Up an Urban Creek: The Role of Development Patterns in Stream Health

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Abstract

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Population growth in urban and sub-urban areas imposes increasing pressure on stream ecosystems. The "Urban Stream Syndrome" explains the observed negative trends in stream conditions associated with increasingly urbanized watersheds. Urban planners are challenged with addressing increased growth and development while also protecting critical areas such as streams and rivers that provide many services to urban dwellers as well as crucial habitat to fish and wildlife. The purpose of this study was to identify which patterns and characteristics of urban development play a role in stream health and might be managed by local planners. Available data for an index of macroinvertebrate communities, the Benthic Index of Biotic Integrity (BIBI), were used to assess stream health in twenty-two moderately-urbanized drainage basins (13 to 15 percent impervious area) in the Puget Sound lowlands. Land cover patterns in the basin and the riparian area (100 m) were calculated using Fragstats v4 and correlated with stream BIBI score and the metrics that comprise the score. I assess the correlations between infrastructure intensity (road density, number of road crossings, and number of stormwater outfalls) and these response variables. The selected landscape metrics did not explain the variability in the BIBI score. The values in this study may have been within the natural range of variation of the BIBI, and possibly the index only provides a coarse level of information about stream condition. Urban patch density was positively correlated to intolerant taxa richness. Number of road crossings and number of stormwater outfalls were negatively related to intolerant richness. No metrics were related to riparian land cover composition or configuration. These findings show that more dispersed urban land are associated with less disturbance in macroinvertebrate assemblages, as well as fewer road crossings and fewer stormwater outfalls. Planners interested in maintaining stream biological integrity should focus on maintaining patchier development patterns, improving road/stream crossings, and shifting from traditional stormwater infrastructure to more natural drainage systems.

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1. Introduction

Within the past few years the world has experienced a first – more people now live in urban areas than rural areas (United Nations 2011). This trend is projected to continue; by 2050, 69% of the world's population will live in urban areas. In the United States, 82% of the population currently lives in urban areas. This has led to expansion of urban land, and in fact, land consumption for urban/suburban uses has increased faster than population (Alig et al 2004).

How does this land cover change affect local ecosystem processes? Analysis of stream health may be a good way to answer this question as the conditions in a stream are a good indication of the ecosystem processes occurring within its watershed (Karr 1998). Much research has been conducted examining the effects of increasing urbanization on stream health - Walsh et al. (2005) define "Urban Stream Syndrome" as the consistent, negative changes consistently observed in urban streams. Because of increased impervious surfaces, stormwater runs off more and infiltrates the ground less, leading to quicker and higher peak flows in streams which reshapes channel morphologies through stream incision. The stormwater is not filtered by plants and soil microbes, and therefore retains many pollutants as it flows into streams. Biologic communities suffer as a result – fish and macroinvertebrate assemblages decline in urbanized streams.

Expected growing numbers of people living in urban areas around the world imply increasing pressure on stream health. Urban planners are challenged with addressing this growth while also protecting critical areas such as streams and rivers. How can planners mitigate the negative effects of urbanization on stream condition? Understanding the mechanisms that cause stream degradation and the factors that can mitigate the impact of urbanization are more important than ever for planners interested in sustainable development.

The literature on urbanization and streams demonstrates that there is great variability regarding the range of impacts (Booth et al. 2004, Walsh et al. 2005, Alberti et al 2007). Some urban streams remain relatively healthy and pollution free, while others decline, despite having similar levels of urban development. Determining the mechanisms leading to such variability would provide valuable information to planners interested in sustainable urban development. Evidence points to the patterns of urban land cover in the streams' drainage basins as playing an important role in stream health (Alberti et al. 2007, Shandas and Alberti 2009). Better understanding of this role may provide insights into strategies for minimizing urbanization's effects on streams.

The objective of this study is to identify patterns and characteristics of urban development that play a role in stream health and that might be managed by local planners.

1.1 The Urban Stream Syndrome

The "Urban Stream Syndrome" describes the degradation of streams in urban areas. In particular, the impervious surfaces associated with urbanization prevent water from filtering into the ground and instead cause it to run over the surface (Arnold and Gibbons 1996). This alters stream hydrology by increasing peak flows and decreasing the lag time between precipitation and those peak flows (Paul and Meyer 2001, Roy et al. 2005, Walsh et al 2005). These "flashy" flows lead to altered erosion and deposition processes as well as generally wider and deeper channels. Sediment regimes change, with initial increases in fine material and decreases in substrate particle size. Over time, increased scouring from flashy streamflows may wash away smaller particles, leaving larger cobbles and more homogenous substrate (Finkenbine 2000, Paul and Meyer 2001). In other urban streams, fine sediment persists as stormwater travels over the surface and picks up soil (Booth and Jackson 1997). Stormwater infrastructure that efficiently transports rain water to streams exacerbates this problem.

Stream water quality also changes in urbanized watersheds. Dissolved oxygen tends to decrease, leading to potential effects on stream biota (Pail and Meyer 2001, Morse et al. 2002). The effects of urbanization on nutrient levels are somewhat variable. Phosphorus concentrations generally increase with urbanization (Paul and Meyer 2001, Hatt et al. 2004, Brett et al. 2005). Effects on nitrogen concentrations are more variable – Morse et al. (2002) found a positive correlation between stream nitrate concentration and percent imperviousness, while Hatt et al. (2004) found a strong positive relationship between percent imperviousness and ammonium but not for nitrate, and Brett et al. (2005) found no relationship between urban land cover and either nitrate or ammonium concentrations.

These changes in physical and chemical processes ultimately lead to changes in biotic communities. Macroinvertebrate communities are particularly responsive to urbanization (Karr 1998). Some taxa are less tolerant than others to disturbances such as changes in flow regime or pollutant loads. Research on the impacts of urban development on macroinvertebrates shows that such development has little impact on the density or the total number of macroinvertebrates in streams (Pederson and Perkins 1986, Paul and Meyer 2001, Morse et al. 2003, Moore and Palmer 2005). Instead, urban development appears to cause a decline in the richness and diversity through a shift in community composition. Disturbance-tolerant taxa replace intolerant taxa, and often the particular taxa present (or absent) can provide insight into what changes are occurring in the disturbance regime. As a result, the composition of macroinvertebrate communities is a potentially useful indicator for stream condition.

1.2. The Benthic Index of Biotic Integrity

The Benthic Index of Biotic Integrity (BIBI) has been developed as a tool to assess stream condition, and particularly the degree of human alteration to stream processes (Karr 1998). The BIBI is a multimetric index that can be used in statistical analyses to test hypothesis about stream condition. This makes it a useful measure for stream health in this study.

This index combines 10 metrics of stream macroinvertebrate communities into one value that ranges from 10 (poor) to 50 (excellent) depending on the region (Table 1) (Puget Sound Stream Benthos). Overall taxa richness is the number of taxonomic groups present (usually species, with some groups only going to family) (Puget Sound Stream Benthos). Ephemeroptera, Plecoptera, and Trichoptera are orders of insects (mayflies, stoneflies, and caddisflies, respectively) that contain many species sensitive to disturbance. As these species disappear, the richness of these three orders decreases. Clingers are invertebrates that cling to rocky substrate and are sensitive to increases in fine sediment. Long-lived invertebrates require at least a year to complete their life cycle, making them vulnerable over a longer time period to human disturbance. Intolerant taxa are those groups that are most sensitive to disturbance and are the first to disappear with urbanization. Percent dominant is the number of individuals in the three most abundant taxa divided by the total number of individuals in the sample, and measures diversity of the sample. Predator percent is the number of individuals that are predators divided by the total number of individuals in the sample. More predators mean there are healthier groups of prey. Percent tolerant measures the proportion of the sample that is tolerant individuals – those that are unaffected by human disturbance.

Table 1: Score card for calculating the Benthic Index of Biotic Integrity score of a stream. For each of the 10 categories, a score of 1, 3, or 5 is assigned based on the values of the category.

Score:	**1** **Very Poor**	**3** **Fair**	**5** **Excellent**
Taxa Richness	[0, 15)	[15, 28]	(28, ∞)
Ephemeroptera Richness	[0, 4]	(4, 8]	(8, ∞)
Plecoptera Richness	[0, 3]	(3, 7]	(7, ∞)
Trichoptera Richness	[0, 5)	[5, 10)	[10, ∞)
Clinger Richness	[0, 8]	(8, 18]	(18, ∞)
Long-Lived Richness	[0, 2]	(2, 4]	(4, ∞)
Intolerant Richness	[0, 2]	(2, 3]	(3, ∞)
Percent Dominant	[80, 100]	[60, 80)	[0, 60)
Predator Percent	[0, 10)	[10, 20)	[20, 100]
Tolerant Percent	[50, 100]	(19, 50)	[0, 19]

Stream BIBI scores decline with increasing urban development. Morely and Karr (2002) studied two streams in western Washington and found a negative correlation between BIBI score and both percent impervious area and percent urban land cover of the stream's sub-basin. Alberti et al (2007), who studied more streams, also found a negative relationship between BIBI score and percent impervious area.

There are many possible mechanisms for why macroinvertebrate communities decline in more urbanized watersheds. Booth et al. (2004) found a relationship between altered hydrology and BIBI score in streams in the Puget Sound area. Streams with higher "flashiness"[1] had lower BIBI scores and were associated with drainage basins containing higher levels of urban development. Kennen et al. (2012) studied streams in the northeastern and midwest United States

[1] "Flashiness" was measured by the fraction of the of a year that daily mean streamflow exceeds annual mean streamflow and the fraction of a multi-year period that streamflow exceeds the streamflow of the twice-a-year flood (Booth et al. 2004)

and found temporal changes in macroinvertebrate assemblages to be related to streamflow metrics.

LWD recruitment also declines with urbanization, which may remove an important component of invertebrate habitat. Stewart et al. (2012) found significantly fewer pieces of large wood and lower volumes of LWD in urban streams than in forest streams. These measures were positively correlated with invertebrate species richness and an "invertebrate community index," suggesting that changes in LWD recruitment may be one mechanism for changing invertebrate assemblages in urban streams.

Changing sediment regimes in urban streams may be another mechanism of macroinvertebrate assemblage shifts. Substratum is an important component of freshwater invertebrate life – if this changes, changes in invertebrate community structure might be expected (Minshall 1984).

1.3 Legal and Policy Context

In 1972, the United States Congress passed the Federal Water Pollution Control Act, or the Clean Water Act (CWA). The goal of the Clean Water Act is "to restore and maintain the chemical, physical, and biological integrity of the Nation's waters" (33 U.S.C. §1251 et seq. 1972). The CWA outlawed point source discharge of pollutants into navigable waterways without a permit and established a structure for states and tribes to regulate water quality of their surface waters. The CWA is administered by the Environmental Protection Agency (EPA) and states and tribes establish water quality criteria that include chemical and biological standards. In 1987, Congress passed amendments to the CWA to address the additional sources of pollution, including urban stormwater, requiring municipalities to obtain permits for stormwater discharge. In Washington State, these permits are obtained from the Department of Ecology (DOE), which establishes and monitors water quality criteria. Better understanding of the mechanisms through which urban development impairs stream condition will help these municipalities meet desired water quality standards in their streams. Identifying characteristics of development that minimize the impact to streams will provide planners with tools to manage their stream health and comply with the CWA.

In 1990, the State of Washington passed the Growth Management Act (GMA), which requires fast-growing cities and counties to contain growth within urban areas while also protecting environmentally critical areas (including streams and rivers) by establishing critical areas ordinances (RCW 36.70A). These requirements are often in conflict with one another, as increasing population density in designated urban growth areas simultaneously degrades stream health (Mills et al. 2008). Again, identifying ways to develop that have less impact on streams may help planners attempting to meet both of these GMA goals.

In 1995, the State amended the GMA to require the use of "best available science" when writing these ordinances (RCW 36.70A.172). Most ordinances rely on buffers (which are to remain undeveloped) to protect rivers and streams – in King County, fish-bearing streams must have 115 feet of buffer (KCC 21A.24.358). Confirming the extent to which such buffers actually protect streams and identifying other potential measures will be important for supporting and improving these ordinances.

1.4. Research Questions and Hypotheses

Stream ecologists have described variability in the extent to which stream health declines with increasing urbanization. However, what explains this variability is unclear. Possibly, the patterns of the urban development matter. Alberti et al. (2007) found that B-IBI scores were negatively related to the aggregation of urban land, mean patch size of urban land, number of road crossings of streams, and road density. However, the study watersheds also had varying quantities of urbanization, which likely contributed to some of the correlations. Shandas and Alberti (2009) found that percent of the riparian area covered in vegetation was positively correlated with B-IBI score. They also found the aggregation of forest land in the watershed to be positively correlated with B-IBI score, but this effect was confounded with varying quantities of forested land. Little research has been published that investigates mechanisms of stream health decline when controlling for quantity of urban land.

This thesis seeks to answer the following questions: (1) Among streams of similar quantities of urbanization, what patterns of urban land cover and infrastructure are related to stream health, as measured by the BIBI? (2) Among streams of similar quantities of urbanization, what differences in physical characteristics are seen between streams with high and low BIBI scores? By controlling for quantity of urban land, I hope to identify factors that lead to stream degradation (or maintain stream health) that could potentially be managed by planners. The following hypotheses were tested:

1. Stream health, as measured by the B-IBI scores is related to various measures of urban patterns (such as aggregation and patch size of urban land, as well as percent forest).
2. Stream health, as measured by the B-IBI scores, is related to infrastructure intensity, such as road density, number of roads crossing streams, and stormwater outfalls.
3. Stream health, as measured by the B-IBI scores, is related to land cover composition within the riparian zone.

2. Methods

2.1 Study Region

This study was conducted on 22 streams in the Puget Sound Lowland region (Figure 1). The Puget Sound region has been the subject of previous studies on urbanization and stream health, and therefore serves as a good location for further understanding the mechanisms of urban stream health and BIBI decline (Morley and Karr 2002, Booth et al. 2004, McBride and Booth 2005, Alberti et al. 2007).

Geologically, the region has been shaped by a series of ice sheets that have advanced and retreated across the Puget Lowland landscape between the Cascade and Olympic Mountain Ranges as recently as 14,000 years ago (DNR 2009). These sheets deposited unconsolidated sediment that dominates the current geology. Prior to European settlement, the region was covered in old-growth conifer forests largely consisting of Douglas fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*), and western red cedar (*Tsuga heterophylla*) (Franklin and Dyrness 1973). Since European settlement a century ago, many people have settled in the region, bringing the current population to 3.9 million (PSRC 2010). This has led to dramatic changes in land cover and more recently, urbanization has been a major driver of land cover change as the Seattle metropolitan region grows (Sorenson, 2013.).

2.2 Site Selection

Study streams were chosen from an online stream monitoring database, Puget Sound Stream Benthos (PSSB). The PSSB database was created in a collaborative effort by the City of Seattle, King County, Pierce County, and Snohomish County to make BIBI monitoring data available for analysis and comparison. An additional 18 organizations/municipalities contribute monitoring data to the database. All streams in this study were sampled by King County Department of Natural Resources.

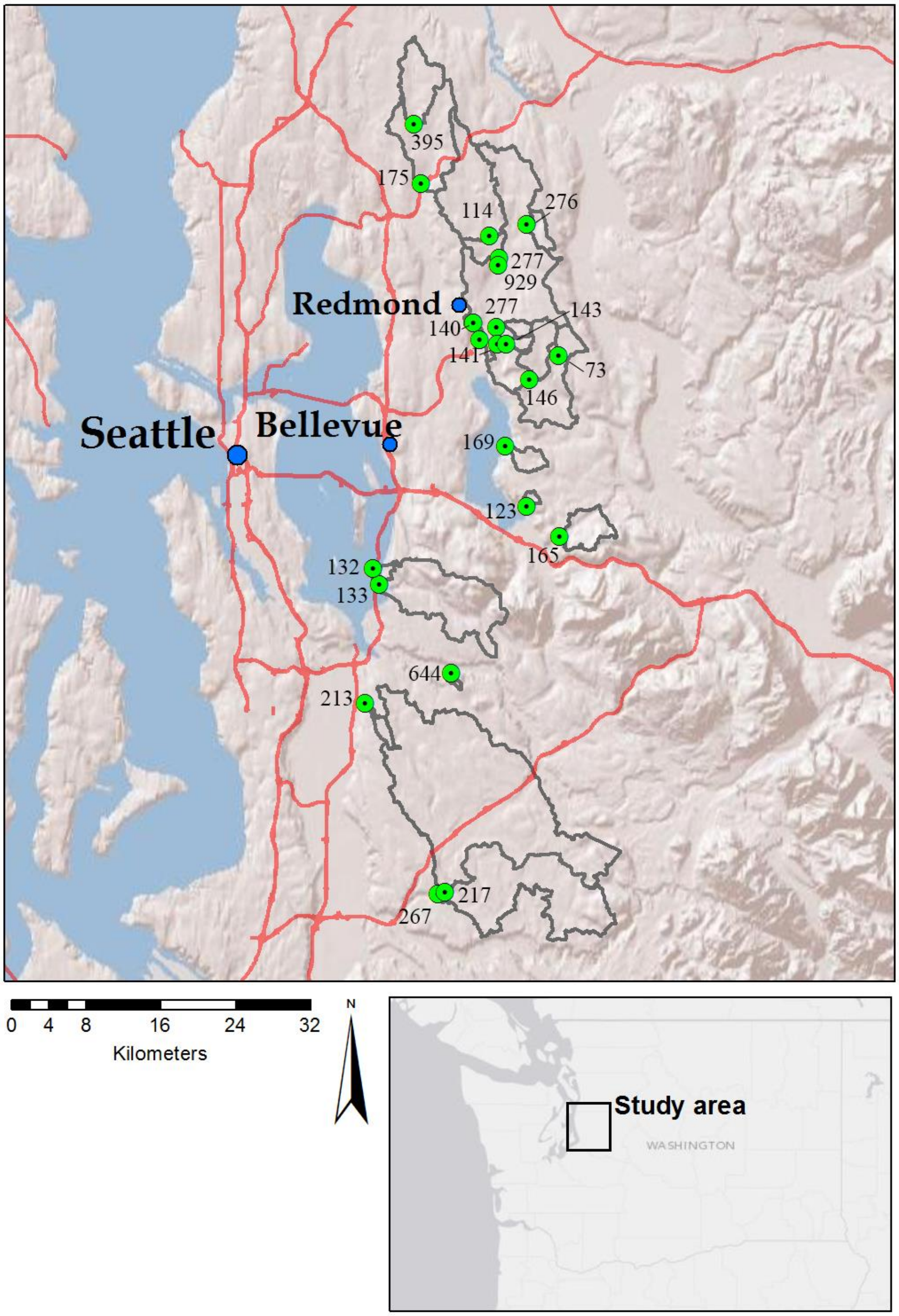

Figure 1: A map of study sites, including codes from Puget Sound Stream Benthos, with drainage basins, major cities, and highways.

In 2011, the Environmental Protection Agency delineated drainage basins for a large portion of the sampling sites within the database. The delineation was performed using a 30-m Digital Elevation Model (DEM), obtained from NHDPlus, with the ArcHydro extension for ArcGIS (ESRI 2011). As part of this collaboration, basic land cover attributes were calculated for the basins using the 2006 National Landcover Dataset.

For this study, the pre-existing land cover data were used to identify stream sample sites in the PSSB database that have 28-32% of their drainage basins covered in urban land. Forty-seven sampling locations fell into this group. These sites were mapped in ArcGIS and sites that overlapped were merged. The sites were sampled over various years between 2002 and 2011, following the procedures described in Morley (2000) (using a species-family identification approach), but no yearly trends were discernible when BIBI scores were plotted over the years. To make the results comparable, only streams that were sampled from 2006-2008 were included (the land cover dataset used in the analyses was from 2007), leaving twenty-two streams.

To better understand the mechanisms leading to differences in BIBI scores, the four highest scoring and four lowest scoring streams of the sample were visited and additional information on habitat quality was gathered. It was assumed that the streams at the extremes would be more likely to display subtle trends. Specific details on these eight study streams are provided in Table 1. It should be noted that these streams were visited in 2012 whereas the BIBI scores are from 2006-2008, so the physical stream measurements collected in this study may not reflect the conditions that existed when the sites were sampled.

Table 2: Streams with the highest and lowest BIBI scores, for which additional habitat data were collected. Unnamed stream is later referred to as "Tiny."

Stream	Median BIBI score 2006-2008	Basin size (ha)	Nested within another stream
Unnamed	34	167	Cottage Lake
Cottage Lake Creek	36	2918	Bear
Covington Creek	38	4540	Soos
Big Soos Creek	36	16015	
Struve Creek	22	292	Bear
Little Bear Creek	22	1205	
Evans Creek	22	3273	Bear
Bear Creek	20	12023	

2.2. Urban Pattern and Infrastructure Analysis

Metrics for various patterns of urban land cover and infrastructure were calculated by first delineating the drainage basin for each sample site, overlaying these drainage basins with land cover data and performing the necessary function (described below).

Drainage basins were delineated using a 10-meter digital elevation model (DEM) obtained from the Natural Resources Conservation Service's Geospatial Data Gateway. The DEM was analyzed using the ArcHydro extension of ArcGIS (ESRI). The coordinates for the BIBI sampling sites were used as the pour points so that only the land area contributing to those sites was included.

A 30-m land cover raster for 2007 (Figure 2) was obtained from the Urban Ecology Research Lab at the University of Washington. The dataset identifies fourteen categories of land classification, including three categories for urban land. "Heavy urban" contains greater than

80% impervious area; “medium urban” contains 50-80% impervious area; and “light urban” contains 25-50% impervious area. For the calculation of urban land patterns, the land cover dataset was re-classified so that urban land cover was assigned a 1 and all other cover was assigned a 0. Two options of reclassification were used – one where “heavy,” “medium,” and “light” urban were considered urban, and one where only “heavy” and “medium” urban were considered urban. The reclassified dataset was then extracted for each drainage basin (Figure 3a) and fed into Fragstats v4 (McGarigal, et al. 2012). Landscape metrics computed for each basin are described in Table 2.

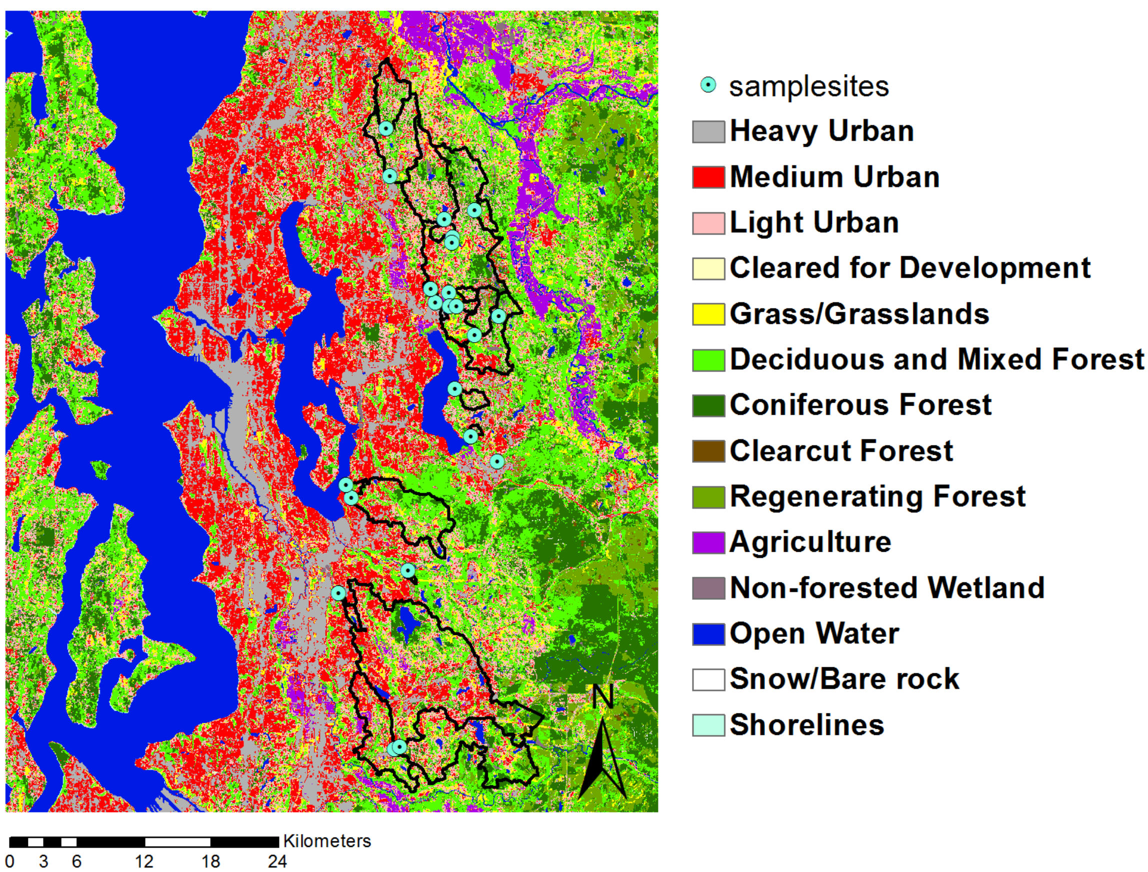

Figure 2: Land cover classification for study area. (Data obtained from the Urban Ecology Lab at the University of Washington)

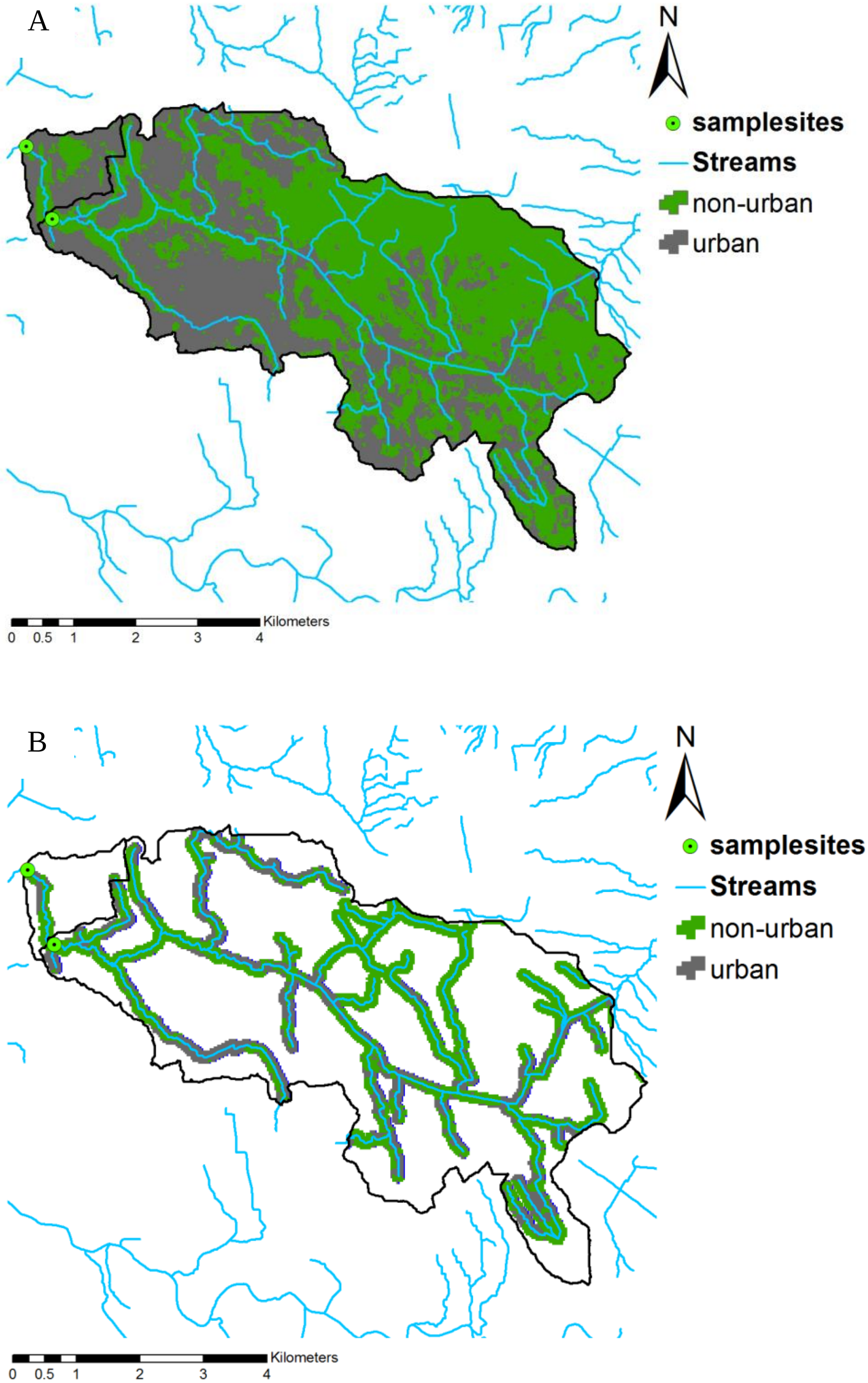

Figure 3: Land cover reclassifications for a selected basin. (A) Urban/non-urban reclassification of landscape (B) 100-m riparian zone reclassification

Table 3: Formulas used to calculate landscape patterns for urban and forested land in the drainage basins and in the riparian area (McGarigal, et al. 2012).

Metric	Formula	Description
Aggregation Index	$\left[\frac{g_{ii}}{\max g_{ii}}\right]100$	Percent like adjacencies divided by maximum possible like adjacencies
Percent Like Adjacencies	$\left(\frac{g_{ii}}{\sum_{k=1}^{m} g_{ik}}\right)100$	Percent of adjacent pixels of the same class type summed over all pixels of class *i*
Mean Patch Size	$\frac{\sum_{j=1}^{n} x_{ij}}{n_i}$	The average size of patches of class *i*
Largest Patch Index	$\frac{\max(a_{ij})}{A}$	Area of the largest patch of class *i* divided by area of landscape
Patch Density	$(\frac{n_i}{A})(10{,}000)(100)$	Number of patches of class *i* divided by area of landscape, converted to 100 hectares
Edge Density	$\frac{\sum_{k=1}^{m} e_{ik}}{A}(10{,}000)$	Sum of edge lengths of patches of class *i* divided by landscape area, converted to hectares

To analyze the riparian area, stream data were obtained from King and Snohomish Counties. A 100-m buffer was generated around the streams and then overlain with the reclassified land cover dataset (Figure 3b). The percent of the riparian zone covered in urban land was then computed using zonal statistics.

The patterns of forested land in the drainage basins were computed similarly to patterns of urban land. The original land cover dataset was reclassified so that forested land was assigned a 1 (including coniferous, deciduous, and regenerating forest), and all other land was assigned a 0. This dataset was then extracted for each drainage basin and fed into Fragstats v4. The metrics calculated are the same as those described in Table 2. Riparian forest was also analyzed using the same buffers as the urban land analysis.

Two types of infrastructure were analyzed – roads and stormwater facilities. Street right-of-way (ROW) data were obtained from King and Snohomish Counties. This is the area of land managed by the county and municipalities as streets and sidewalks. In the study region the ROWs are paved. The proportion of each basin covered in ROWs was calculated. The ROWs were also intersected with the stream layer to identify places where roads crossed streams. The number of crossings per kilometer of stream were counted for each basin.

Stormwater facility data were obtained from King County and any overlapping municipalities (Woodinville, Redmond, Sammamish, Newcastle, Renton, Kent, Covington, and Maple Valley). Stormwater outfalls within 20ft of the streams layer were extracted for each basin and counted, and then standardized by dividing by the length of the stream and multiplying by ten.

2.3 Data Collection at Extreme-Scoring Streams

The eight extreme-scoring streams were visited in September of 2012. The coordinates for the BIBI sampling sites were plotted in Google Earth (2012) to identify the exact locations for sampling (a map of the streams is in Appendix C). Upon arrival at the stream, bankfull width was measured and then multiplied by ten to determine the size of the reach. Bankfull width and depth (at the thalweg) were then measured at five equal intervals along the reach.

A modified Wolman pebble count (Wolman 1954) was also conducted along the reach, which was divided into ten equal intervals for the count (so interval widths varied by length of reach). At each interval, the wetted width of the stream was divided into ten equal sub-intervals. A pebble was picked blindly at each sub-interval and its b-axis was measured to the nearest tenth of a centimeter. If only sand, silt, or clay was present at the sub-interval, b-axis length was estimated using standard sizes for these soil categories (1mm for sand, 0.05mm silt, and 0.001mm for clay). A total of 100 pebbles were measured for each stream.

Large woody debris (LWD) (greater than 10cm in diameter and 3m in length) were tallied along the reach, as well. Only LWD that was physically in the stream (either partially or entirely) was counted.

Finally, a sediment sample was taken from a representative area of the stream bed, generally riffles (except for one stream that had no riffles in the reach). If the sediment appeared to change dramatically from one part of the reach to another, then an additional sample was taken and combined with the original. Samples were taken using a McNeil core sampler that was inserted 10cm into the streambed. All sediment contained in this 10cm-deep area was scooped out and placed in a bucket. The sediment samples were allowed to air dry for approximately one month and then were sorted using soil sieves into three groups – larger than 4mm, between 2 and 4mm, and less than 2mm. Two mm is considered a biologically significant size for invertebrates and salmon (Finkenbine et al. 2002). This was used to calculate percent fines for each reach (percent less than 2mm).

Bankfull discharge was estimated using regression equations specific to the Pacific Northwest from Castro and Jackson (2001) and verified using regression equations from the United States Geologic Survey for Washington State (Sumioka, Kresch, and Kasnick 1998).

2.4 Statistical Analyses

All land cover pattern metrics and infrastructure metrics were compared to both median and mean BIBI scores for the 3-yr period (2006-2008) using a Pearson correlation analysis for metrics that were approximately normal (patch density, percent like adjacencies, mean patch size, largest patch index, edge density, aggregation index) and using Spearman's rank analysis for metrics that did not meet the assumption of normality (road crossings, stormwater outfalls). A similar analysis was carried out for metrics that comprise the BIBI score. Metrics for the high-scoring and low-scoring streams were compared using a two-sample t-test assuming unequal variances, or a one-sample t-test if the correlation analysis indicated a directional trend. Because sample sizes were small, statistical significance was indicated by p-values less than 0.10.

3. Results

3.1 Basin Descriptions

Median BIBI scores varied from 20 to 38 for the study basins between 2006 and 2008 (Figure 4). Median intolerant taxa richness, which proved to be significantly related to several factors, ranged from 0 to 3 (Figure 5). One stream had a value of 10 in 2007 – several other metrics were also high for this stream in 2007, as well as for other streams nearby. As a result, there was no conclusive evidence to consider this stream an outlier, and it was included for all the analyses. Median values of intolerant richness for the three study years were used in correlations to moderate such outliers.

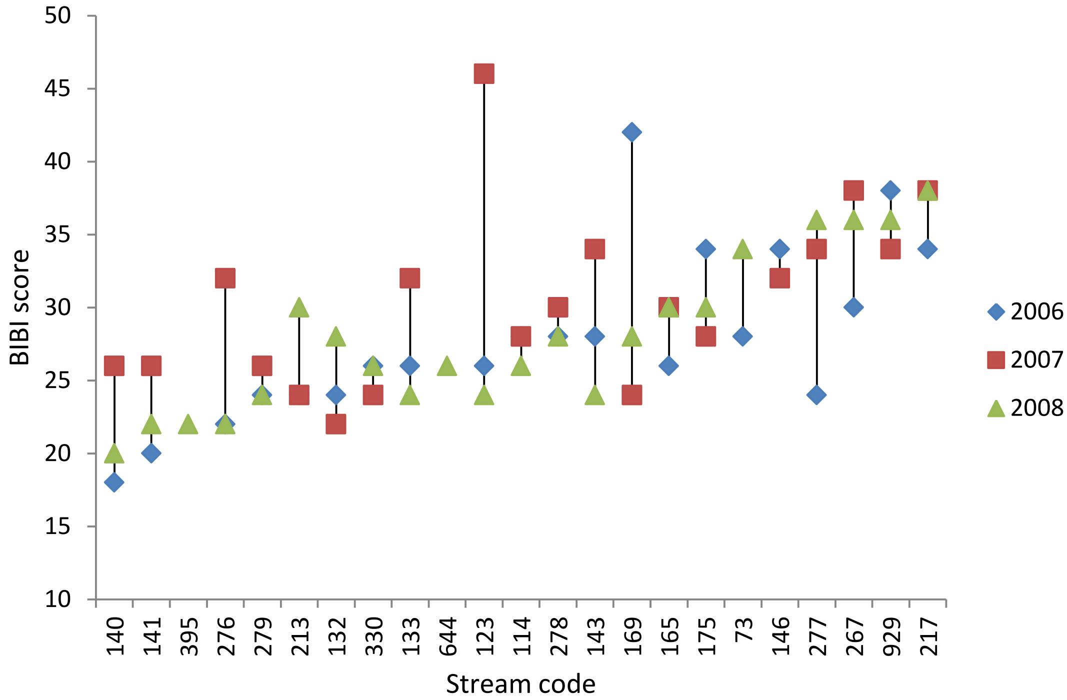

Figure 4: BIBI scores for study stream obtained from Puget Sound Stream Benthos. Streams are ordered left to right by increasing median BIBI score.

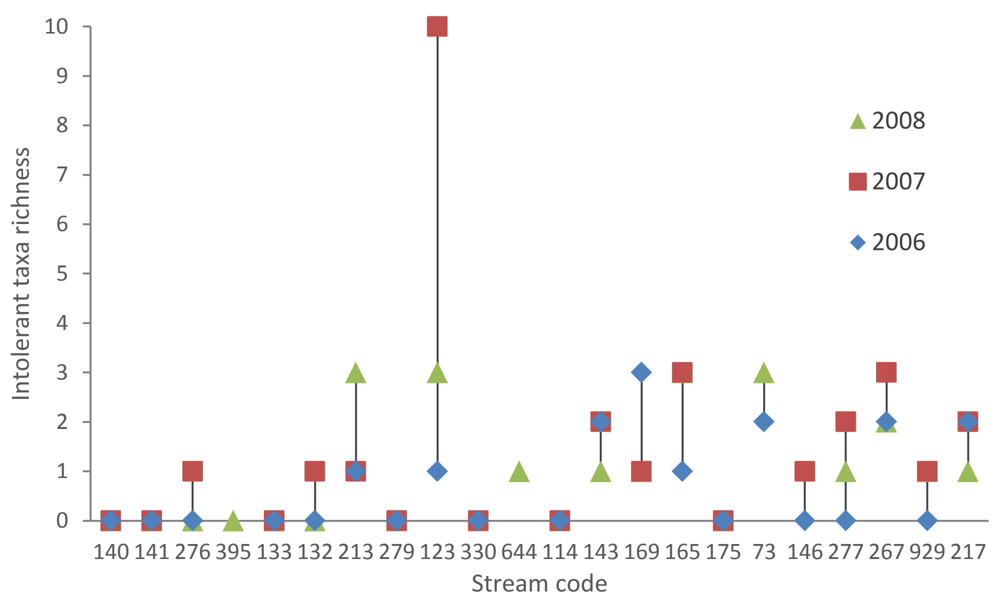

Figure 5: Intolerant taxa richness measured at each stream between 2006 and 2008. Streams are ordered left to right by increasing median BIBI score. See text for analysis of stream 123 with a large value for 2007.

Figure 6 shows land cover composition among the study basins based on the 2007 land-cover dataset and Table 3 shows summary statistics for the basins. The analysis with the 2007 land cover yields different results for basin composition from the analysis used to initially identify the study streams. This is likely because the land cover classifications differ between this dataset and the 2006 Landsat dataset. A simple linear regression found no relationship between percent urban (either for considering "light urban" as urban or not) and BIBI score or for any metric that comprises the BIBI score.

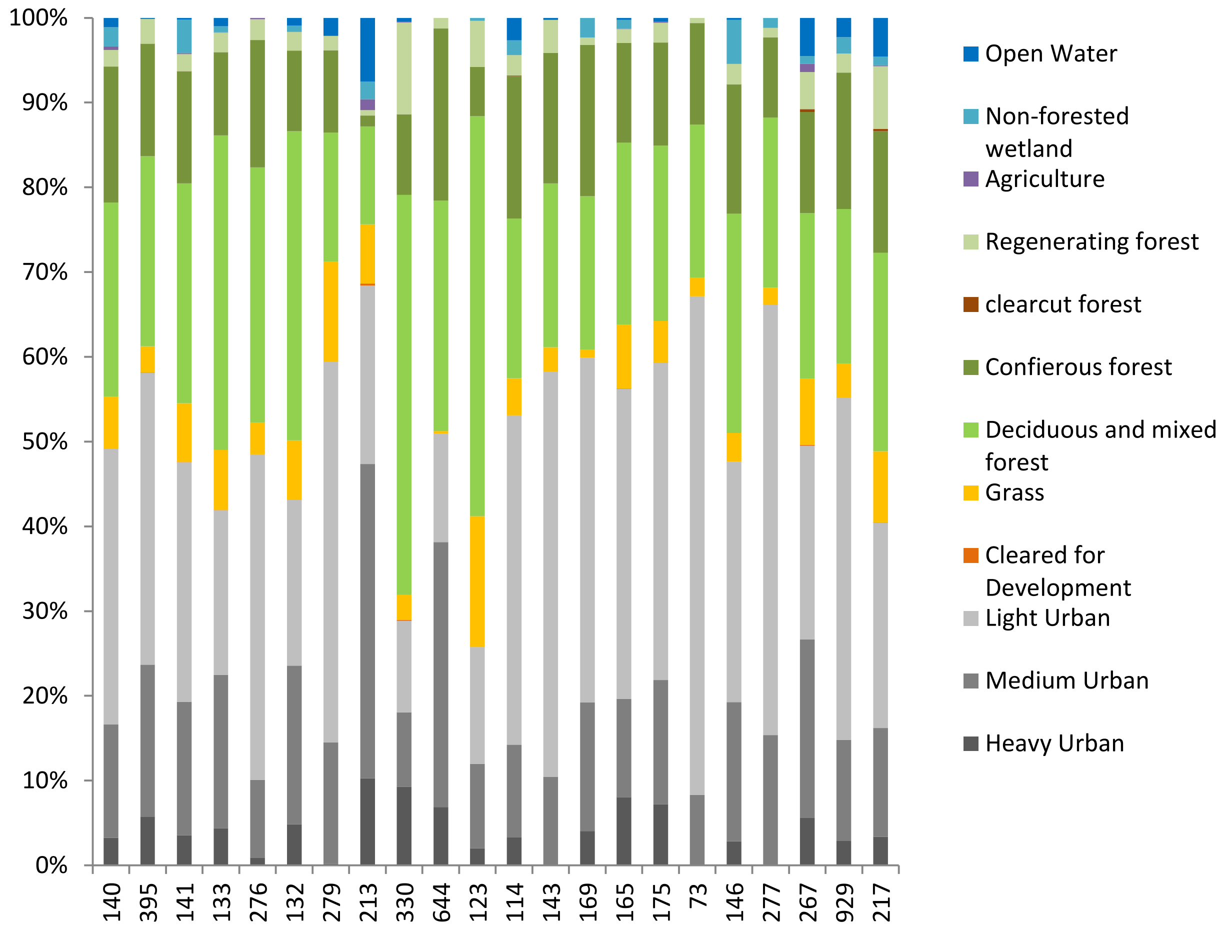

Figure 6: Composition of land covers for each study basin in 2007. Data obtained from the Urban Ecology Laboratory at the University of Washington. Basins are order left to right by increasing median BIBI score.

Table 4: Summary statistics for basins in the study.

	Area (ha)	Impervious Surface (%)
Mean	2783	13.7
Median	1616	13.1
Minimum	29	9.6
Maximum	16015	23.7

The total area of the sample basins varied from 29 ha to 16,000 ha. A Pearson correlation found no relationship between basin area (log-transformed for normality) and BIBI score, but there was a significant negative relationship with intolerant taxa richness (p=0.02, r = -0.49). Figures 7 and 8 show the distribution of areas and percent impervious surface among the basins. Impervious surface only varied slightly among the basins, with a mean of 13.7 percent and was unrelated to either BIBI score or intolerant species richness.

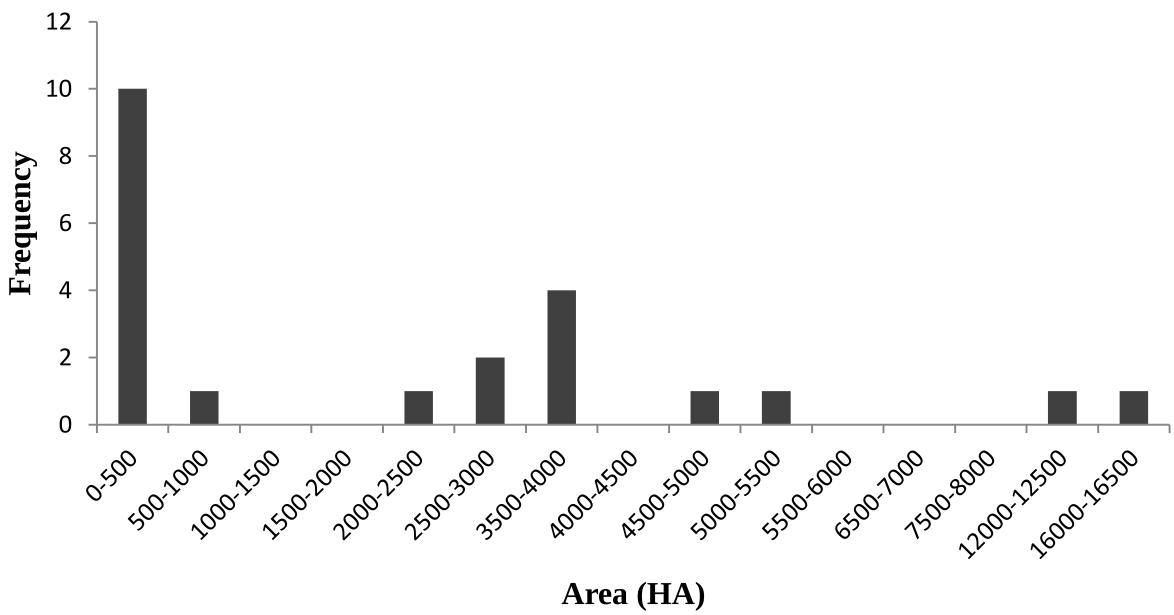

Figure 7: Frequency of basin sizes among study basins.

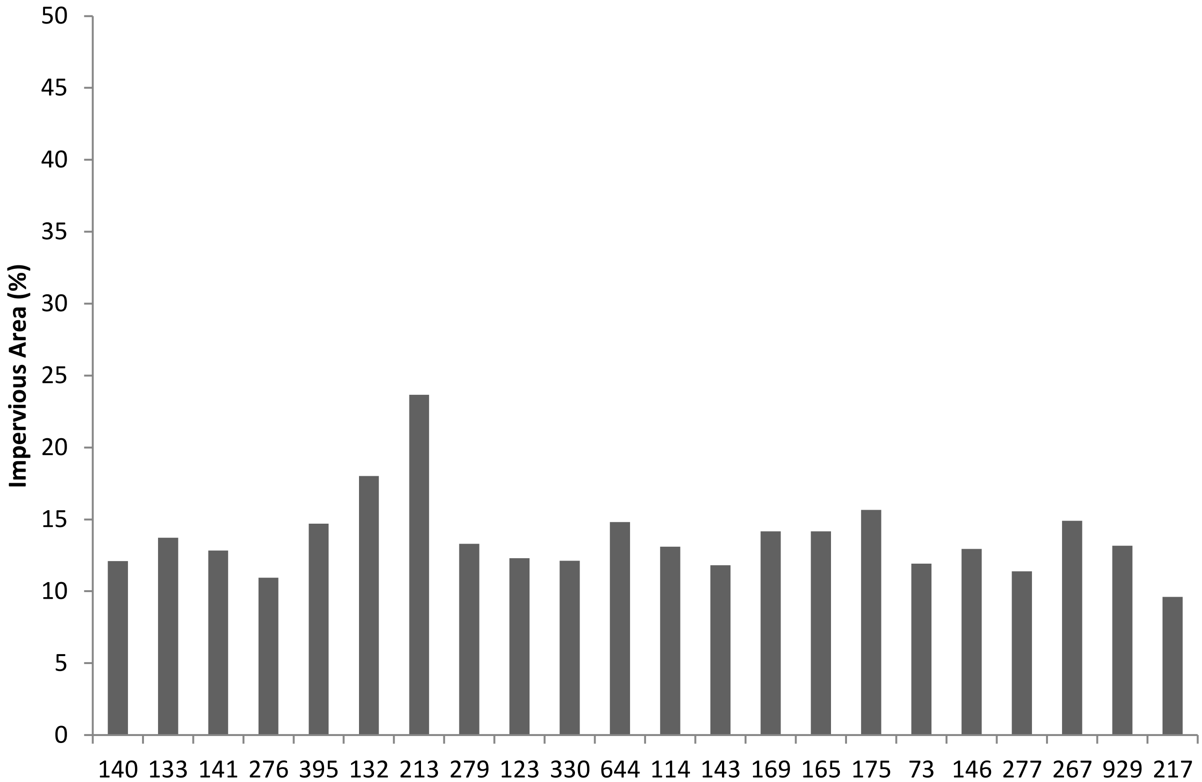

Figure 8: Percent of each study basin covered in impervious surfaces in 2006. Data obtained from the Washington Department of Ecology.

3.2 Basin Urban Patterns

Summary statistics for urban patterns are shown in Tables 4 and 5. These metrics changed somewhat depending on whether light urban was considered as urban or not, with generally lower values when light urban was excluded, except for patch density, which was higher.

Table 5: Summary statistics for urban patterns (heavy, medium, light urban) among basins. PD = patch density, LPI = largest patch index, ED = edge density, PLADJ = percent like adjacencies, AI = aggregation index

	PD	LPI	ED	MN_PatchArea (m^2)	PLADJ	AI
Mean	0.58	36.7	29.3	124.1	82.3	84.3
Median	0.53	35.1	30.1	93.1	83.9	85.3
Minimum	0.11	8	20.3	17.2	60.2	65.8
Maximum	1.50	66.1	37.9	594.3	86.6	88.8
Std. Dev.	0.29	17.8	4.7	115.1	5.4	4.7

Table 6: Summary statistics for urban patterns (heavy, medium urban) among basins. PD = patch density, LPI = largest patch index, ED = edge density, PLADJ = percent like adjacencies, AI = aggregation index

	PD	LPI	ED	MN_PatchArea (m^2)	PLADJ	AI
Mean	0.95	7.78	15.7	27.1	72.8	75.8
Median	0.89	6.09	14.6	24.3	77.2	81.3
Minimum	0.42	0.16	11.1	4.47	40.9	41.5
Maximum	3.31	26.82	22.4	83.8	84.4	85.2
Std. Dev.	0.60	6.31	3.29	19.0	11.9	10.8

Urban Infrastructure

Summary statistics for urban infrastructure in the study basins are shown in Table 6. The percent of land covered in roads varied from 4 to 10 percent and the number of road crossings per kilometer of stream varied from 0 to 2.63.

Table 7: Summary statistics for urban infrastructure in study basins.

	Percent roads	Road-Stream crossings per km of stream
Mean	7.5%	1.22
Median	7.6%	1.11
Min	4.0%	0.0
Max	10.2%	2.63

Forest Metrics

Summary statistics for forested land in the study basins are presented in Table 7. Forest cover in the basins varied from 15.5 to 67.5 percent with all but four basins between 30 and 50 percent.

Table 8: Summary statistics for forest patterns among basins. PD = patch density, LPI = largest patch index, ED = edge density, PLADJ = percent like adjacencies, AI = aggregation index

	Percent	PD	LPI	ED	MN_PatchArea (m^2)	PLADJ	AI
Mean	41.4	0.86	25.7	28.1	64.9	79.6	81.8
Median	39.5	0.79	20.7	28.7	49.4	81.0	83.0
Minimum	15.5	0.24	4.7	17.8	13.3	62.9	68.6
Maximum	67.5	2.30	65.8	36.2	278.3	89.8	92.0
Std. Dev.	10.9	0.44	14.7	4.74	54.2	6.8	6.11

Riparian Area

Summary statistics for urban land cover in the 100-m riparian area are shown in Table 8 and for forest land cover in Table 9. Percent urban varied from 10.3 percent to 80.1 percent, with a mean of 37.5 percent. Percent forest varied from 19.9 to 82.2 percent, with a mean of 53.5 percent.

Table 9: Summary statistics for urban patterns (heavy, medium, light) within 100m of the study streams. PD = patch density, LPI = largest patch index, ED = edge density, PLADJ = percent like adjacencies, AI = aggregation index

	Percent	PD	LPI	MN_PatchArea (m^2)	PLADJ	AI
Mean	37.5	1.64	15.2	27.4	67.8	72.1
Median	36.8	1.62	6.9	22.6	70.7	72.0
Minimum	10.3	0.73	2.2	5.3	48.2	58.0
Maximum	80.1	3.16	80.1	109.5	77.0	85.3
Std. Dev.	14.9	0.51	19.2	22.1	8.1	6.8

Table 10: Summary statistics for forest patterns within 100m of stream. PD = patch density, LPI = largest patch index, ED = edge density, AI = aggregation index

	Percent	PD	LPI	MN_PatchArea (m^2)	AI
Mean	53.5	1.19	28.7	69.3	80.4
Median	52.7	1.06	19.3	48.3	80.6
Min	19.9	0.26	2.59	5.4	53.3
Max	82.2	3.66	82.2	310.0	92.4
Std. Dev.	15.8	0.72	23.5	71.5	7.2

3.3 Correlation Results

BIBI score was not correlated with any of the landscape metrics computed for this study. An analysis of the metrics that comprise the BIBI score found that one of these metrics, the median intolerant taxa richness for the three study years was related to some landscape metrics. The results for those correlations are below.

Both the Pearson correlation and Spearman's Rank Correlation found significant correlations between intolerant taxa richness and several landscape metrics (Tables 10 and 11). The Pearson correlation found that percent like adjacencies of urban land (including light urban) was negatively correlated with intolerant taxa richness at a significance level of 0.10. Urban patch density and riparian forest patch density were positively correlated at a significance level of 0.05.

Table 11: Pearson correlation coefficients (r) for factors with significant relationships to intolerant taxa richness. Bold = significant at 0.05; Italics = significant at 0.10. Empty cells = no significance. HML = heavy, medium, and light urban land; PD = patch density; PLADJ = percent like adjacencies; rip = riparian, for = forest; area = area of the basin

	Intolerant	HML_pd	HML_pladj	Crossings	rip_for_pd	outfalls	log_area
Intolerant		**0.564**	*-0.366*		**0.437**		**-0.426**
HML_pd			**-0.792**		**0.518**		**-0.541**
HML_pladj				*0.412*			**0.445**
Crossings					*0.363*		
rip_for_pd							**-0.478**
outfalls							**0.534**
log_area							

Table 12: Spearman's rank correlation coefficients (rs) for factors with significant relationships to intolerant taxa richness. Bold = significant at 0.05; Italics = significant at 0.10. Empty cells = no significance. HML = heavy, medium, and light urban land; PD = patch density; PLADJ = percent like adjacencies; rip = riparian, for = forest; area = area of the basin

	Intolerant	HML_pd	HML_pladj	Crossings	rip_for_pd	outfalls	log_area
Intolerant		*0.375*		*-0.415*		**-0.497**	**-0.430**
HML_pd			**-0.798**				**-0.484**
HML_pladj							
Crossings							
rip_for_pd							**-0.427**
outfalls							**0.776**
log_area							

Several of these factors showed covariation. Urban patch density and percent like adjacencies are strongly correlated ($r > 0.7$), likely because they are both measures of aggregation. Urban patch density was moderately correlated with riparian forest patch density and basin area. Number of road crossings was moderately correlated with urban percent like adjacencies and patch density of riparian forest. All factors were moderately correlated with basin area, except for road crossings. However, plots of basin area and these factors suggest that these correlations are not meaningful (see Figure 9 for an example). It should be noted that two basins were removed from the stormwater analysis because of unreliable data for stormwater facilities for the area.

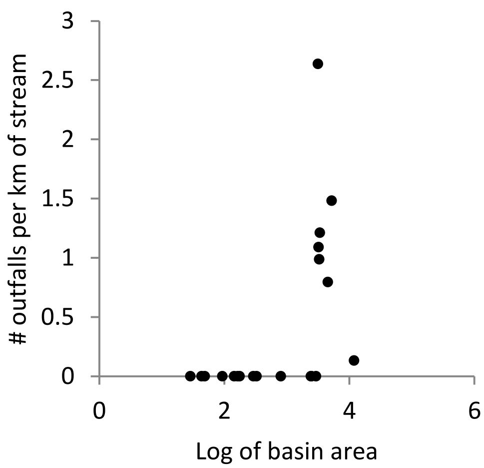

Figure 9: Number of stormwater outfalls per 10 kilometers of stream against area of the drainage basin. Although there is a statistically significant correlation between these variables, the shape of the plot suggests this correlation is not meaningful.

The Spearman's Rank correlation found that number of stormwater outfalls was negatively correlated with intolerant richness at a significance level of 0.05. At a significance level of 0.10, urban patch density and number of road crossings are also correlated with intolerant richness (positively and negatively, respectively). Again, several factors are correlated with basin area, but these correlations appear to be physically meaningless.

Figure 10 shows plots of these metrics with intolerant species richness. Although statistically significant, the plots exhibit substantial scatter and no clear trends. The plot of urban percent like adjacencies suggest that the correlations are being driven largely by a single stream with a low percent like adjacencies value. When this point is removed, the correlation reverses directions (although it is not significant). The plot of stormwater outfalls versus intolerant richness shows that with no stormwater outfalls along the stream, there is considerable variability in richness, but as outfalls are added, richness drops off quickly.

Figure 10c shows a stream with a higher number of road crossings and also a high intolerant taxa richness, which contradicts the general trend. However, this stream has a high urban patch density, which may explain why its intolerant richness remains so high. Looking at Figure 10a, there is another stream with lower urban patch density, but high intolerant richness – this stream has very few road crossings.

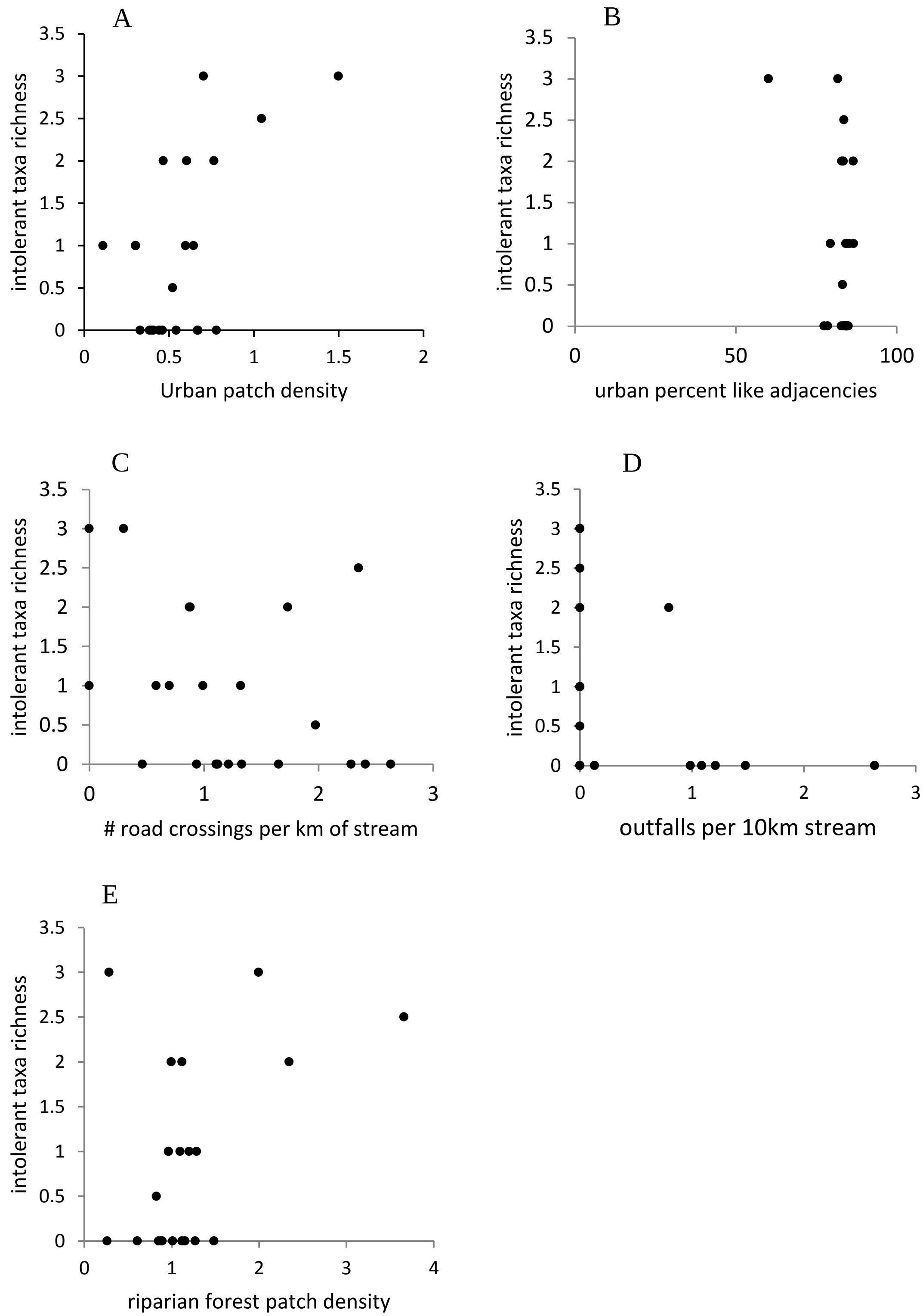

Figure 10: Plots of variables found to be significantly correlated with intolerant taxa richness. (A) Urban patch density (B) Urban percent like adjacencies (C) Number of road

crossings per kilometer of stream (D) Number of outfalls per 10 kilometers of stream and (E) Patch density of riparian forest

3.4 Analysis of extreme-scoring streams

Channel dimensions

Bankfull thalweg depth of the eight extreme-scoring streams ranged from 0.54 m to 2.12 m (Table 12). Bankfull width ranged from 2.73 meters to 20.35 m. Predicted bankfull discharge ranged from 0.44 $m^3 s^{-1}$ to 26.40 $m^3 s^{-1}$. There was no difference in width to depth ratio between the high-scoring and low-scoring streams.

Table 13: Channel dimensions and predicted discharge of extreme-scoring streams.

BIBI score	Name	Bankfull width (m)	Bankfull depth (m)	Q (cms)
High	Unnamed	3.04	0.48	0.55
	Cottage Lake	6.77	0.91	2.79
	Covington	8.7	0.54	4.66
	Big Soos	20.35	2.12	26.40
Low	Struve	2.73	0.45	0.44
	Little Bear	5.05	0.57	1.54
	Evans	6.74	0.71	2.77
	Bear	16.16	1.48	16.49

Sediment size

The results from the modified Wolman Pebble Count are shown in Figure 11. In general, cumulative pebble size distributions for high-scoring streams are shifted to the right of similarly-sized low-scoring streams. This suggests that the high-scoring streams have larger sediment sizes than low-scoring streams. An analysis of different size percentiles found some significant differences at an alpha of 0.10. Figure 12 shows the pebble size at different percentiles for each stream. D10, D25, and D50 were significantly larger in high-scoring streams than in low-scoring streams ($p = 0.08$, 0.07, and 0.09, respectively). Percent fines (< 2mm) was significantly higher in the low-scoring streams than in the high-scoring streams ($p = 0.04$) (Figure 13).

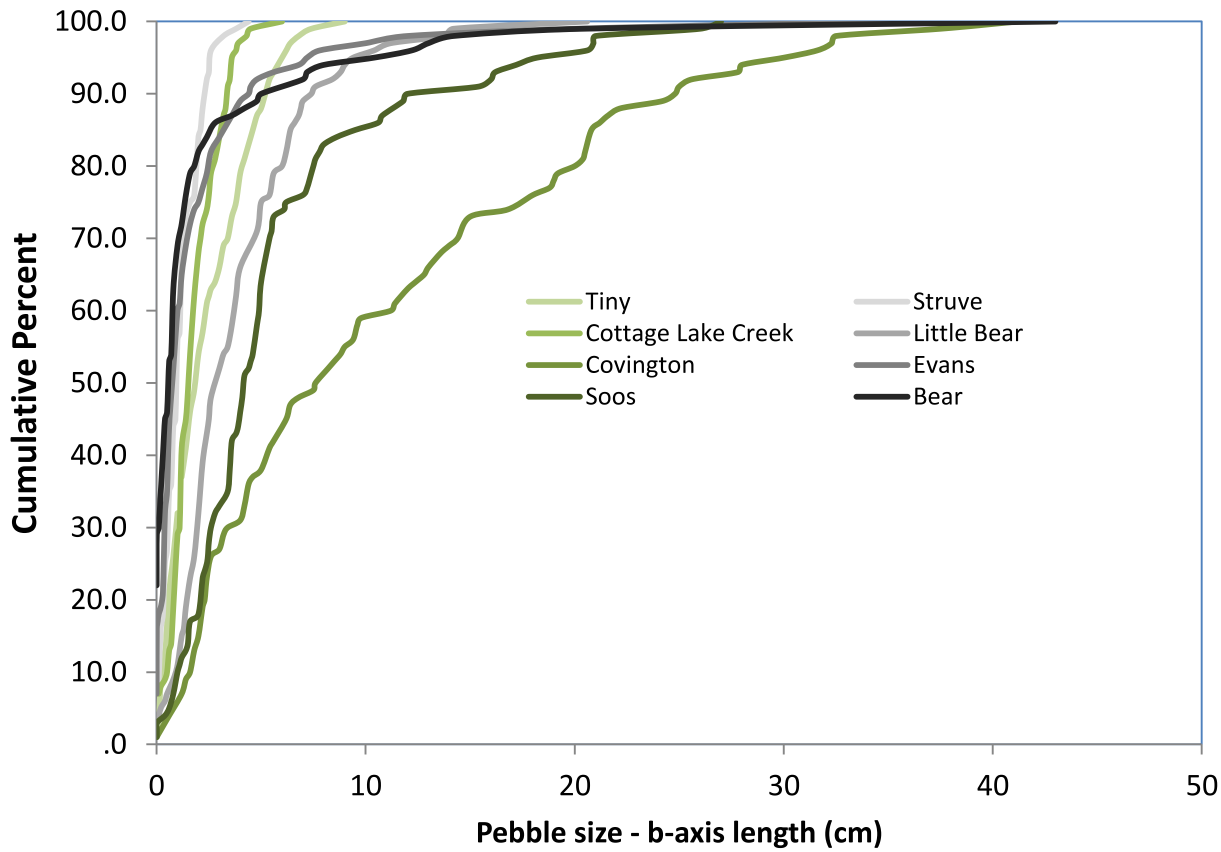

Figure 11: Cumulative percent of pebble size streams with extreme BIBI scores. Green = streams with high BIBI scores; grey = streams with low BIBI scores. Darker shades indicate larger streams.

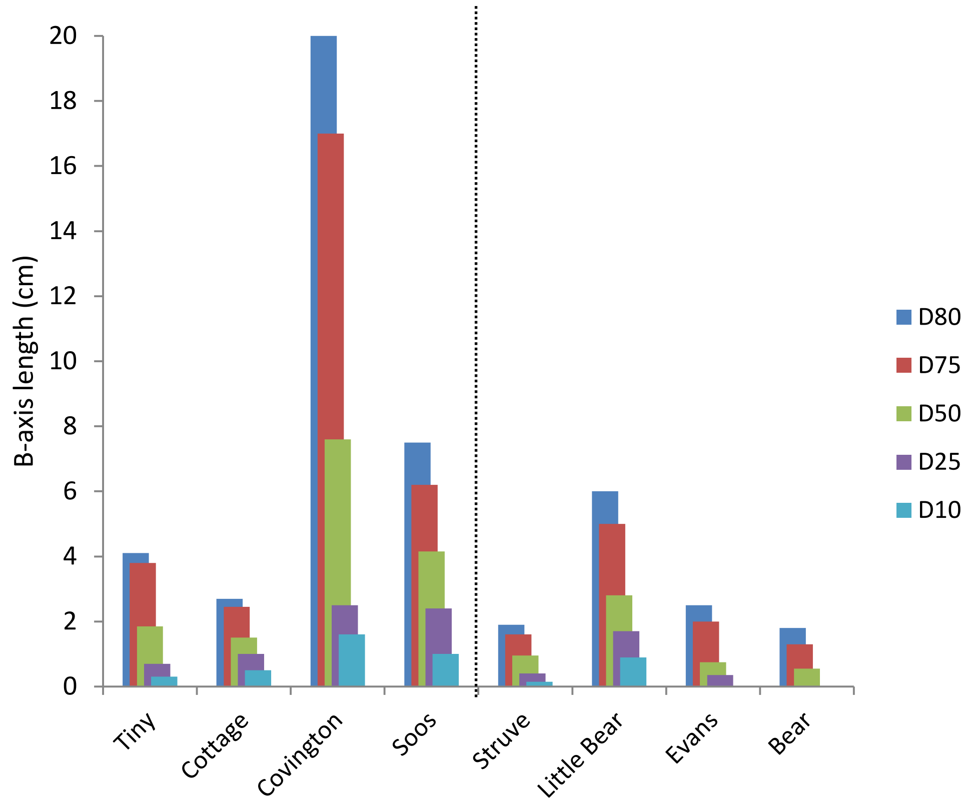

Figure 12: D10, D25, D50, D75 and D80 particles sizes in streams with extreme BIBI scores. Streams on the left half have high BIBI scores; streams on the right have low BIBI scores. The difference between the two groups is significant for D10, D25, and D50.

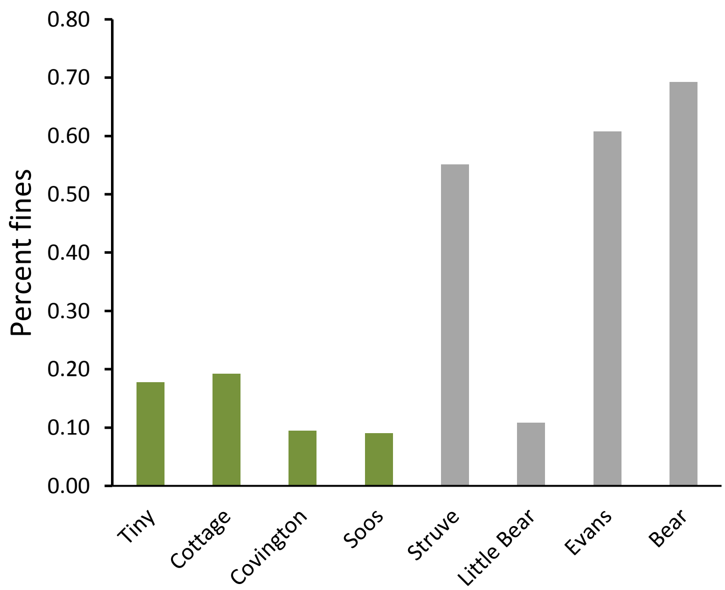

Figure 13: Percent fines (<2mm) in samples from each of the streams with extreme BIBI scores. Green = high-scoring streams, grey = low-scoring streams. The difference between the two groups is significant.

Large woody debris

The number of LWD per 100 m of stream reach varied from 0 to 35 per 100 m (Figure 14). Although the mean was slightly higher for the high-scoring streams (18.7 pieces per 100 m versus 16.6 pieces per 100m), the difference was not significant.

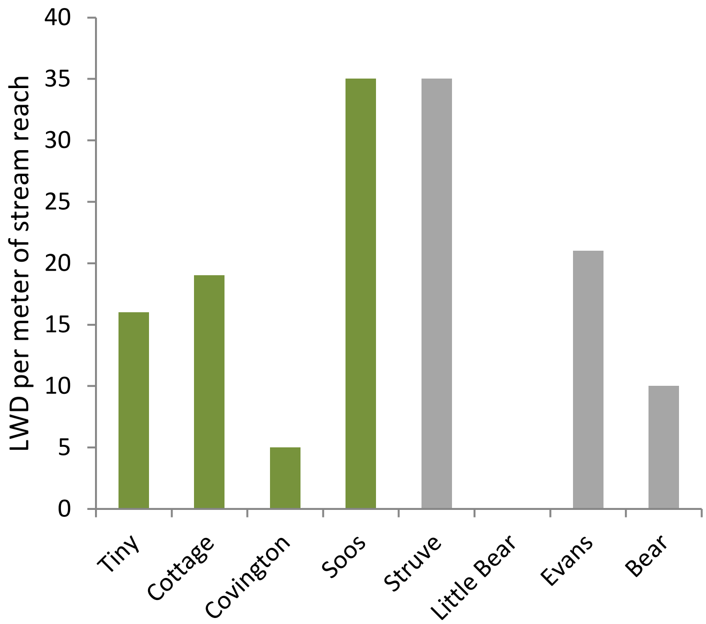

Figure 14: Number of large woody debris (LWD) per meter of stream reach in streams with extreme BIBI scores. Green = high BIBI scores; grey = low BIBI scores. The difference between the two groups is not significant.

Urban pattern analysis

Comparisons between urban patterns for extreme-scoring streams found only one metric to be significantly different between the two groups of streams. Mean number of road crossings per kilometer of stream were lower in the low-scoring streams than in the high-scoring streams ($p = 0.06$, one-tailed) (Figure 15). Number of outfalls also showed a difference with high-scoring streams generally having few to no outfalls, and the low-scoring streams having between 0.1 and 1.5 outfalls per 10km (Figure 16). Big Soos Creek was excluded from this analysis because of unreliable outfall data.

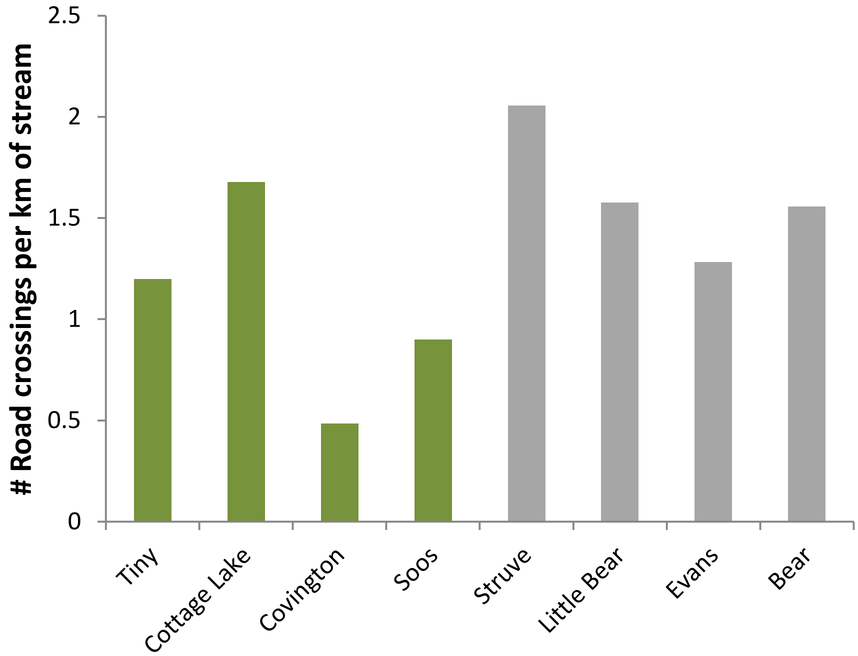

Figure 15: Number of road crossings per kilometer of stream in streams with extreme BIBI scores. Green = high BIBI score; grey = low BIBI score. The difference between the two groups is significant.

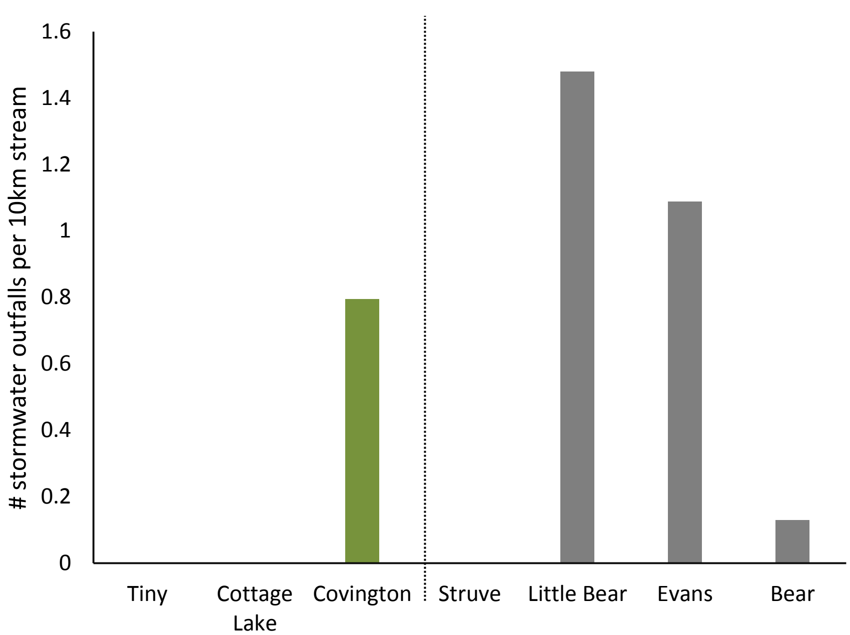

Figure 16: Number of stormwater outfalls per 10km of stream in streams with extreme BIBI scores. Left side of line = high BIBI score; right side = low BIBI score

4. Discussion

4.1 Urban patterns and stream health

Correlation and regression analysis did not identify any relationships between the BIBI score and the landscape metrics tested. Previous studies have found that the BIBI score does decline with increasing urbanization, and is related to many of these landscape metrics (Morley 2000, Alberti et al 2007). However, in this study, among basins of similar urbanization, no metrics could explain any of the variability in the BIBI scores. Possibly the values of the BIBI scores in this study are within the natural range of variation of the index. Although the index varies from 10 to 50 with values at every integer in between, the index may not be able to provide information on such a fine resolution. Instead, it may be better-suited to providing a coarse description of stream health when differences in urbanization levels among basins are larger.

Although the BIBI score proved to be a poor indicator for stream health in this study, some of the metrics that comprise the index may be useful – in particular the intolerant taxa richness. This metric was slightly related to several of the landscape metrics calculated in this study. Whether a taxon is considered intolerant to disturbance is based on a taxa attribute list generated from Wisseman (1998) (for a list of the intolerant taxa, see Appendix A). Intolerant taxa are the first to disappear following a disturbance to the stream and thus may be considered useful as a sensitivity index to that type of disturbance. The metrics that were most related to intolerant taxa richness were patch density of urban land (heavy, medium, light), the number of road crossings, and number of stormwater outfalls.

Because urban patch density is normally distributed, the Pearson correlation coefficient was used, which showed the patch density was positively related to intolerant taxa richness, although weakly. Patch density is the number of patches divided by the area of the basin – the more dispersed a given area of urban land is in a drainage basin, the higher the patch density. There is also in a general increase in number of intolerant taxa associated with this. A possibily mechanism for this relationship is that undeveloped, vegetated land interspersed among the developed land helps slow rain runoff and limit the changes in hydrology typically associated with urbanization. Other studies have found negative relationships between BIBI score and measures of urban aggregation, supporting the conclusion that dispersal/aggregation of urban land is related to macroinvertebrate communities (Alberti et al. 2007).

Interestingly, the patch density of urban land (or any metric) when excluding light urban from consideration had no relationship with intolerant taxa richness. The inclusion of light urban as urban land cover *did* lead to such a relationship. Light urban occurs in less densely developed areas and is more associated with single-family residential, suburban areas. These low density residential areas may not behave that differently from higher density, urban areas, in a hydrological sense. Gregory (2004) found infiltration rates on residential lots to decrease by 80 to 97 percent from pre- to post-development. Woltemade (2010) found that residential lots developed within the past decade had significantly slower infiltration rates than agricultural lots of similar soils (older residential lots were not different). This, in conjunction with the results of this study, suggest that light urban land that is primarily low density residential lots behaves like medium and heavy urban land in a hydrologic sense and should be consider “urban” when conducting stream health studies.

The number of road crossings was not normally distributed and thus the Spearman’s rank correlation coefficient was used. Road crossings were negatively, although weakly, related to intolerant taxa richness. The streams with the four lowest BIBI scores had significantly more road crossings than the streams with the four highest BIBI scores lending support to the existence of a relationship between macroinvertebrate communities and road crossings, although this relationship is not strong. Other studies have found negative relationships between road crossings and BIBI score (Alberti et al. 2007). This relationship may exist because road crossings appear to alter the physical conditions (such as channel dimensions, LWD abundance, bank stability, structural complexity, embeddedness, and cementation) immediately downstream, with some variation with different types of crossings (McBride and Booth 2005). If there is indeed an impact to stream physical conditions from road crossings, this may in turn impact habitat for macroinvertebrate, and eventually alter their community composition. One possibility for the weakness of the relationship in this study is that these impacts may only occur for a limited distance downstream from the road crossing. The macroinvertebrate community at a certain point may not experience any influence from road crossings that are farther than a certain distance away.

The interaction of patch density and road crossings may be relevant, too. There was one stream that had a high number of road crossing and a high intolerant richness, but the stream also had a high urban patch density. Possibly there is an interaction between these variables and this may be a useful direction for further research.

Intolerant taxa richness also appears to be related to the number of stormwater outfalls upstream of the sampling point (because this metric was not normally distributed, Spearman's rank coefficient was used). Traditional stormwater infrastructure collects rain water and conveys it quickly to streams through pipes and eventually outfalls. As a result, water is not cleaned or filtered by the soil, and it reaches streams quickly, leading to higher peak flows than a natural stream (Booth and Jackson 1997). Some outfalls have detention ponds that hold and slow down the water's entry into the stream, but they do not treat the water quality, and only partially mitigate the increased peak flows. The result is a reduction in habitat quality for stream biota (Booth and Jackson 1997, McBride and Booth 2005).

Several metrics were notably absent from any of the correlations or models generated. None of the metrics for forest cover in the basins (such as percent forest or aggregation index of forest) were related to intolerant taxa richness. This is counterintuitive as forest cover is generally expected to maintain more natural hydrological processes and sediment regimes (Arnold and Gibbons 1996). This may be because there is not enough variation in quantity of forest land cover among the basins to detect a trend. This is likely a result of the basins being selected to have similar levels of urban development, thereby limiting the variability of other land cover classes.

4.2 Riparian Area Effects

Land cover in the riparian area was not found to be related to intolerant taxa richness (or BIBI score), even though previous studies of invertebrates have found such a relationship (Moore and Palmer 2005, Alberti et al. 2007). However, a growing body of literature is suggesting that riparian areas may not have as much influence on stream conditions as previously thought, particularly in urban catchments where altered hydrology overwhelms any benefits from riparian forests (Walsh et al. 2007). Groffman et al. (2002) found that water in an urban basin was bypassing the riparian soil, which affected the cycling of nitrogen in the stream. Roy et al. (2005) found no difference in macroinvertebrate communities between open-canopy and closed-canopy reaches, and in fact found higher dissolved oxygen in open-canopy reaches, likely due to increased photosynthesis (contradicting the hypothesis that increased light would increase temperatures and thus decrease dissolved oxygen). Walsh et al. (2007) found total impervious area to be a much stronger correlate with macroinvertebrate assemblage composition than riparian forest or canopy cover. Imberger et al. (2011) found that the increased input of coarse particulate organic matter (CPOM) (potential food for invertebrates) in urban stream reaches with riparian forest was negated by decreased retention and storage of CPOM caused by altered hydrological processes. Ultimately, riparian area forest may not be able to compensate for urbanized drainage basins until actions to improve natural drainage processes are implemented.

4.3 Analysis of extreme-scoring streams

Some metrics differ between the high-scoring and low-scoring streams, but others did not. Only one of the landscape metrics computed showed a difference between the two sets of streams, including those that showed significant relationships with intolerant taxa richness. This is likely because the high-scoring streams do not necessarily have high intolerant taxa richness. In fact, when ordering streams by intolerant taxa richness, a new set of streams is at the top while the low-scoring streams are still at the bottom with mostly zero intolerant taxa.

High-scoring streams had significantly fewer road crossings than low-scoring streams, which provides further evidence that there is a relationship between road crossings and stream macroinvertebrate communities. There were also fewer stormwater outfalls in the high-scoring streams, supporting the idea that stormwater infrastructure is also altering these communities.

Among the habitat variables sampled at the extreme-scoring streams, only sediment size showed a difference between high- and low-scoring streams. In general, the high-scoring streams seem to have larger sediment. Percent fines was the strongest difference, with low-scoring streams having significantly higher percent fines. D10, D25, and D50 were significantly smaller in low-scoring streams than in high-scoring streams, but not D75 or D80. This suggests that the biggest 20-25 percent of cobbles in these streams were about the same size, but the smallest 50 percent were smaller in low-scoring streams. It is at these smaller sizes that invertebrate's ability to interact with the sediment changes (Minshall 1984). This finding supports the notion that sediment size is important for benthic communities.

Possibly another factor, such as changes in hydrology, may be the actual cause of the difference in BIBI score and sediment size, with the latter two variables only being correlated. However, other studies have also found relationships between sediment size and measures of macroinvertebrate communities. Zweig and Rabeni (2001) found that overall taxa richness, richness of EPT, overall macroinvertebrate density and density of EPT were all negatively correlated with the amount of deposited sediment (sediment that is less than 2mm). Morse et al. (2003) found increased numbers of taxa tolerant to silt in streams with smaller sediment. Wagenhoff et al. (2012) found declines in taxa richness and EPT density after experimental manipulation of sediment sizes, suggesting a causal relationship.

The time lag between sediment sampling and BIBI sampling (5 years) should be also be taken into account. The sediment in these streams has likely changed from when the BIBI was sampled. Larger streams would tend to have more power and be able to transport larger sediment than smaller streams. The sediment size at a given location may tend to increase more in larger streams over time, as sediment is transported. Alternatively, this sediment may be replaced with similarly sized sediment from further upstream, leading to no net change in sediment size, if hydrological regimes and erosion do not change much.

The lack of a statistical difference in LWD between high- and low-scoring streams was unexpected. Stewart et al. (2013) found that invertebrate species richness and an index of invertebrate communities were positively related to both number and volume of LWD. They found that LWD presence was associated with pool formation and changes in sediment, which possibly leads to changes and improvements in habitat structure for invertebrates. The reason no difference was detected here is likely because the sample size was too small given the amount of variability in LWD abundance. Another important factor is that LWD was tallied in 2012, whereas the BIBI scores were measured in 2006-2008. There may have been changes to the stream in the intervening years and 2012 LWD may not represent 2006-2008 LWD. Finally, LWD count may not explain the difference in BIBI score among streams of similar urbanization.

One thing worth noting is that Little Bear creek, a low scoring stream, had relatively large sediment but had no LWD (Figures 12 and 14). Conversely, Covington Creek, a high scoring stream had relatively little LWD, but extremely large sediment relative to the other high-scoring streams. Possibly there is a relationship between sediment size and LWD that is influencing macroinvertebrate communities.

4.4 Future Changes

Another concern that planners must consider is future changes in hydrology due to changing climate. In the Puget Sound region, temperatures are expected to rise, leading to more precipitation falling as rain and less as snow (Elsner et al 2010). Snowpack, which provides winter storage of water and maintains streamflows into the summer as it melts, will also decrease. This will likely lead to changes in the timing of streamflow peaks in streams and rivers fed by snowpack. Cuo et al. (2011) found that streams in the Puget Sound lowlands (below 500m elevation), such as the streams in the current study, are less sensitive to such changes because they are generally rain-dominated, not snow-dominated. These streams were found to have a modest increase in winter runoff due to increased precipitation. Instead, land cover change will be a large factor in hydrological change in these basins with increased impervious surfaces generating increased runoff throughout the year.

4.5 Implications and Applications

The results of this study imply that BIBI score might not be able to provide information about a stream's health at a fine resolution, and instead is better used to place streams into broad categories of health (i.e. poor, medium, and good). Local governments with macroinvertebrate monitoring programs should consider maintaining raw invertebrate data to use in analyzing differences among local streams and not rely entirely on the computed BIBI score. This will also allow water quality planners and managers to better understand why a stream's health is poor, particularly if certain functional groups are missing from the macroinvertebrate community in that stream. Planners should also include other stream health variables when assessing the condition of local streams.

The positive relationship between patch density of urban land and intolerant taxa richness suggests that developing land in smaller separated chunks may lead to less disturbance in streams than developing in larger consolidated lots (given that the area of development is to remain the same). One way local land use planners may be able to achieve such fragmentation in development is through the use of cluster development. Cluster development sites houses on smaller lots, clustered together, which leaves a large portion of the property undeveloped (Church 2013). Cluster developments also have other benefits, such as wildlife habitat protection, open space for community recreation, and generally reduced costs of development.

The finding that light urban (i.e. single family residential areas) likely behaves similarly to medium and high urban has important implications. Many fewer people can be housed in low density single family residential areas, so the area of land encompassed by this land use is necessarily larger than the area of land needed for high density single family and multi-family land uses of comparable population size. If this land behaves similarly to higher density land uses, land use planner should consider reducing its usage in order to minimize the area of land covered in stream-damaging uses. Again, cluster developments, as mentioned above, may be one technique for achieving this. Another is focusing growth in designated urban growth areas, something already required in Washington by the Growth Management Act.

The results of this study as well as others mentioned above suggest that road crossings matter for stream health. They alter the physical stream structure downstream from the crossing, which appears to alter the biologic communities in the stream. Local governments should consider the location of streams when planning new roads to minimize the number of crossings and use designs that have less impact on streams when crossings must occur.

The negative correlation between intolerant taxa richness and stormwater outfalls provides further evidence that traditional stormwater infrastructure is having a detrimental impact on streams, with sensitive taxa disappearing with even one outfall into a stream. Local governments should begin to emphasize the use of low impact development techniques and green stormwater infrastructure in place of traditional grey infrastructure. Results from several "Street Edge Alternative Streets" in Seattle, Washington show that such techniques can dramatically reduce rain runoff volumes and pollutant concentration (Horner, Lim and Burges 2004, Horner and Chapman 2007).

Maintaining complexity in streams, also appears to be important. The four streams with the highest BIBI scores in this study had significantly larger sediment than the four streams with the lowest BIBI scores, suggesting that changes in sediment size may be another driver in changes to macroinvertebrate communities following urbanization. There was also generally more LWD in high-scoring streams (although the difference was not significant). This increased complexity can provide more habitat options for varying invertebrate taxa, leading to healthier invertebrate communities. Achieving this complexity is tricky because sediment sizes cannot be directly changed. Instead maintaining a more natural hydrological regime through the use of low impact development techniques (as mentioned above) may be a way to approach this.

Finally, the finding that riparian metrics are unrelated to macroinvertebrate community health suggests that current policies of limiting development in riparian buffers may not be able to protect rivers and streams in urban areas as well as previously thought. Planners may consider basin-wide approaches to maintaining natural drainage processes such as those mentioned above instead.

4.6 Complications with the Data

Drainage Basin Delineation A few of the drainage basins delineated by ArcHydro did not match perfectly with the existing stream data, with either the exclusion of land that is actually part of the drainage basin or the inclusion of land that does not belong to the drainage basin. This is likely because of inaccuracies in the DEM used to delineate the basins and manmade drainage changes. These errors are assumed to have occurred randomly, with an overall negligible effect on basin land cover composition.

Land Cover Composition The land cover classification used in this study’s analysis is different from the land cover dataset used by King County, so after basins were identified for this study, land cover composition was found to differ somewhat from the record in King County’s database. While the aim was to identify basins with similar levels of urbanization, there was more variability in quantity of urban land cover than expected. However, the similar levels of impervious surfaces among the basins supports the idea that the basins are mostly similar in quantity of urbanization.

Stormwater Data Data on locations of stormwater facilities were obtained from local municipalities and the accuracy of these data is limited and varies from city to city. These datasets were created for other purposes, and so can only provide limited information for this study, such as the identification of any possible trends that should be studied further.

5. Conclusions and Recommendations

5.1 Major Findings

Urban streams are declining in health across the country. This study provides the first assessment of stream health among streams that are similarly urbanized, which allows for identification of factors that may lessen the effect of urban development on streams. Evidence here points to several such factors in moderately urbanized drainage basins (approximately 13 to 15 percent imperviousness) in the Pacific Northwest. The fragmentation of urban land is positively related to the richness of sensitive macroinvertebrate taxa, suggesting that spatial arrangement of urban land is a consideration for local policies. Conversely, infrastructure intensity is negatively related to sensitive taxa richness. The layout of roads should be planned carefully to minimize crossings of streams where possible. Finally, traditional stormwater infrastructure has a considerable impact on streams, and implementation of green stormwater infrastructure that mimics natural drainage processes should become the norm in local drainage utilities instead of the exception.

Interestingly, land cover within the riparian zone did not appear to play a role in macroinvertebrate community health in this study, despite the fact that many local governments in the Puget Sound area rely on buffers to protect streams. Growing evidence in the literature suggests that riparian forests in urbanized basins are unable to mitigate the drastically altered hydrology. However, riparian forests likely provide other benefits (such as wildlife habitat and recreational opportunities), so policies protecting these areas should not be retracted. Instead, they should be augmented with policies that maintain more natural hydrological regimes, such as implementation of green stormwater infrastructure and low impact development techniques.

Finally, the finding that BIBI score was unrelated to any metrics explored in the study suggests that local governments should use such indices with caution, particularly when setting policy. Instead, the BIBI should be used with a suite of environmental monitoring data to understand the condition of local streams.

5.2 Future Research Directions

Future studies should also investigate how BIBI scores and macroinvertebrate communities in general change over time. The streams in this study showed variation in score from year to year, so understanding how the score varies, and what potential climatic and land cover change factors influence this will help future researchers and managers better interpret BIBI scores.

This study found that road crossings were related to intolerant taxa richness but did not look at different types of road crossings. Knowing how crossing type affects stream health and what potential designs can minimize the impact to the stream would be beneficial for managers and planners and is an important direction for future research. Research on the ability to restore stream crossings to less impactful designs would also be useful for local planners interested in improving already degraded streams.

Finally, the use of low impact development techniques has been touted as a way to reduce the effect of development on natural hydrological processes (Church 2013). Quantifying the impacts of such techniques on stream health will be crucial to understand just how much they mitigate the effects of development and which techniques should be prioritized.

Ultimately, the research presented here identifies several factors in urban landscapes that contribute to stream health. Urban planners can use this list as a starting point for better understanding the ecology of their watersheds when making land use and infrastructure decisions.

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Appendix A: List of Invertebrate Taxa Considered 'Intolerant'

Phylum	Class	Order	Family	Subfamily	Genus	Species
Arthropoda	Insecta	Diptera	Empididae	Oreogetoninae	Oreogeton	
Arthropoda	Insecta	Diptera	Pelecorhynchidae			
Arthropoda	Insecta	Diptera	Pelecorhynchidae		Glutops	
Arthropoda	Insecta	Diptera	Blephariceridae			
Arthropoda	Insecta	Diptera	Blephariceridae	Blepharicerinae	Agathon	
Arthropoda	Insecta	Diptera	Blephariceridae	Blepharicerinae	Agathon	arizonica
Arthropoda	Insecta	Diptera	Blephariceridae	Blepharicerinae	Bibiocephala	
Arthropoda	Insecta	Diptera	Blephariceridae	Blepharicerinae	Blepharicera	
Arthropoda	Insecta	Diptera	Blephariceridae	Blepharicerinae	Dioptopsis	
Arthropoda	Insecta	Diptera	Blephariceridae	Blepharicerinae	Philorus	
Arthropoda	Insecta	Diptera	Deuterophlebiidae			
Arthropoda	Insecta	Diptera	Deuterophlebiidae		Deuterophlebia	
Arthropoda	Insecta	Diptera	Chironomidae	Chironominae	Xenochironomus	

Arthropoda	Insecta	Diptera	Chironomidae	Chironominae	Stempellina	
Arthropoda	Insecta	Diptera	Chironomidae	Diamesinae	Potthastia	
Arthropoda	Insecta	Diptera	Chironomidae	Diamesinae	Pseudokiefferiella	
Arthropoda	Insecta	Diptera	Chironomidae	Diamesinae	Sympotthastia	
Arthropoda	Insecta	Diptera	Chironomidae	Orthocladiinae	Bryophaenocladius	
Arthropoda	Insecta	Diptera	Chironomidae	Orthocladiinae	Cricotopus	
Arthropoda	Insecta	Diptera	Chironomidae	Orthocladiinae	Heterotrissocladius	
Arthropoda	Insecta	Diptera	Chironomidae	Orthocladiinae	Krenosmittia	
Arthropoda	Insecta	Diptera	Chironomidae	Orthocladiinae	Parachaetocladius	
Arthropoda	Insecta	Diptera	Chironomidae	Orthocladiinae	Pseudorthocladius	
Arthropoda	Insecta	Diptera	Thaumaleidae			
Arthropoda	Insecta	Diptera	Tanyderidae			
Arthropoda	Insecta	Diptera	Tanyderidae		Protanyderus	
Arthropoda	Insecta	Diptera	Tipulidae	Limoniinae	Hesperoconopa	
Arthropoda	Insecta	Diptera	Tipulidae	Limoniinae	Rhabdomastix	
Arthropoda	Insecta	Plecoptera	Leuctridae			
Arthropoda	Insecta	Plecoptera	Leuctridae	Leuctrinae	Despaxia	
Arthropoda	Insecta	Plecoptera	Leuctridae	Leuctrinae	Leuctra	
Arthropoda	Insecta	Plecoptera	Leuctridae	Leuctrinae	Moselia	infuscata
Arthropoda	Insecta	Plecoptera	Leuctridae	Leuctrinae	Paraleuctra	
Arthropoda	Insecta	Plecoptera	Leuctridae	Leuctrinae	Perlomyia	
Arthropoda	Insecta	Plecoptera	Leuctridae	Megaleuctrinae	Megaleuctra	
Arthropoda	Insecta	Plecoptera	Nemouridae	Nemourinae	Visoka	cataractae
Arthropoda	Insecta	Plecoptera	Nemouridae	Nemourinae	Zapada	columbiana
Arthropoda	Insecta	Plecoptera	Nemouridae	Nemourinae	Zapada	frigida
Arthropoda	Insecta	Plecoptera	Chloroperlidae	Paraperlinae	Kathroperla	perdita
Arthropoda	Insecta	Plecoptera	Chloroperlidae	Paraperlinae	Paraperla	
Arthropoda	Insecta	Plecoptera	Peltoperlidae	Peltoperlinae	Sierraperla	
Arthropoda	Insecta	Plecoptera	Peltoperlidae	Peltoperlinae	Soliperla	
Arthropoda	Insecta	Plecoptera	Peltoperlidae	Peltoperlinae	Yoraperla	
Arthropoda	Insecta	Plecoptera	Peltoperlidae	Peltoperlinae	Yoraperla	brevis
Arthropoda	Insecta	Plecoptera	Peltoperlidae	Peltoperlinae	Yoraperla	mariana
Arthropoda	Insecta	Plecoptera	Perlidae	Acroneuriinae	Doroneuria	
Arthropoda	Insecta	Plecoptera	Perlodidae	Perlodinae	Frisonia	picticeps
Arthropoda	Insecta	Plecoptera	Perlodidae	Perlodinae	Megarcys	
Arthropoda	Insecta	Plecoptera	Perlodidae	Perlodinae	Salmoperla	
Arthropoda	Insecta	Plecoptera	Perlodidae	Perlodinae	Setvena	
Arthropoda	Insecta	Plecoptera	Perlodidae	Perlodinae	Cultus	
Arthropoda	Insecta	Plecoptera	Perlodidae	Perlodinae	Kogotus	
Arthropoda	Insecta	Plecoptera	Perlodidae	Perlodinae	Osobenus	yakimae
Arthropoda	Insecta	Plecoptera	Perlodidae	Perlodinae	Pictetiella	expansa

Arthropoda	Insecta	Plecoptera	Perlodidae	Perlodinae	Rickera	sorpta
Arthropoda	Insecta	Plecoptera	Perlodidae	Perlodinae	Chernokrilus	
Arthropoda	Insecta	Plecoptera	Perlodidae	Perlodinae	Diura	
Arthropoda	Insecta	Plecoptera	Pteronarcyidae	Pteronarcyinae	Pteronarcys	princeps
Arthropoda	Insecta	Trichoptera	Glossosomatidae	Glossosomatinae	Anagapetus	
Arthropoda	Insecta	Trichoptera	Hydropsychidae	Arctopsychinae	Parapsyche	elsis
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Ptilocolepinae	Palaeagapetus	
Arthropoda	Insecta	Trichoptera	Apataniidae		Allomyia	
Arthropoda	Insecta	Trichoptera	Apataniidae		Pedomoecus	sierra
Arthropoda	Insecta	Trichoptera	Apataniidae	Apataniinae	Apatania	
Arthropoda	Insecta	Trichoptera	Goeridae	Goerinae	Goeracea	genota
Arthropoda	Insecta	Trichoptera	Limnephilidae	Dicosmoecinae	Allocosmoecus	partitus
Arthropoda	Insecta	Trichoptera	Limnephilidae	Dicosmoecinae	Cryptochia	
Arthropoda	Insecta	Trichoptera	Limnephilidae	Dicosmoecinae	Dicosmoecus	atripes
Arthropoda	Insecta	Trichoptera	Limnephilidae	Dicosmoecinae	Ecclisocosmoecus	scylla
Arthropoda	Insecta	Trichoptera	Limnephilidae	Dicosmoecinae	Ecclisomyia	
Arthropoda	Insecta	Trichoptera	Limnephilidae	Dicosmoecinae	Eocosmoecus	frontalis
Arthropoda	Insecta	Trichoptera	Limnephilidae	Dicosmoecinae	Eocosmoecus	schmidi
Arthropoda	Insecta	Trichoptera	Limnephilidae	Limnephilinae	Desmona	bethula
Arthropoda	Insecta	Trichoptera	Limnephilidae	Limnephilinae	Desmona	mono
Arthropoda	Insecta	Trichoptera	Limnephilidae	Limnephilinae	Homophylax	
Arthropoda	Insecta	Trichoptera	Limnephilidae	Limnephilinae	Halesochila	taylori
Arthropoda	Insecta	Trichoptera	Limnephilidae	Limnephilinae	Chyrandra	centralis
Arthropoda	Insecta	Trichoptera	Limnephilidae	Limnephilinae	Philocasca	
Arthropoda	Insecta	Trichoptera	Limnephilidae	Pseudostenophylacinae	Pseudostenophylax	edwardsi
Arthropoda	Insecta	Trichoptera	Uenoidae	Thremmatinae	Neophylax	occidentis
Arthropoda	Insecta	Trichoptera	Uenoidae	Thremmatinae	Oligophlebodes	
Arthropoda	Insecta	Trichoptera	Uenoidae	Uenoinae	Farula	
Arthropoda	Insecta	Trichoptera	Uenoidae	Uenoinae	Neothremma	
Arthropoda	Insecta	Trichoptera	Uenoidae	Uenoinae	Sericostriata	surdickae
Arthropoda	Insecta	Trichoptera	Philopotamidae	Philopotaminae	Dolophilodes	
Arthropoda	Insecta	Trichoptera	Phryganeidae	Yphriinae	Yphria	californica
Arthropoda	Insecta	Trichoptera	Rhyacophilidae		Himalopsyche	phryganea
Arthropoda	Insecta	Trichoptera	Rhyacophilidae		Rhyacophila	oreta
Arthropoda	Insecta	Trichoptera	Rhyacophilidae		Rhyacophila	verrula
Arthropoda	Insecta	Trichoptera	Rhyacophilidae		Rhyacophila	
Arthropoda	Insecta	Trichoptera	Rhyacophilidae		Rhyacophila	
Arthropoda	Insecta	Trichoptera	Rhyacophilidae		Rhyacophila	
Arthropoda	Insecta	Trichoptera	Rhyacophilidae		Rhyacophila	
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae		Caudatella	
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae		Caudatella	edmundsi

Arthropoda	Insecta	Ephemeroptera	Ephemerellidae		Caudatella	heterocaudata
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae		Caudatella	hystrix
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae		Drunella	doddsii
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae		Drunella	pelosa
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae		Drunella	spinifera
Arthropoda	Insecta	Ephemeroptera	Baetidae		Baetis	bicaudatus
Arthropoda	Insecta	Ephemeroptera	Heptageniidae		Cinygma	
Arthropoda	Insecta	Ephemeroptera	Heptageniidae		Epeorus	grandis

Appendix B: GIS Data Dictionary

File Name	Type	Description	Source	Cell Size
20121022_RentonStormwaterNetwork	geodatabase	Renton stormwater facilities	City of Renton	
DEM	DEM	Digital Elevation Model	National Resources Conservation Service	10 m
lc07_14cl_u	GRID	2007 Land Cover	Urban Ecology Laboratory, University og Washington	30 m
mv_SWM_040513	geodatabase	Maple Valley stormwater facilities	City of Maple Valley	
outfall_king	shapefile	Stormwater outfalls in King County	King County	
outfall_snoho	shapefile	Snohomish County stormwater outfalls	Snohomish County	
outfalls_BD	shapefile	Stormwater outfalls in Black Diamond	City of Black Diamond	

outfalls_Cov	shapefile	Stormwater outfalls in Covington	City of Covington	
row_king	shapefile	King County right-of-ways	King County	
row_snoho	shapefile	Snohomish County right-of-ways	Snohomish County	
stormwatersystem	geodatabase	Newcastle stormwater facilities	City of Newcastle	
streams	shapefile	Streams in King County	King County	
wa_2006_impervious	GRID	Impervious Surfaces	Washington Department of Ecology	30 m