehabilitation efforts. The "zig-zag" data were closely related with those from McNeil sediment cores (see Figure 21) and the procedure has been used in several forest management related work (Potondy and Hardy, 1994; Bevenger and King, 1995).

Large Woody Debris

The quantity and quality of LWD in PSL urban streams was significantly affected by urbanization (see Figures 39-41). LWD has not been quantified in PSL urban streams, so most comparative data in the PNW are from forested streams. Nevertheless, the importance of LWD in urban streams of the PSL is similar to that in streams draining natural forests in other PNW ecoregions. LWD is the "key" for maintaining spatial heterogeneity or habitat complexity (Maser et al, 1988) and may be the most important component for salmonid habitat.

LWD Frequency and Volume

The decrease in LWD frequency and volume from the cumulative effects of forest management activities is well-documented (Peterson et al, 1992) and a similar trend was shown with urbanization. Deliberate removal of LWD with stream "cleaning" programs has also reduced instream LWD in the PNW (Sedell and Luchessa, 1981; Bryant, 1983; Maser et al, 1988; Meehan, 1991). Historic timber harvest from riparian zones created conditions for early-successional species (Red Alder), which replaced the previously dominant conifers in much of the PNW (Maser et al, 1988). LWD recruitment from alder-dominated riparian forests is usually faster than from coniferous forests, but alder tends to be smaller and less resistant to decomposition, especially compared to cedar (Bilby and Ward, 1991). Second-growth riparian zones tend to become LWD-impoverished (Grette, 1985).

The amount of LWD is highly variable even in streams of unmanaged, natural watersheds. Table 27 shows the LWD frequency for several studies in unmanaged forested watersheds in the PNW. Because LWD quantity varies with stream size, a nominal channel width (BFW) of 5-10 m from the forest data was used for comparison. The mean LWD frequency (#/ km) for small, natural forest streams ranged from about 200-400 and the BFW-spacing (#LWD/BFW) was consistently greater than 2. In contrast, LWD frequency in streams draining harvested watersheds were usually significantly lower (Sedell et al, 1984; Grette, 1985; Murphy and Koski, 1989; Bilby and Ward, 1991).

LWD frequency (pieces of LWD/km) was strongly related with urbanization (Figure 39). Discretizing data into urban development categories (Table 25) highlights the large difference between reference and urbanized streams (Figure 50). With few exceptions, the only stream-segments with LWD frequency comparable to that of

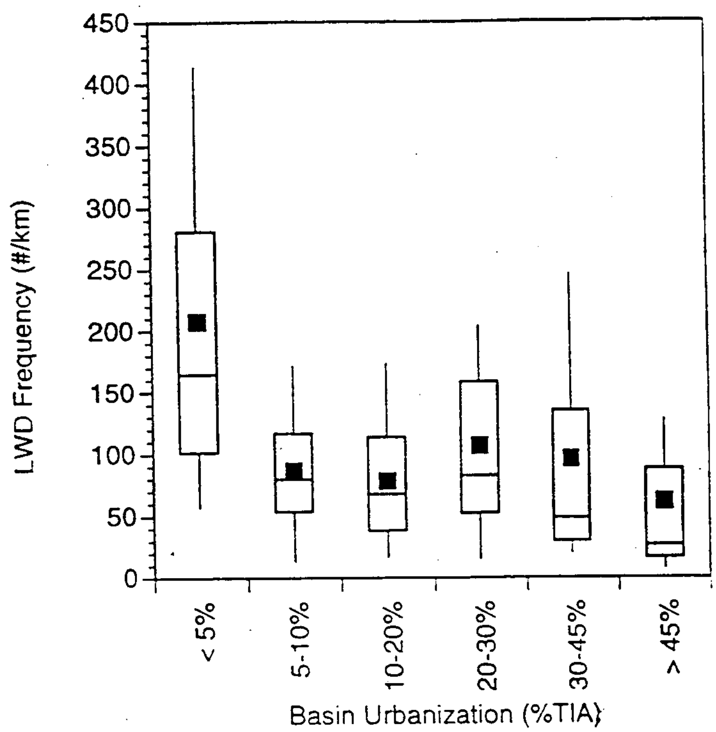

Figure 50. LWD frequency discretized into categories of urbanization.

unmanaged forested streams were those draining undeveloped (TIA <5%) watersheds (Table 27). Two urbanized streams with more-than-expected LWD were sites with extensive "habitat enhancement". The McCollum Park section of North Creek is the site of the Adopt-a-Stream Foundation interpretive trail. Lower Miller Creek is the location of an on-going LWD enhancement project by the local chapter of Trout-Unlimited. LWD was installed (placed and anchored) in both streams. The only urban stream-segments with a greater LWD frequency (#/km) than 300, a typical value for unmanaged streams, were those with >60% of their upstream riparian buffer width >30 m (see Figure 40).

Few of the stream segments surveyed had a LWD-BFW-spacing (# LWD/BFW) similar to that in unmanaged, forested watersheds (Table 27). Only 11 of 120 segments had values greater than 2.0, which was recommended for similar size streams in forested watersheds (Peterson et al, 1992). Of these 11 segments, 9 were in sub-basins with TIA <5% and the other two had relatively wide and mature riparian zones (see Figure 40). These results show the importance of maintaining wide and continuous riparian buffers along streams and wetlands. Riparian buffer width should be based, in part, on specific LWD functional attributes, the condition of the stream, and the surrounding development

Table 27: Large woody debris (LWD) frequency for natural, forested streams in Pacific Northwest (PNW).

Reference Study	# LWD/BFW * Mean	# LWD/km Range (mean)
Bilby and Ward, 1989	2.3	220-550 (390)
Murphy and Koski, 1989	2.4	150-460 (300)
Robison and Beschta, 1990	2.1	250-410 (320)
Ralph et al, 1994	2.9	150-400 (240)
Sedell et al, 1984	N/A	180-450 (210)
Cederholm et al, 1989	2.7	180-520 (400)

* From Peterson et al, 1992; using BFW Range = 5-10 m typical of PSL streams

intensity, as previously indicated. A "one-size-fits-all" rule for buffer width will probably not meet long-term requirements for LWD recruitment.

With few exceptions, LWD volume (m^3/km) was also adversely affected by urbanization (Figure 39). The disparity between undeveloped, reference streams(TIA <5%) and urban streams is further illustrated by discretizing the data (Table 5). LWD size and range of volumes were also much greater in reference streams than in any urban stream (Figure 51). There are two possible explanations for the latter; 1) the natural variability of instream LWD is high and undeveloped PSL streams are within the natural range of forested streams in the PNW, 2) almost no old-growth riparian zones exist in the PSL, even along reference streams, likely due to historical forest management and/or agricultural activity affecting almost all watersheds in the region. Until recent protection of riparian corridors, removal of instream LWD was probably quite common and is reflected by present-day quantities. Both are reasonable, but the latter is most probable. LWD size measurements support this suggestion, although frequency data do not. Results were similar from a recent study of managed and natural forested streams, in which there was no significant difference in LWD frequency between logged and unlogged streams, but there was for LWD size (Ralph et al, 1994).

Even more significant than the decrease in LWD volume in urban streams was the lack of "key" pieces of LWD (Figure 52). "Key" pieces of LWD are >0.5 m in diameter or >1/2 BFW in length (Peterson et al, 1992) and are considered an essential component of natural forested streams in the PNW (Maser et al, 1988). These large structural elements provide long-term hydrologic roughness, habitat diversity, and channel stability. They are especially important for maintaining channel stability through very large storm events (Bilby, 1984). This attribute should be especially important in urban streams with more frequent large flow events. An average of 40% of the LWD was >0.5 m in diameter in PSL reference-stream segments. Recent observations in unmanaged and managed forest streams showed respective values of 60 and 40%, respectively, for key LWD (Ralph et al, 1994). Thus, PSL reference (TIA <5%) streams appear similar to managed forest streams. In contrast, only 20% of the LWD was of key size in urbanized PSL

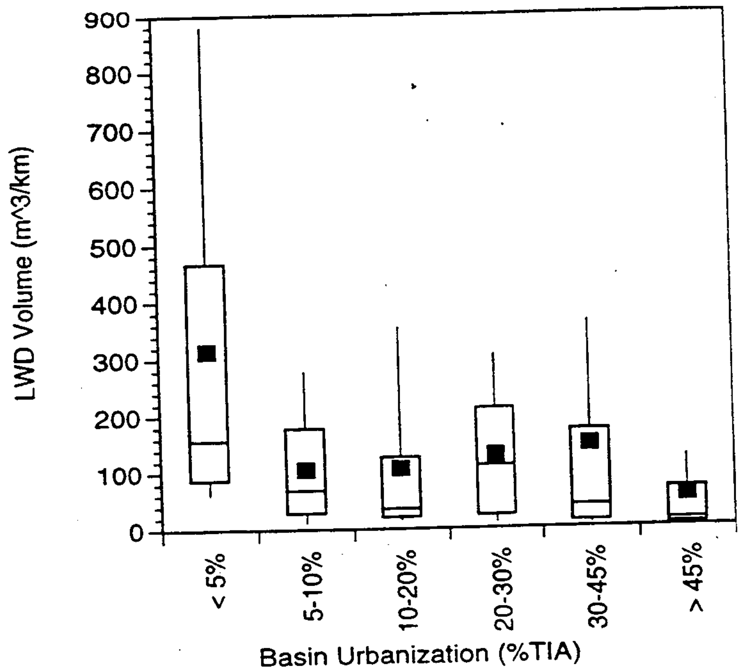

Figure 51. LWD volume discretized into categories of urbanization.

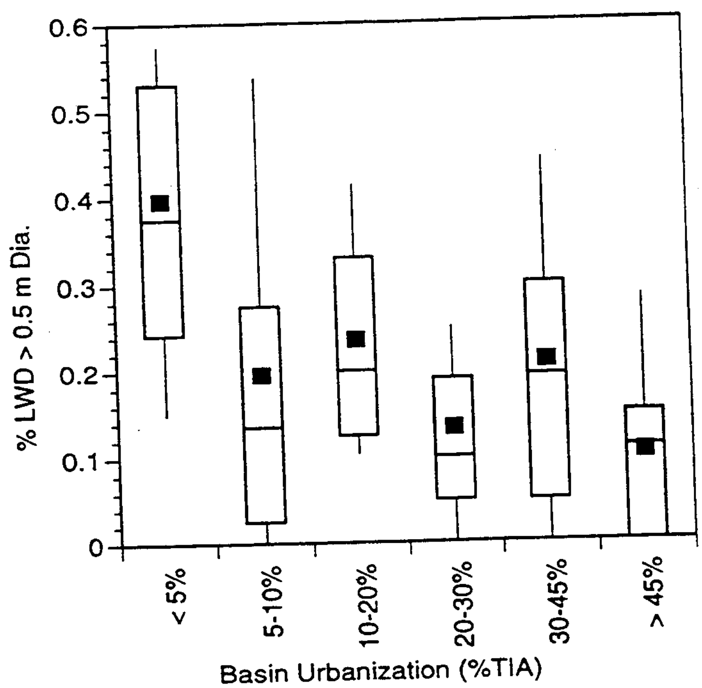

Figure 52. LWD "key" pieces discretized into categories of urbanization.

streams, indicating that activities associated with urbanization remove more "big" LWD than logging.

Riparian integrity includes riparian buffer width (quantity) and several aspects of its quality, e.g., stand-age and species composition, had an even greater effect on LWD. The strong influence of buffer width on LWD volume is shown in Figure 40. LWD was observed at significant frequency and size only in stream-segments with >70% of their riparian buffer >30 m wide. All stream segments with LWD frequency (# LWD/BFW) >2 had at least 70% of their riparian corridor >30m in wide (see Figure 42). Riparian corridor encroachment (Figure 41) had the opposite effect on LWD quantity. Only stream-segments with <10% of their upstream riparian corridor lost to urban encroachment (<10 m wide) had 2 or more pieces of LWD/BFW (Figure 42). This indicates that prevention of LWD removal may be more important than maintaining LWD recruitment potential. Wider, delineated riparian buffers should discourage access and resulting adverse effects such as stream "landscaping". There is extensive evidence that LWD recruitment potential depends heavily on riparian corridor quality and size (Murphy and Koski, 1989; Van Sickle and Gregory, 1990; Johnson and Ryba, 1992; Fetherston et al, 1995). The general consensus of these and other workers is that nearly all LWD is derived from the riparian zone within 30-100 m of the active (BFW) stream channel.

The presence of a wide, natural riparian corridor, however, will not always insure adequate LWD, because there are several other factors that influence the sources and distribution of LWD in urban streams. Maturity (stand-age) of the riparian forest strongly determines LWD quantity and quality. An average of 40% of the riparian corridor along reference streams was composed of mature forest, either predominantly coniferous or mixed (Figure 49). In contrast, less than 20% of urban stream corridors had mature forest. Bilby and Ward (1991) found that LWD frequency was significantly greater in streams with old-growth riparian forests than in those with either second-growth or clear-cut forests. In channels 5-10 m wide, LWD frequency in clear-cut and second-growth sites was 77% and 56%, respectively, of that in old-growth sites. These comparable results indicate that urbanization affects LWD frequency similarly to forest practices (logging and road-building).

Direct removal of LWD from streams is also detrimental by destabilizing remaining LWD, which promotes flushing of wood downstream, decreasing the LWD frequency (Bilby, 1984). While no specific records of LWD removal were found for PSL streams, its prevalence in the PNW (Sedell and Luchessa, 1981) suggests that these streams were similarly affected. The more "clumped" nature of LWD (i.e., concentrated in debris-jams or channel margins) also suggests the occurrence of removal activities and the adverse effect of other LWD barriers.

LWD Loss Mechanisms

The loss and/or removal of LWD, along with the increased flows due to urbanization, can produce a feedback effect causing further losses of LWD. The relative effect of high flows versus decreased LWD are often difficult to distinguish. The lack of LWD in urbanizing streams leads to degradation of instream physical conditions (Booth, 1991). Channels experience greater scour, erosion, and lateral instability, the flux of sediment is greater and more closely tied to individual high-flow storm events, and the habitat diversity of natural streams is replaced by a uniform channel profile and cross-section (glide-dominant). Many of these same effects are products of increased discharge that normally accompanies urbanization, but the lack of LWD to dissipate stream power is substantial (Keller and Swanson, 1979; Bisson et al, 1987; Ralph et al, 1994). The mechanisms by which LWD is lost from urban stream channels are similar to those in streams affected by timber harvest, but are generally more pervasive and continuous.

LWD removal by humans is probably the most widespread and difficult to detect mechanism and is unique to urban streams. There is little human contact with riparian buffer following timber harvests, but near-continuous human intervention is possible in an urban setting, which affects LWD recruitment from the riparian zone as well as inchannel LWD. Concerns over flooding have also driven removal of inchannel LWD. Mobilized LWD is often lodged in culverts or under bridges at road-crossings, necessitating removal. Culverts are especially problematic, because they prevent natural redistribution of LWD downstream. Landowners also remove instream "obstructions" (LWD) to "control" drainage or flooding. Encroachment of development further exacerbates removal of LWD from riparian areas and/or the loss of potential LWD from riparian forests for, among other reasons, "aesthetic improvement". Gregory and Davis (1993) observed that while people preferred "natural" streams, they also preferred channels without "debris". People seem to find streams with natural LWD "messy", despite the ecological advantages of LWD. They also preferred riparian areas with natural but not "wild" vegetation (Mosley, 1989; House and Sangster, 1991; Gregory and Davis, 1993). Wood-cutting for firewood and to "prevent" wind-fall home damage are also reasons for removal.

Washout from higher, more frequent and longer-lasting discharge events is also a major mechanism for LWD loss from urban streams (Hollis, 1975; Barker et al, 1991; Booth, 1991). These high flows can move all but the largest of LWD and expand the channel cross-sectional profile through lateral expansion and/or incision (Booth, 1990). This exposes more of individual LWD pieces to high flows, as well as undermines and destabilizes previously anchored or buried LWD, increasing its chance of movement and/or washout. Bedload stored behind LWD also becomes susceptible to transport, increasing sedimentation and scour of the downstream channel (Bilby and Likens, 1980; Bilby, 1984; Lisle, 1987).

Stranding is a related mechanism of LWD loss in urban streams and is caused by the same high flows as washout. LWD may become beached outside the active stream channel due to the combined effects of channel enlargement and incision. LWD also becomes suspended above the streambed and ineffective at providing flow resistance, primarily because increased sediment transport in urban streams and lack of bedrock gradient control in the PSL results in down-cutting (incision) of the streambed. The further loss of flow resistance by stranded LWD usually accelerates the incision process and may result in catastrophic LWD washout as well. Channel widening can also strand

LWD, resulting in additional loss of flow resistance, instream habitat, and sediment storage in urban streams. Both of these conditions were observed in PSL streams.

LWD losses from streams are not always gradual. Human activity in forest stream riparian zones led to rapid and generally severe changes in stream channel structure (Bilby and Ward, 1991). The rapid decline in instream habitat quality and quantity observed to accompany the onset of urbanization in PSL streams supports the results from forest streams. That over 100 years is required for a mature, natural riparian forest to reestablish after logging (Grette, 1985) is evidence of the need to protect and nurture wide riparian corridors along urban streams. For optimum LWD recruitment, the target condition for riparian buffers should be a mature forest dominated by conifers.

Salmonid Habitat

Historically, some of the most productive freshwater habitat for Pacific salmon were small streams surrounded by mature, native (primarily coniferous) forests. The streams of the PSL region were once a vast network of sloughs, beaver ponds, and complex multi-channel streams with an abundance of LWD (Maser et al, 1988). Urbanization generally leads to more homogenous, simple stream channels and a loss of habitat complexity. Lower stream quality and less habitat complexity, in turn, leads to less species diversity and a long-term reduction in salmonid abundance (Reeves et al, 1993).

Spawning and Incubating Habitat

The process of spawning site selection by female salmonids is not fully understood (Groot and Margolis, 1991), however, there are a combination of factors that determine acceptability for most species. These include substrata particle size distribution, water depth, water velocity, gravel permeability (fine sediment content), streambed topography, and cover. The exact characteristics of the selected spawning site depends on the species, size of the female, and site availability. The quantity (surface area) of spawning habitat in PSL streams was found to be adequate, based on the above indices and measurement procedures, across the gradient of development levels. Particle size distribution was apparently adequate for most species and a variety of sizes of spawners (see D50 values in Table 10). Within a range of acceptable spawning sites, female salmonids apparently exert a second set of criteria in selecting the optimum site for redd construction. One of those criteria is believed to be adequate intragravel flow for incubation and development of embryos (McNeil, 1966; Vaux, 1968; Crisp and Carling, 1989). Suitable intragravel flow conditions occur where the interchange of oxygenated surface water into the streambed is enhanced by a combination of bed topography, instream hydraulic conditions, and gravel permeability. Pool-tailouts are most often identified as the optimum sites for redd construction due to the natural down-welling flow common to these areas.

Land-use activities such as timber-harvest, mining, grazing, and urbanization, as well as catastrophic natural events (i.e. floods or debris-flows), can adversely affect spawning habitat. Besides scouring and sometimes complete removal of spawning materials during high flows, human activity also damages spawning areas with excessive deposition of fine sediment in stream substrata (Everest et al, 1987). Female salmonids have the ability to modify the streambed and improve spawning habitat by winnowing the fines from the redd during digging. However, sediment deposition after redd construction can significantly degrade the environment for egg incubation. Excessive fine sediment has been associated with reduced survival to alevin emergence (STE) due to reduced intragravel water flow (Burton et al, 1990). Reduced survival and size of emerging alevins has been directly related to the supply of DO to the incubating eggs (Shumway et al., 1967). Therefore, intragravel DO (IGDO) was selected as the most biologically significant indicator of spawning and incubating habitat quality.

Recommended criteria for IGDO to protect salmonids varies. Minimum DO of 5-6 mg/L is widely accepted for adequate survival (US EPA, 1986). These are considered minimum limits and carry the assumption that normal variations in DO will include concentrations much greater than the minimum (Welch, 1992). Nevertheless, saturation (at least 9 mg/L) was required for maximum growth from egg to fry and should be the goal in order to sustain maximum productivity, although lower DO concentrations are acceptable for short periods (Doudoroff and Shumway, 1967; Bjornn and Rieser, 1991; Welch, 1992). The classic work by Douderoff and Shumway (1967) showed that some detrimental effects on embryo, alevin, and fry development will probably occur if DO levels drop below 100% saturation, but that the greatest damage (reduced growth, delayed emergence, or lower STE) occurs at DO concentrations < 5 mg/L. Recent work in Idaho showed that STE increased significantly if mean IGDO was above 8 mg/L or 70% saturation (Maret et al, 1993). DO concentration should be specified, rather than % saturation, because DO concentration at saturation decreases as temperature increases, which competes against the increased metabolic demand for DO with increased temperature (Welch, 1992).

Efforts to establish a national or regional standard for IGDO has been hampered by lack of a standardized monitoring technique. The IGDO device used here was modified from that used in Idaho by Maret et al. (1993) and is described in the methods (see Figures 5 and 6). IGDO monitors were located at the sites where % fines (McNeil) and macroinvertebrates were determined. IGDO was usually inversely related with % fines, although the variability of % fine data was quite high (Wydzga, 1996).

IGDO concentrations in PSL streams ranged from <1 mg/L to >10 mg/L, with the lowest concentrations were considered low enough to cause reduced embryo survival or development. A trend toward lower IGDOs with increasing urbanization was indicated from seasonal mean IGDO/DO interchange ratios for reference and urban streams (see Table 20 and Figure 33). The IGDO/DO interchange ratio for undeveloped, reference streams (TIA <5%) was consistently above 80%, while the ratio for highly-urbanized (TIA >45%) streams was typically much less. The trend is clearly illustrated by results from three reference and three urban streams with IGDO/DO values for the 1995-96 incubation period (Figure 53). Although IGDO and DO values varied throughout the monitoring period, the interchange ratios for undeveloped sites were consistently >80% whereas those for highly urbanized streams sites were all <80%. That effect was expected to be largely explained by fine sediment deposition, however, mean IGDO/DO

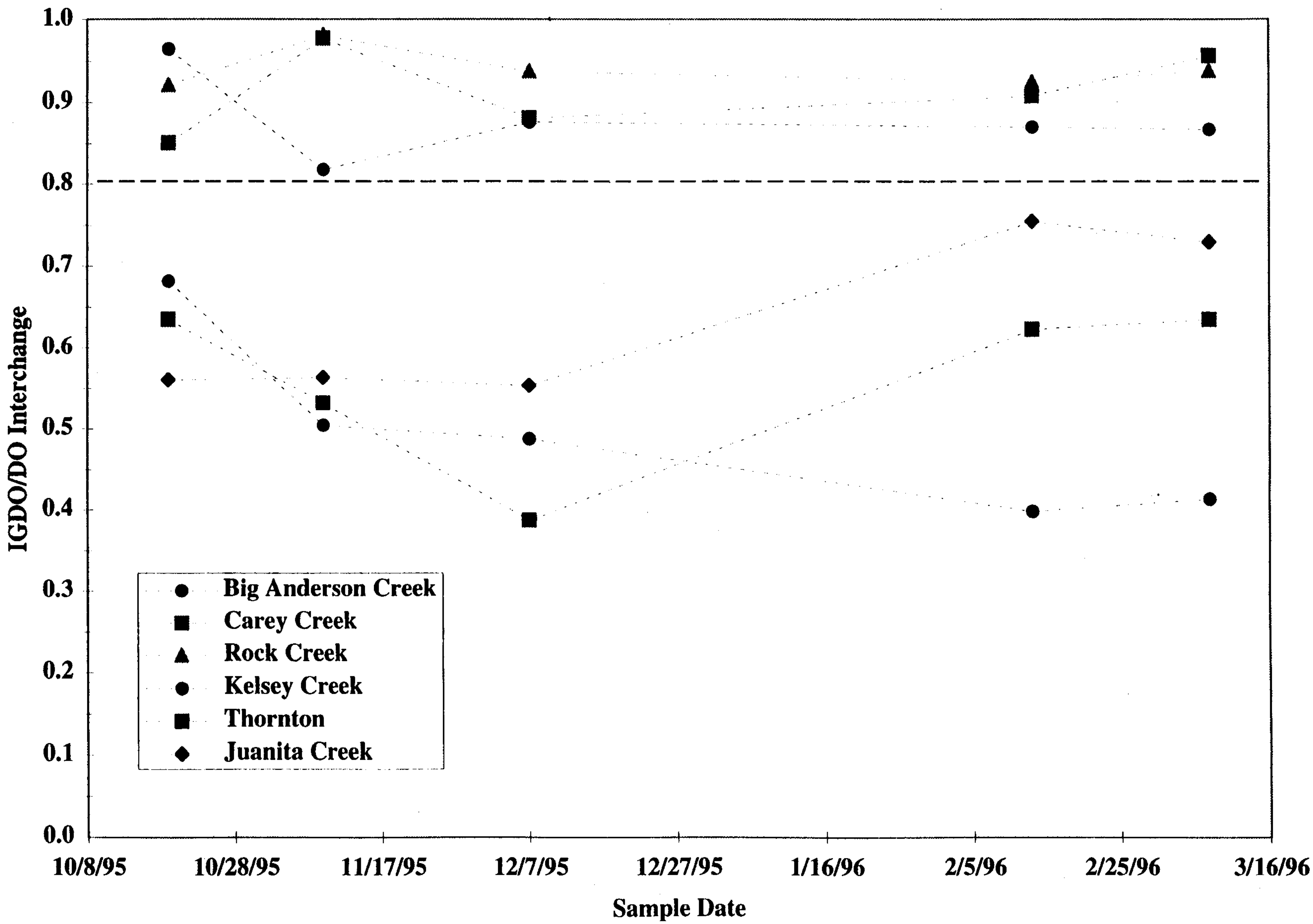

Figure 53: IGDO/DO interchange ratio for typical undeveloped (natural) and urbanized streams in the Puget Sound lowlands during the1995-96 salmonid incubation period.

values were not closely related to substratum fine sediment content (Wydzga, 1996). Nevertheless, if the 80% interchange value is used as a "target" condition, the data indicate that interchange should remain less than that value if % fines are less than around 15%, indicative of acceptable spawning and incubating success. These levels are identical to the limit suggested in Idaho where fine sediment, IGDO, and STE of trout were related in a stream affected by NPS pollution (grazing) (Maret et al, 1993). That level is also similar the target (<11% fines) proposed for forested streams in the PNW (Peterson et al, 1992). Results are similar to Ringler and Hall (1975), who found that sediment from logging reduced IGDO by up to 40% and significantly decreased resident salmonid populations.

Despite the high variability of fine sediment measurements in PSL streams, excessive fines are probably at least a partial explanation for the low observed IGDO/DO interchange. Other factors which may be influencing IGDO concentrations include biochemical oxygen demand within the substrata caused by decaying organic matter, although Wydzga (1996) found no evidence of a relation of IGDO and organic matter in substrata cores. Embeddedness of streambed particles may also prevent the exchange of surface water with the intragravel region resulting in low interchange ratios. Substrata embeddedness was found to be a common problem in highly urbanized streams due to a combination of armoring caused by high flows and sediment deposition. These streams usually had IGDO/DO ratios < 80%.

These results suggest that IGDO is an effective tool for determining adverse effects of urbanization on the quality of salmonid spawning habitat. IGDO has been used to monitor streams in forest management zones in the PNW (Quinn and Peterson, 1994). Additional data are needed to refine the protocol, including the use of in-situ egg-baskets to more specifically establish the link between % fines, IGDO, and STE. In the mean time, a IGDO/DO interchange ratio of 80% appears to be a reasonable.

LWD apparently also has an indirect effect on the quality of spawning habitat. found that Low-gradient reaches in PSL streams may be predisposed to fine sediment deposition, because baseflows are unable to flush fines from the substrata (Wydzga, 1996; Cooper, 1996). All sites with >20% fines were in these low-gradient reaches. However, there were also several low-gradient sites with low % fines. With one exception, all low-gradient sites with low % fines were in reaches with an abundance of instream LWD and wide, intact riparian corridors. LWD appears to have an instream sediment storage function in high-quality PSL streams.

Rearing Habitat

Adequate high-quality rearing habitat is generally recognized as one of the critical "limiting" factors influencing salmonid productivity. This is especially true of winter rearing habitat (high-flow refuge and cover) for juvenile coho salmon (Brown and McMahon, 1987; Reeves et al, 1989; Nickelson et al, 1992). As discussed earlier, LWD is probably the key component of salmonid rearing habitat in small streams in the PNW and is critical to over-winter survival of juvenile coho salmon (Bustard and Narver, 1975; Brown and McMahon, 1987; McMahon and Hartman, 1989; Nickelson et al, 1992). LWD not only provides habitat structure and high-flow refuge, it also provides instream cover for young fish. Coho, in particular, have a strong preference for pools with structurally complex (LWD) micro-habitat (McMahon and Hartman, 1989; Shirvell, 1990). Cutthroat trout appear to have similar preferences for rearing habitat as coho, but may be more adaptable to less than ideal conditions (Heggenes et al, 1991).

Watershed land-use (logging, mining, grazing, and urbanization) has reduced the quantity of rearing habitat (pools) and degraded pool quality (Meehan, 1991). In addition, an overall loss of instream habitat area normally results from human activity in a watershed as tributary streams, headwater wetlands, and off-channel (floodplain) areas are lost. These areas have historically been important nursery areas for young salmonids (Hartman and Brown, 1987). The pervasive and long-term nature of urbanization has effectively reduced instream habitat, especially rearing habitat (Scott et al, 1986; Imhof et al, 1991; Lucchetti and Fuerstenberg. 1993).

The cumulative effects of urbanization have significant reduced the quantity and quality of rearing habitat in these PSL streams. The fraction of total stream area classified as pool habitat decreased as sub-basin development increased, while the fraction of stream area in riffles did not change significantly. The result was a shift in habitat dominance from a "balanced" pool-riffle to a glide-dominant structure. Glides are intermediate habitat units which have characteristics of both pools and riffles but provide few of the functional benefits of either. Although relatively deep and slow during baseflow conditions like pools, glides provide little cover or flow refuge during peak-flow periods. the data were again discretized into development categories to illustrate this shift in habitat character (Figure 54).

There was usually a lack of pool habitat in all PSL streams, as well as a decrease in pool area even with low levels of urbanization (see Figure 27). This lack of stream habitat in PSL streams in general, especially in those urbanized, is apparently due to a corresponding lack of LWD. LWD was consistently and rather strongly related with pool measurements and relationships were non-linear as indicated by better fits with log-transformed data. For example, there was a strong relationship between pool BFW-spacing (# BFWs between pools) and LWD BFW-spacing (# pieces of LWD/BFW length of stream) on a log-log scale. Using the BFW-spacing variables to describe the frequency of LWD and pools, normalizes for stream size and allows for a more representative comparison. Also, the "target" values for proposed here are very similar to those for forested streams (Peterson et al, 1992), indicating the very close linkage between LWD and rearing habitat in PSL streams.

In PSL streams, which are dominated by a pool-riffle morphometry, LWD is primarily responsible for pool formation (Bisson et al, 1987), provides cover on pools, ensures longevity of stable habitat units (Andrus et al, 1988) , and promotes complexity of rearing habitat (Quinn and Peterson, 1994). In short , LWD provides the required instream structure and promotes habitat diversity (Maser et al, 1988). Reduction in rearing habitat quantity and quality in PSL streams is largely due to the combined effects of changed hydrologic regime and decreased LWD quantity (frequency and size) and quality (mature and coniferous).

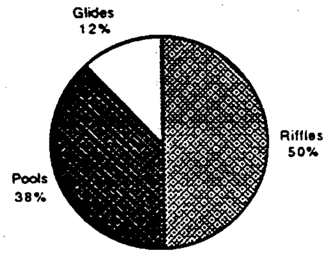

Figure 47a: Sub-basin %TIA < 5%

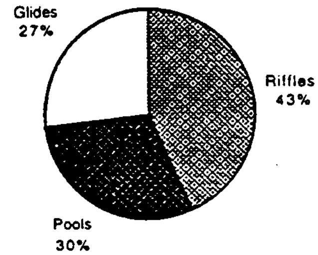

Figure 47b: Sub-basin %TIA 5-10%

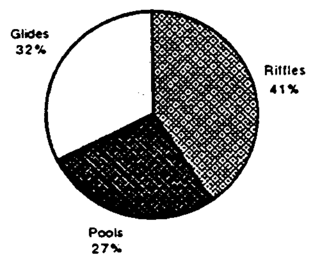

Figure 47c: Sub-basin %TIA 10-20%

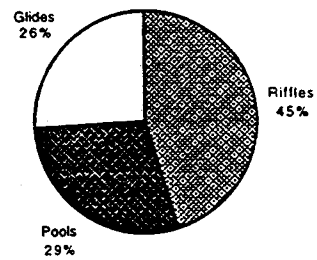

Figure 47d: Sub-basin %TIA 20-30%

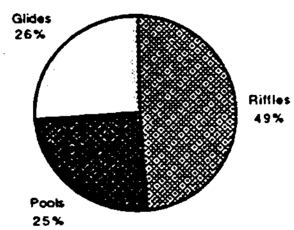

Figure 47e: Sub-basin %TIA 30-45%

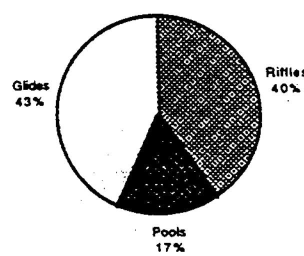

Figure 47f: Sub-basin %TIA > 45%

Figure 54. Habitat unit distribution by development categories for Puget Sound lowland streams.

Stream obstructions can also impair salmonid rearing habitat. Side-channels, backwater areas, and riparian wetlands are some of the most productive coho rearing habitat (Peterson et al, 1992). Roads in the floodplain next to streams often disrupt fish access to these off-channel areas by physically isolating them from the main channel. Even if connected by culverts for drainage purposes, these areas are still often inaccessible to juvenile fish. Culverts are generally designed to allow passage by adult spawners but current velocity or water level may not be appropriate for juveniles.

Target conditions or criteria are herewith proposed for instream physical habitat characteristics based on this work and the literature pertaining to natural, forested and low-gradient lowland streams in the PNW (Table 28). A multi-level approach to habitat quality is used so that resource managers can tailor their effort to meet expectations and feasibility of the situation (see later discussion of stream rehabilitation).

Table 28: Recommended criteria ("targets") for instream habitat in PSL streams.

Instream Habitat Parameter	Salmonid Life-Phase Influenced	Indication of Poor Habitat Quality	Target for Fair Habitat Quality	Target for Good Habitat Quality
Pool Habitat (Surface Area)	Rearing	<30%	30-50%	>50%
Pool Frequency (BFW-Spacing)	Rearing	>4 BFWs	2-4 BFWs	<2 BFWs
LWD Frequency (BFW-Spacing)	Rearing	<1/BFW	1-2/BFW	>2/BFW
Key LWD (Dia. >0.5 m)	Rearing	<20%	20-40%	>40%
Pool Cover	Rearing	<25%	25-50%	>50%
IGDO/DO Interchange	Spawning and Incubating	<60%	60-80%	>80%
Pebble-Count D10 (mm)	Spawning and Incubating	<3 mm	3-5 mm	>5 mm
Fine Sediment (<0.85 mm)	Spawning and Incubating	>20%	15-20%	<15%

LWD = Large woody debris
BFW = bankfull width
IGDO = Intragravel dissolved oxygen

Instream Habitat Quality

As was the case with several quantitative habitat measures (LWD, pools, and IGDO), QHI scores suggest that there is a initial dramatic decline in habitat quality as urbanization (TIA) increases above the 5-10% range (Figure 44). Adverse effects of human intrusion on streams become evident during the transition from largely undeveloped to rural land-use. Similar incipient declines in habitat quality are often observed for other land-use activities; especially timber-harvest. Identifying specific stressors responsible for such a decrease in habitat quality is difficult, however, a multi-metric index such as the QHI was able to detect this change and differentiate between the "best" and "worst" stream-segments, providing an initial assessment tool for stream quality. Not surprisingly QHI values in the mid-range of urbanization (suburban) were quite variable, due to the difficulty of accurately assessing the complexity of stream systems and the variety of impact variables.

The QHI (or a similar survey), performed by trained personnel, can provide an initial estimate of the positive and negative characteristics of a stream section and may be sufficient for management. The QHI can identify problem areas to further evaluate or be used for periodic re-assessment to identify emerging problems. The close relationship between the QHI and various quantitative habitat measures (see Table 21) indicates its value as first-step in the assessment process. The QHI also correlated well with measures of instream biological condition, especially the B-IBI (see Figure 46). Several agencies in Washington and elsewhere use some form of qualitative habitat assessment and usually in conjunction with biological indices (Plafkin et al, 1989; Rankin, 1989; Maxted et al, 1994). In this way, the QHI can provide a link between physical habitat quality and biological integrity. However, the QHI alone is probably inappropriate for proposing restorative and/or management actions, because quantitative assessment is still needed.

Biological Integrity

A close link between watershed urbanization, as basin imperviousness, riparian integrity, and physical stream habitat characteristics has been demonstrated. Watershed urbanization, as well as instream habitat characteristics, was also linked with biological integrity. Demonstrating this relationship between the cumulative effects of development and instream biological integrity was an important objective of this project (see Figure 9).

Salmonids

Watershed urbanization has restructured the salmonid community in small streams of the PSL. Human-induced disturbance, including modification of the hydrologic regime, loss of instream structure (LWD), rearing habitat alteration, loss of riparian forests, and degradation of spawning and egg incubating habitat, has apparently adversely affected coho salmon more than cutthroat trout. This has resulted in a shift in the stream fish community composition with cutthroat significantly out-numbering juvenile coho. Whether this is due to a subtle shift in competitive advantage or to the loss of specific habitat features for coho, or both, is unknown, because the relative susceptibility of coho and cutthroat to disturbance is poorly understood. Results from other workers) are inconsistent, suggesting that neither species has the ability to adapt to all stressors (Moring and Lantz, 1975; Scott et al, 1986; Lucchetti and Fuerstenberg, 1993. Analysis of available fish abundance data for the PSL streams (Figure 45) agreed with results of a previous study of Lake Washington drainage streams that showed a direct relationship between the proportion of impervious surface and the diversity of the fish community (i.e., % cutthroat trout; Scott et al, 1986). A similar trend was observed in streams in the suburban Washington DC area (Ragan et al, 1977). Replacement of a sensitive species by one more tolerant of disturbance is a frequent response of communities to stress (Ward and Stanford, 1983; Resh et al, 1988; Reeves et al, 1993).

Individual habitat indices (character of spawning and rearing habitat) were generally well related to the salmonid index. Habitat diversity and complexity (i.e., LWD quantity and quality) also reflected the shift toward cutthroat over coho. More homogeneous habitat appears to be important in the decline of coho abundance in urban streams. Cover over pools was an especially significant; coho were dominant only in stream segments with >60% cover over pools.

Macroinvertebrates

Both biological indices (B-IBI and coho/cutthroat ratio) displayed the highest values in undeveloped reference streams and proceeded to decrease in a general linear fashion as urbanization increased beyond background levels (see Figure 45). This relationship was very similar to that between urbanization and measures of instream habitat quality where the highest habitat values occurred in undeveloped streams. The transition between undeveloped, reference streams (TIA <5%) and urbanized streams is represented by a B-IBI score of about 33, which corresponds to a coho/cutthroat ratio of between 4 and 5. Only four sites with TIA <5% had B-IBI scores below 32. One of these (lower Big Beef Creek) is located downstream from an impoundment, in a segment with low streambank stability and relatively high % fines in the substratum. Another site (upper Little Bear Creek) is located downstream from several horse farms, while the remaining two have roads and road-crossings nearby.

B-IBI decreased to lower scores as urbanization increased beyond the 5% level of imperviousness (see Figure 45). However, several values from rural and suburban streams are higher than expected. Two sites each on Swamp and North Creeks had B-IBI scores much higher than others at similar urbanization levels. Physical habitat features were common among several reaches that may explain the higher than expected levels, including; wide riparian buffers (>70% of the corridor >30 m wide), few breaks in the riparian corridor (<2/km), a stormflow/baseflow ratio only slightly >20, low substrata % fines and embeddedness, good instream LWD abundance (>1/BFW), good rearing habitat (>40% pools), and good chemical water quality. The cumulative effect of these characteristics is apparently enough to maintain biological integrity at a relatively natural level. The relatively high variability and gradual decrease in nearly all stream quality indices in this rural-suburban range of urbanization indicates that this is a zone where many competitive and synergistic factors affected stream quality. Above a TIA of 45%, the coho/cutthroat ratio was consistently less than one (indicating cutthroat dominance) and the B-IBI was generally below 15, indicating poor biological integrity. Instream habitat was also typically poor above that level as well.

Several physical habitat characteristics significantly affect the benthic macroinvertebrates, and therefore, the B-IBI. The flow regime and its interaction with streambed character are important determinants of taxa composition and diversity in the benthic community (Pederson and Perkins, 1986; Palmer et al, 1992; Borchardt, 1993). B-IBI decreased with an increase in the 2-year stormflow/baseflow ratio (Cooper, 1996). Three streambed characteristics, median particle size, % fines and bed scour or embededness apparently affected the macroinvertebrates.

The highest B-IBI scores usually occurred in reaches with large median (D50) streambed particle size. When D50 was >40 mm, all B-IBI scores were >20 and conversely, when D50 was <25 mm, no B-IBI score was >20. The three highest B-IBI scores were associated with D50s >65 mm. Substrata fine sediment content and embeddedness also affected the amount of living space and habitat for the macroinvertebrate community. B-IBI scores were inversely related with substrata embeddedness and % fines. High B-IBI scores (>35) were observed only in reaches with an embeddedness <20%. The inverse relations between B-IBI and streambed/flow characteristics show that stream quality, as indicated by the macroinvertebrate index, was strongly associated with these physical characteristics. However, the relations do not demonstrate cause/effect; i.e., the cause for the biological effect could be all, any one or none of these variables. Nevertheless, a higher quality benthic community is usually associated with larger streambed particles, because larger particles are more stable during peak flow periods and are less likely to scour, and larger particles create more interstitial habitat for benthic organisms. McClelland and Brusven (1980) clearly showed that macroinbertebrates were increasingly excluded from a cobble-gravel substratum as the content of fine sediment increasingly filled the substratum spaces.

A Benthic IBI for the Pacific Northwest

Macroinvertebrates, such as insects, crustaceans, molluscs, and worms, have many advantages for biomonitoring in the Pacific Northwest. Northwest streams have few fish species (low species richness) but many kinds of macroinvertebrates (Fore et al. 1996). Macroinvertebrate assemblages are ubiquitous, abundant, relatively easy to sample, and encompass taxa with differential responses to a broad spectrum of human activities. Because the life cycles of some benthic invertebrates extend several years, they are better integrators of past human influences than water chemistry is.

Our studies of benthic invertebrates in Puget Sound lowland streams extend work on the effects of landscape alteration on hydrological processes and water quality (Booth 1991), salmon populations (Bottom 1996), and the effects of urbanization on Lake Washington (Edmondson 1991). The decline of Lake Washington involved an easily defined culprit: nutrient enrichment caused by growing quantities of human waste released into the lake. The solution was simple and specific. King County created a regional sewage management agency to control the problem; the improvement in the lake was rapid and dramatic (Edmondson 1991). But without detailed biological studies of streams, managers and scientists cannot always so clearly convey the full consequences of poorly planned development, or reverse degradation.

Based on an integrative analysis of a number of recent studies in the Pacific Northwest, Tennesse, Grand Teton Mountains, and Japan, Karr (1997) recommends a regionally appropriate IBI with 10 metrics (Table 29) in four groups: (1) taxa richness and composition: total number of taxa; number of mayfly, stonefly, caddisfly, and long-lived taxa; (2) tolerant and intolerant taxa: number of intolerant taxa and number of clinger taxa; proportion of individuals belonging to tolerant taxa; (3) trophic or feeding ecology: proportion of individuals that are predators; and (4) population attributes: dominance (percent) of the three most abundant taxa.

As shown by 15 years' experience, when biological indices, conventional measures of chemical water quality, instream habitat assessment and hydrologic analyses

Table 29. Metrics selected for inclusion in a Pacific Northwest benthic invertebrate index of biological integrity (B-IBI) based on six studies of invertebrate responses to human activities (From Karr 1997).

Metric	Direction of Change with IncreasingHuman Influence
Taxa richness and composition	
Total number of taxa	Decrease
Number of Ephemeroptera taxa	Decrease
Number of Plecoptera taxa	Decrease
Number of Trichoptera taxa	Decrease
Number of long-lived taxa	Decrease
Tolerance	
Number of intolerant taxa	Decrease
Number of clinger taxa	Decrease
Proportion of individuals in tolerant taxa	Increase
Feeding ecology	
% of predator individuals	Decrease
Population attributes	
% dominance (2 or 3 taxa)	Increase

complement one another, water resource protection improves; the results will be more cost effective and better safeguard biological resources (Davis and Simon 1995).

Stream Quality Indices

Indices were evaluated by partial least squares (PLS) correlation (a multi-variate statistical technique) and graphical analysis to determine their relative usefulness in detecting stream degradation. Indices were grouped into four broad categories, including urban impact, physical habitat, physio-chemical and biological.

A total of 25 urban impact indices were included. These included measures of basin urbanization (%TIA, road-density, basin % forest, and BMPs) as well as multiple measures of riparian integrity (lateral buffer widths, longitudinal continuity, and quality). Instream physical habitat indices (26) included measures of rearing habitat (pools), LWD quantity and quality, streambank stability, salmon spawning habitat, and the QHI. Physio-chemical indices (12) included measures of fine sediment, IGDO, streambed particle size, baseflow and stormflow chemical water quality, and hydrologic regime and biological indices were B-IBI scores for both sample years.

The urban impact indices that were related most closely ($r = 0.77$) to physical habitat were %TIA, % basin forest cover, riparian corridor width > 30 m, riparian corridor width < 10 m, riparian corridor continuity, and riparian quality (maturity). Urban impact indices (land-use and riparian) were more closely related to physical stream habitat on a segment than cumulative upstream scale, although the difference was not great. The same urban impact indices that were closely related with stream physical habitat were also related with physio-chemical indices ($r = 0.75$). In addition, the % riparian corridor in rural and urban land-use and the number of stormwater outfalls/km were also closely related to physio-chemical indices. The close relation between urban impact and chemical indices is logical because the latter included hydrologic regime (2-year stormflow/baseflow ratio), chemical water quality, and fine sediment. The same urban impact indices that were closely related to both physical habitat and physio-chemical indices were similarly related to biological indices ($r = 0.77$).

TIA and riparian buffer integrity (% corridor with buffer width >30 m) alone were nearly as well related physical habitat indices($r = 0.63$) as the full set of urban impact indices ($r = 0.77$) and were fully as predictive of physio-chemical indices as the full set. The two were also nearly as predictive of biological indices ($r = 0.70$) as the full set of indices ($r = 0.77$).

Physical habitat indices were more closely linked with biological indices (B-IBI) ($r = 0.76$) than the more remote urban impact indices. Although macroinvertebrate index was determined with data from riffles only, non-riffle indices (pool habitat, LWD, and streambank stability) were nevertheless identified as good predictors of biological condition.

Table 30 summarizes the evaluation of stream quality indices. Results of the analysis suggest that there are numerous indices of stream quality that are nearly equal in importance. Depending on the assessment objectives, selection of several indices from each category, along with the B-IBI, would seem reasonable to effectively characterize a particular stream segment.

The urban impact indices identified show the importance of basin imperviousness, natural forest cover, riparian integrity, and roads in defining watershed conditions. Each index identifies a unique and significant feature of a watershed. TIA alone is probably insufficient to describe watershed condition, although it is a well-accepted integrative index of urbanization.

Table 30: Quality indices considered most effective for determining urbanization effects in Puget Sound lowland streams.

Urbanization Indices	Instream Physical Habitat Indices	Physio-Chemical Indices
%TIA	QHI	% Fine Sediment
% Basin Forest Cover	LWD Frequency	IGDO/DO Interchange
% Riparian Buffer > 30 m	LWD Volume	Streamflow Ratio
Stream-Crossings per km	Pool frequency	Pebble-Count D10
% Riparian Buffer < 10 m	Streambank Stability Rating	Sediment Pb and TZn
Basin Road-Density	% Pool Habitat	Stormflow (EMC) TZn
Stormwater Outfalls per km	% Glide Habitat	Baseflow Conductivity
Riparian Quality/Maturity	% Pool Cover	
	% Embeddedness	

PLS = Partial least squares correlation analysis
%TIA = Total impervious area
LWD = Large woody debris
QHI = Qualitative Habitat Index
IGDO = Intragravel dissolved oxygen
EMC = Event mean concentration

The relative effects of riparian integrity and basin imperviousness on biological integrity are illustrated in Figure 55 after the model by Schueler (1995c) and earlier by Steedman (1988). The diagonal dashed lines suggest how riparian integrity may be interacting to modify the effects of urbanization, as indicated by imperviousness, such as high peak flows, scouring, deposition of fine sediment and other physical characteristics listed in Table 30. While high B-IBI scores (>35) were confined to TIAs <5%, wide riparian corridors did exist in these streams at higher TIAs, enlarging the area of excellent quality. Higher than expected B-IBIs occurred in the good to fair quality area that were associated with high riparian integrity. Some of those mid-range urbanization streams contained headwater wetlands, as well as high riparian integrity, which was discussed earlier as a possible explanation for the high biological integrity. Currently, however, riparian integrity declines with urbanization, so that the potential for high riparian integrity to maintain high biological integrity in spite of increasing urbanization can not be adequately tested with these data.

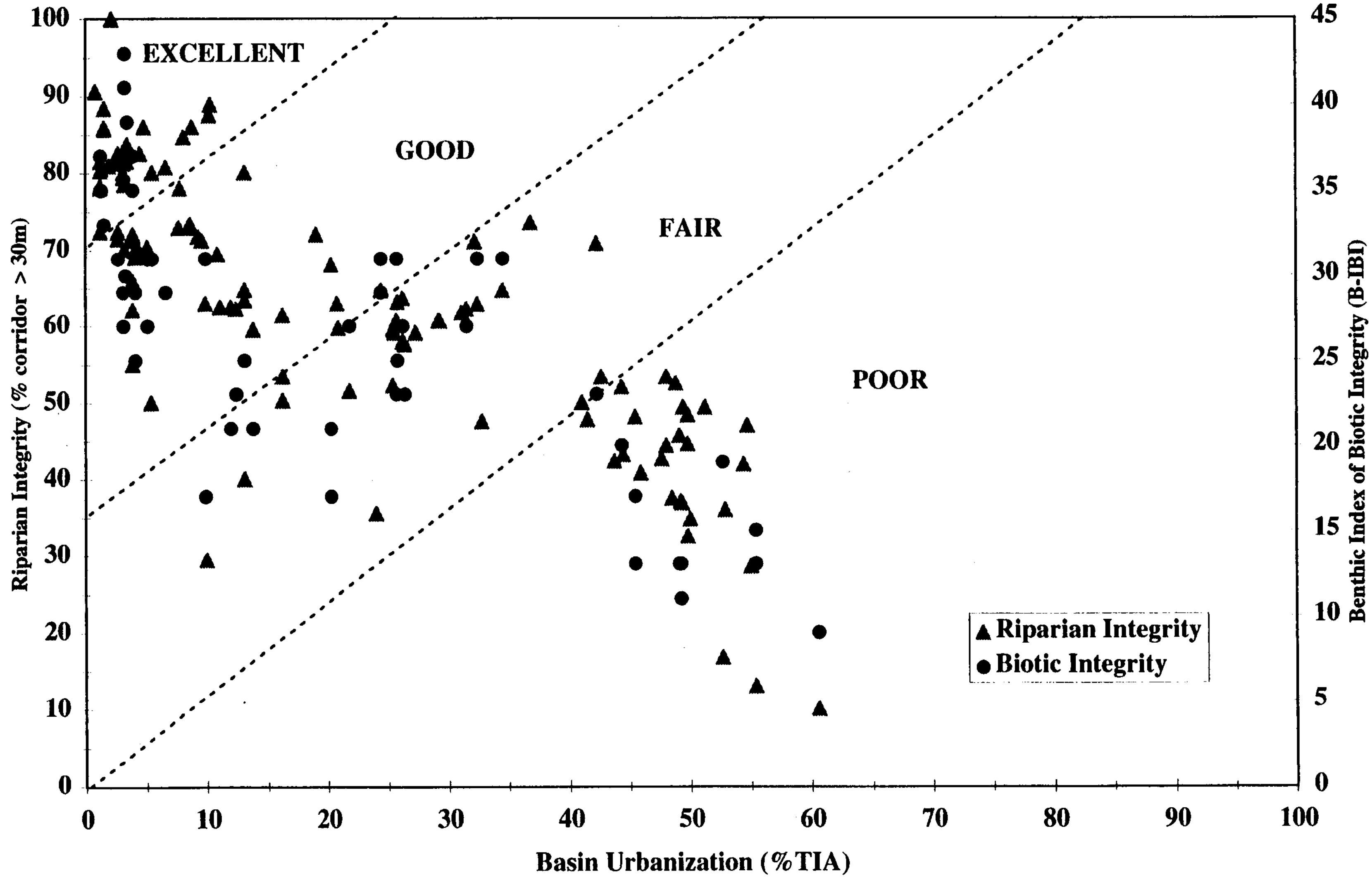

Figure 55: Stream quality; the linkage between watershed urbanization and biological integrity, including the effects of riparian corridor integrity (after Steedman, 1988 and Schueler, 1995).

SUMMARY AND CONCLUSIONS

- Urbanization of Puget Sound Lowland streams was shown to substantially disturb and degrade their physical habitat. Disturbance and degradation of the physical habitat was apparently promoted by greater peak flows as indicated by an increase in the peak (storm) flow to base flow ratio with urbanization. Higher and more frequent peak flows increased stream power as evidenced by greater streambed scour, bank erosion, channel expansion and tributary incision with increased imperviousness. A peak to base flow ratio of 20 corresponded to a decrease in forest area below 30% and increased imperviousness beyond 30%, as well as to a 50% increase in drainage density.
- Physical habitat characteristics that were degraded include; the volume and frequency of large woody debris (LWD), the amount of pool habitat for coho salmon rearing, fine sediment content of the substratum, intra-gravel dissolved oxygen content and a multi-factor qualitative habitat index (QHI). The level of urbanization was indicated by % total impervious surface as well as % forest cover, frequency of stream crossings, road density and quality and extent of the riparian corridor. Relationships between urbanization level indices and stream physical habitat characteristics were represented by correlation coefficients between 0.6 and 0.8.
- Biological condition in the streams was represented by two indices; a benthic index of biological integrity (B-IBI) and the ratio of coho salmon to cutthroat trout. Both indices declined with increasing urbanization. B-IBI declined rapidly in early stages of urbanization such that high B-IBI scores (>35) were observed only at low levels of imperviousness (<5-10%), which indicates greater sensitivity of streams to urbanization than previously reported.
- Degradation of biological condition was apparently caused more by changes in physical characteristics in the stream and riparian corridor than by chemical water quality constituents, because criteria for metals were usually not exceeded either in stream water or sediment unless imperviousness exceeded 45%, while B-IBI had

declined to relatively low values at much lower levels of urbanization. Relative importance of physical habitat characteristics to B-IBI could not be discerned, due to equally strong relationships between B-IBI and the several measures of habitat.

- Riparian corridor quality and quantity also decreased with urbanization, directly affecting the size and frequency of large wood debris (LWD) as well as stream channel stability. LWD of coniferous origin was well represented only in undeveloped streams (imperviousness <5%). Like B-IBI, the frequency and volume of naturally-accrued LWD decreased rapidly as urbanization increased and only those streams with wide (>30 m) continuous buffers (>70% of corridor) had adequate LWD.
- A set of target conditions are recommended for PSL streams based on observations and developed relationships between watershed characteristics and physical/chemical indicators in streams and, in turn, between physical/chemical indicators and biological indicators and findings from the literature and are listed in Table 28.

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