The Role of Patterns of Urban Development in Stream Benthic Index of Biotic Integrity Score

Maria Sandercock

A thesis
submitted in partial fulfillment of the
requirements for the degree of

Master of Science
School of Environmental and Forest Sciences

University of Washington
2013

Chair: Daniel Vogt
Committee: Marina Alberti, Susan Bolton

Program Authorized to Offer Degree:
School of Environmental and Forest Sciences

University of Washington

Abstract

The Role of Patterns of Urban Development in Stream Benthic Index of Biotic Integrity Score

Maria Sarah Sandercock
Chair of the Supervisory Committee:
Associate Professor, Daniel Vogt
School of Environmental and Forest Sciences

The "Urban Stream Syndrome" describes the consistent, negative changes that streams in urbanized watersheds experience. Increasing numbers of people living in urban areas around the world imply increasing pressure on stream health. Understanding the mechanisms that cause stream degradation and the factors that can mitigate the impact of urbanization are more important than ever. Here, existing data on 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 or ~ 30% urban land cover) 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. Infrastructure intensity (road density, number of road crossings, and number of stormwater outfalls) was also correlated with these response variables. A subset of streams – the four highest and four lowest-scoring streams – were visited to collect additional habitat data (sediment size, large woody debris, and channels structure). No landscape metrics were correlated with BIBI score. However, patch density of urban land was weakly positively correlated with intolerant taxa richness, while number of road crossings and number of stormwater outfalls were weakly negatively correlated with intolerant taxa richness. No macroinvertebrate metrics were related to land cover in the riparian area. The variability in BIBI score remained unexplained by any metric in this study, and it is possible that BIBI scores in this

study were within the natural range of variation of the BIBI. This index may only be able to provide a coarse level of information on stream condition. The results of this study imply that more fragmented urban land is associated with higher numbers of intolerant taxa. Fewer road crossings and fewer stormwater outfalls are also associated with more intolerant taxa. Exploring these relationships further and with other measure of stream condition is recommended.

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Acknowledgements

I would first like to thank my advisor, Daniel Vogt, for his help, guidance, and support throughout this process. I would also like to thank my committee members, Susan Bolton and Marina Alberti for their advice on study design, help throughout the process, and review of this manuscript. I am also grateful to my field helper, Josefina Naravarja, who fearlessly trekked to every stream with me and provided top quality support for collecting data. My labmate, Stephen Gmur, provided much help in conducting the GIS analysis for this project, for which I'm extremely thankful. I would also like to thank Jo Wilhem from King County who met with me and helped me acquire the BIBI data used in in this study. Finally, I would like to thank my labmate, Lauren Grand, who read through my thesis and provided moral support during long days in the lab.

1. Introduction

The "Urban Stream Syndrome" describes the consistent, negative changes that streams in urbanized watersheds experience (Walsh et al. 2005). Because of increased impervious area, more stormwater runs off surfaces and less infiltrates the ground, leading to quicker and higher peak flows in streams (Paul and Meyer 2001, Walsh et al. 2005). When this occurs, channel morphologies can change through stream incision. Stormwater accumulates pollutants when it flows over urban surfaces and is not filtered by plants and soil microbes. This impairs water quality, with increases in pollutants, changes in nutrient cycling, and increases in temperature as a result. These physical and chemical changes in streams ultimately affect biological communities, such as fish and macroinvertebrates, which see decreases in species richness and loss of sensitive species.

At the same time, an increase of urban land cover is happening across the globe, with more and more of the world's population living in urban areas (United Nations 2011). Understanding the mechanisms that cause stream degradation and the factors that can mitigate the impact of urbanization is more important than ever. According to the Environmental Protection Agency's National Rivers and Streams Assessment (2013), 55% of river and stream miles in the United States do not support healthy biologic communities, with 7% fewer biologically healthy streams in 2009 than in 2004.

In urban areas, there are multiple stressors on aquatic ecosystems that are hard to distinguish (Walsh et al. 2005). As a result, there is still much about the relationship between watershed processes and stream health that we do not know. Notably, the degree of degradation in streams varies from stream to stream, even if they are similarly urbanized (Booth et al. 2004, Walsh et al. 2005, Alberti et al. 2007). Why some streams remain relatively healthy and others decline rapidly in the face of similar quantities of urbanization is not well known, but answering this is an important start to addressing urbanization impacts on streams. Evidence points to the patterns of land cover in the streams' drainage basins as playing a role in stream health (Alberti et al. 2007, Shandas and Alberti 2009). A better understanding of this role may provide some insight into strategies for minimizing urbanization's effects on streams. Identifying relationships between these factors and stream health can be tricky, and requires a good metric for measuring stream health.

Macroinvertebrate assemblages may fulfill this role as they have previously proven to be useful measures of stream condition in studies of stream disturbance due to anthropogenic drivers (Fore et al. 1996, Jones and Clark 1997, Walsh et al. 2001, Morley and Karr 2002, Roy et al. 2003, Moore and Palmer 2005, Walsh et al. 2005, Alberti et al. 2007, Brown et al. 2009, Li et al. 2012). The taxonomic and functional diversity of freshwater macroinvertebrates make them excellent biotic indicators - as watershed processes change in response to urbanization, species composition shifts and species intolerant to certain disturbances disappear while species tolerant to such disturbance increase in abundance.

The Benthic Index of Biotic Integrity (BIBI) was developed as a way to assess the health of streams, particularly with the degree of human impact to the stream in mind. The BIBI is a multimetric index based on characteristics and ecology of stream marcoinvertebrate communities and is a potential tool for understanding the mechanisms of streams' ecological decline in urban watersheds (Karr 1998). This index combines 10 metrics of stream macroinvertebrate communities into one value that generally ranges from 10 (poor) to 50 (excellent) (Fore et al. 1996). This score can then be used to compare streams of varying urban patterns to detect any relationships.

1.1 Ecology of Freshwater Invertebrates

Freshwater invertebrates include many phyla and have a range of attributes. Most are partially aquatic and partially terrestrial, although some are fully aquatic (Smith 2001). The adult stages of most insects are terrestrial. Life history traits vary among taxa – some exhibit one generation per year, some multiple generations per year, and some multiple years per generation. Dormancy is common in regions with large differences in seasons (either in terms of temperature or precipitation). These invertebrates frequently use environmental cues to time their emergence. They are seen at many trophic levels – herbivore, carnivore, and detritivore. Temperature regimes play a large role in the life histories of aquatic invertebrates in terms of magnitude and timing of egg growth, larval development, and adult emergence (Sweeney 1984). Nutrient inputs from organic matter, which can vary seasonally and from stream to stream, also have a similar influence.

Common categories in which to divide freshwater invertebrates are “clingers,” “sprawlers,” “climbers,” and “burrowers” (Smith 2001). Clingers are those that cling to the substrate (e.g. rocks) using modified appendages and tend to have flatter body shapes. Sprawlers move around the substrate and tend to be found under rocks or among debris. Climbers reside on plants in slower reaches of streams. Burrowers reside in soft substrate, usually clay, silt, or silt-sand.

Macroinvertebrates can also be grouped by feeding strategies. Primary consumers may be filter-feeders, deposit-collectors, scrapers, or shredders (Lamberti and Moore, 1984). Deposit-collectors usually reside in areas with slow-moving current, while scrapers, which remove algae and detritus from rocks, are more abundant in faster currents. Shredders consume living plant matter or coarse particulate organic matter, such as fallen leaves. Other invertebrates are predators – the type and abundance of these predators can alter the composition and abundance of invertebrates in a particular stream reach (Peckarsky 1984).

Substratum quality plays a major role in freshwater invertebrate ecology (Minshall 1984). There is a large variety of substratum types with varying combinations of organic and inorganic materials that can also vary in size. Macroinvertebrates have evolved a suite of adaptations for using and coping with such different substrata, with some taxonomic groups specializing on a specific substratum. These taxa tend to be found in higher abundances on the substrate for which they are specialized. Overall invertebrate abundance appears to increase with increased size of inorganic particles, peaking at about 30 mm. However, abundance increases more with heterogeneous substrate sizes, likely because this allows for a more diverse assemblage due to a wider variety of habitats.

1.2 Macroinvertebrates and Urbanization

Current research assessing the impacts of urban development on macroinvertebrates indicates little influence on the abundance (total number of individuals) of macroinvertebrates in streams (see below for references). Instead, urban development appears to cause a decline in the species richness (number of different species) 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.

In Washington, Pederson and Perkins (1986) found no difference in macroinvertebrate abundance between an urban stream and a forested stream, but did find differences in taxa composition. In Maine, streams in urbanized watersheds had significantly lower taxa richness than streams in forested watersheds (Morse et al. 2003). In Maryland, urban streams showed a similar trend, with significantly lower taxa richness than streams in either forested or agricultural watersheds (Moore and Palmer 2005). Those same streams had fewer collectors, filterers, predators, and scrapers than other streams. However, shredder richness was not affected.

Macroinvertebrate richness also shows a negative correlation with the percent of the watershed covered in impervious area. Percent impervious area explained 70% of the variation in taxa richness in Maryland streams (Moore and Palmer 2005). In Alaska, total macroinvertebrate abundance could not be explained by percent impervious area, but the abundance of certain taxa could (Ourso and Frenszel 2003). Abundance of Ephemeroptera, Plecoptera, and Trichoptera (EPT, taxa generally associated with less-disturbed streams), percent scrapers, and percent shredders were negatively correlated with percent impervious area. Alternatively, percent filterers, percent collectors, and percent predators were not affected by impervious area. Interestingly, this response is generally opposite of the response seen in Maryland (described above), suggesting that regional factors contribute to the specific response of stream macroinvertebrate communities.

Richness also shows a negative correlation with human population density in the watershed. In northern Virginia, total numbers of macroinvertebrates were unaffected by human density, but genus richness and diversity were significantly lower in watersheds with greater than 10 people per hectare (Jones and Clark 1987). In Connecticut, macroinvertebrate richness was negatively correlated to human population density, as well (Urban et al. 2006).

BIBI scores for streams decline with urbanization, as well. Morley and Karr (2002) studied 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) also found a negative relationship between BIBI score and percent impervious area. In addition, they found that the aggregation of urban land, mean patch size of urban land, road density, and the number of road/stream crossings were negatively correlated with BIBI score, suggesting that the patterns of development may affect macroinvertebrate communities, in addition to quantity.

There are many possible mechanisms for why macroinvertebrate communities decline in more urbanized watersheds. Urban development increases the imperviousness of the ground, reducing infiltration of stormwater and increasing runoff (Paul and Meyer 2001). As a result, stormwater reaches streams more quickly, leading to higher streamflow peaks with shorter time lags following a storm event (although duration of this flow is shorter in time). Groundwater is not recharged, causing lower low flows between storms. These changes in hydrology alter the shapes of the stream channels, widening and simplifying them. Sediment size and retention of large woody debris (LWD) may also change, which in turn may degrade the habitat for benthic invertebrates (based on the above description of macroinvertebrate habitat needs).

Booth et al. (2004) found a relationship between this 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 midwestern United States and found temporal changes in macroinvertebrate assemblages to be related to streamflow metrics.

Urbanization also leads to declines in water quality, another possible explanation for decline in macroinvertebrate assemblages. Urbanization alters both nitrogen and phosphorus fluxes in streams (Groffman et al. 2004, Hatt et al. 2004, and Walsh et al. 2005). Streams in the

[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 0.5 year flood, a flow that occurs on average twice a year (Booth et al. 2004).

Seattle, Washington area have shown significant increases in phosphorus concentration with increases in urbanization. However, the relationship between nutrient levels and macroinvertebrate assemblages is not well documented in the literature. The varying tolerances of different species to pollution suggest that macroinvertebrate assemblages change in response to altered stream chemistry (Jones and Clark 1987, Morse et al. 2003, Roy et al. 2003, Chessman and McEvoy 2012).

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. As mentioned above, substrate is an important component of freshwater invertebrate life – if this changes, then changes in invertebrate community structure might be expected. The effects of urbanization on freshwater macroinvertebrates depend on the amount of time that has elapsed since the development was initiated (Paul and Meyer 2001). Immediately after development, there tends to be an increase in fine sediment and bare land as the watershed is eroded. Alternatively, Finkenbine at al. (2000) found that in a set of streams developed at least 20-yrs prior to the study, particle sizes were larger and percent fines were smaller than in reference streams. They hypothesized that over time, streams reach new sediment equilibriums after development. How these changes impact invertebrate communities is still poorly understood. Using experimental manipulation of stream sediment levels, Wagenhoff et al.

(2012) found pollution-intolerant taxa richness and density to be negatively affected by increased levels of fine sediment.

Despite these negative relationships between macroinvertebrate assemblages and urbanization, there is still unexplained variability. Streams of similar levels of urban development have been reported to have high variability in BIBI scores (Albert et al. 2007). This result could have occurred for a variety of reasons.

One possible explanation is a difference in urbanization patterns, not just quantity of urbanization. Moore and Palmer (2005) found that among the urban streams in their study in Maryland headwater streams, macroinvertebrate richness was positively correlated with the percent of the riparian area that was forested (when non-urban streams were included, the correlation disappeared). Alberti et al. (2007) found negative relationships in Puget Sound streams between BIBI and several sub-basin metrics in addition to quantity of urban land: road density, number of roads crossing streams, mean patch size of urban land, and an aggregation index of urban land. These results suggest that the particular spatial arrangement of urban development and the quantity of urban infrastructure may play a role in macroinvertebrate community health.

Most research of urban streams has examined streams of varying quantities of urban development. To date, no published studies have assessed streams with similar quantities of urban land but varying patterns of development. This study attempts to do this – to isolate the role that different patterns of development play in stream BIBI score by controlling for quantity of urbanization. The overall objective of this study is to better understand the mechanisms that lead to decline of BIBI score in urban streams and to identify ways to minimize impacts from urbanization. Such information has the potential to inform local governments about how to better manage their land uses in ways that have less impact on local streams.

Caution should be used, however, with this approach. Identifying such characteristics is inherently tricky as watershed processes are incredibly complex and urban streams experience multiple stressors that are hard to tease apart. Practices that may be found to have a beneficial impact on BIBI score may not have an overall beneficial impact on the stream, and may even have a detrimental impact on other components of the local ecosystems.

1.3 Research Questions and Hypotheses

This study sought to answer the following questions: (1) Among streams with similar quantities of urbanization, are patterns of urban land cover and infrastructure related to BIBI score and/or some of the metrics that comprise the BIBI scores? and (2) Among streams with similar quantities of urbanization, what differences in physical characteristics are seen between streams with high and low BIBI scores? To try to resolve these questions, the following hypotheses were tested in this study:

1. BIBI score is related to various measures of urban patterns, such as aggregation, patch size or urban land, and percent forest.
   i. Measures of aggregation are negatively related to BIBI score.
   ii. Patch size of urban land is negatively related to BIBI score.
   iii. Percent forest is positively related to BIBI score.
2. BIBI score is related to infrastructure intensity, such as road density, number of road crossings the streams, and stormwater outfalls.
   i. Road density is negatively related to BIBI score.
   ii. Number of roads crossing streams is negatively related to BIBI score.

iii. Stormwater outfalls are negatively related to BIBI score.

3. BIBI score is related to land cover composition in 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 of 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.

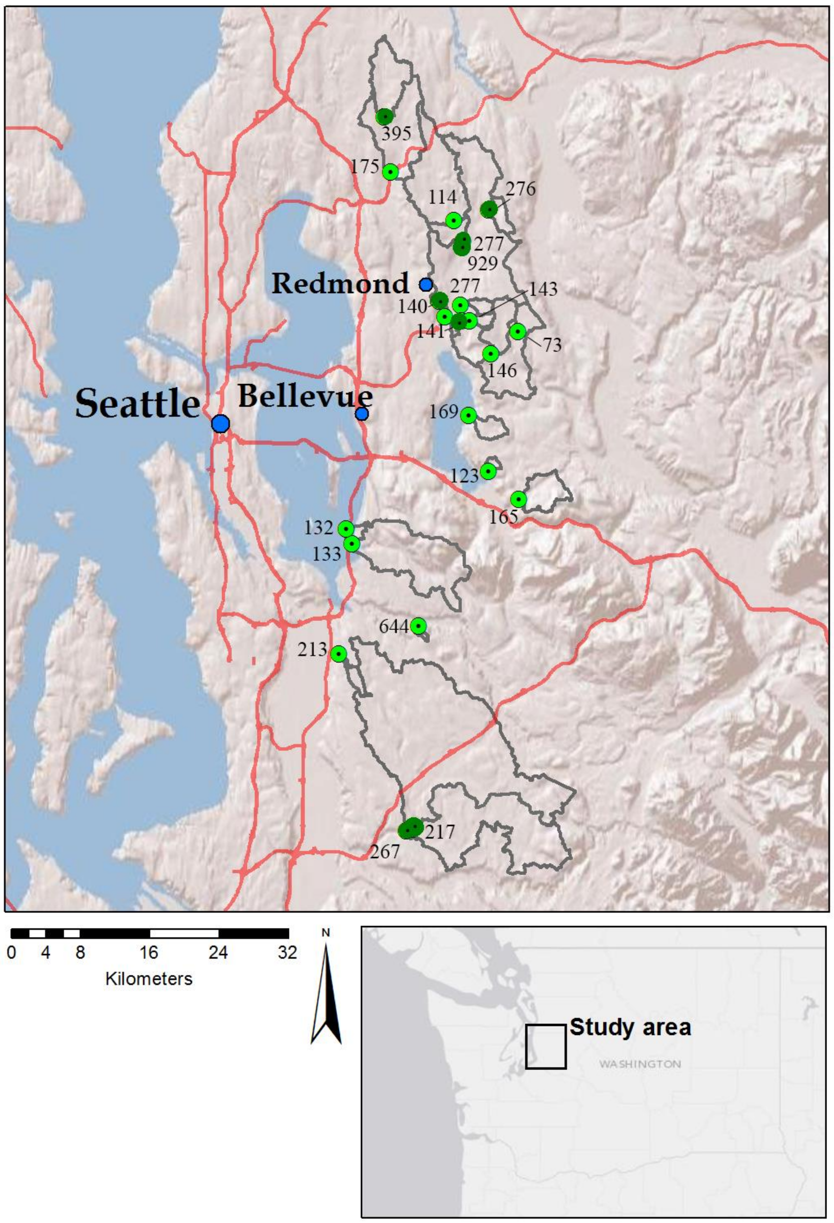

Figure 1: A map of study sites, including codes from Puget Sound Stream Benthos, with drainage basins, major cities, and highways. Dark green dots indicate the extreme-scoring streams that were visited.

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 the 22 streams that were sampled from 2006-2008 were included (the land cover dataset used in the analyses was from 2007). The stream codes and names associated with these sites are listed in Appendix A.

To better understand the mechanisms leading to differences in BIBI scores, the four highest scoring and four lowest scoring streams of the 22 study streams 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 any subtle trends. Specific details on these eight study streams are provided in Table 1 (note that one stream lacked a name, it is later referred to as "Tiny" in this study). 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 for BIBI.

Table 1: 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 basin
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 classes were 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.

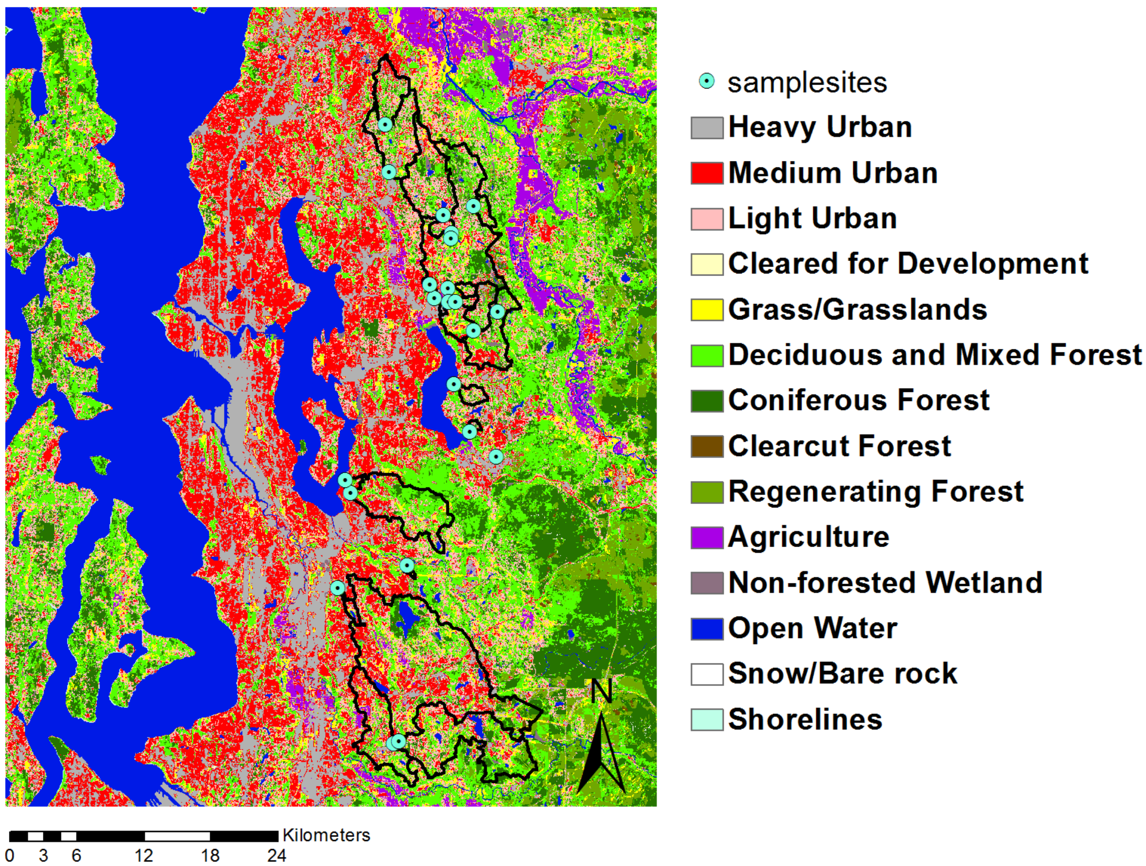

Figure 2: Land cover classification for study area. (Data obtained from the Urban Ecology Lab at the University of Washington)

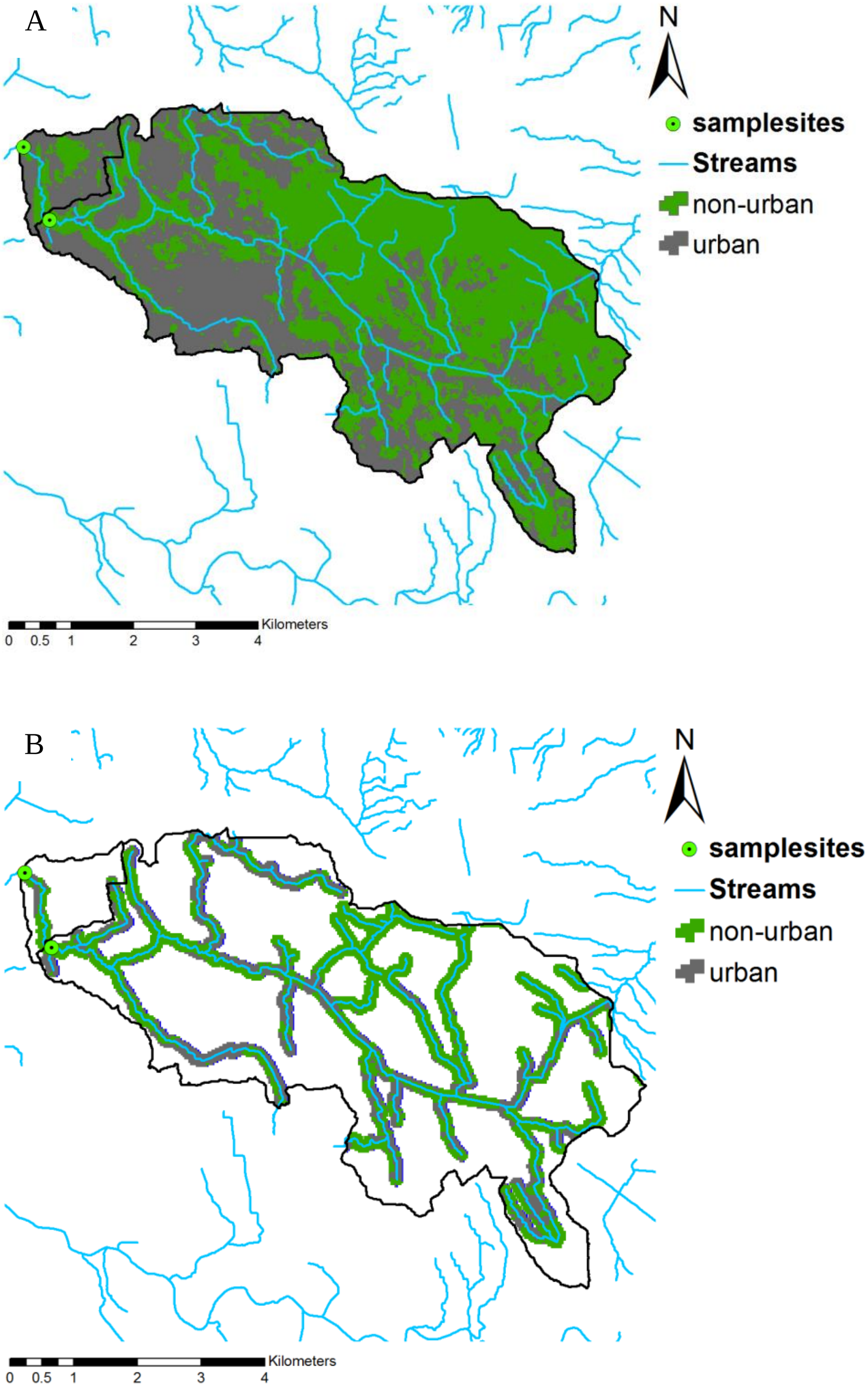

Figure 3: Land cover reclassifications for a selected basin. (A) Urban/non-urban reclassification of landscape (B) 100-m riparian zone reclassification

Table 2: 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 6.1m (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. Upon arrival at the stream, bankfull width was measured and then multiplied by ten to determine the length of the reach to use. 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) (≥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 diameter is considered a biologically significant sediment size for invertebrates and salmon (Finkenbine et al. 2002). This was used to calculate percent fines by weight 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 the influence of such outliers.

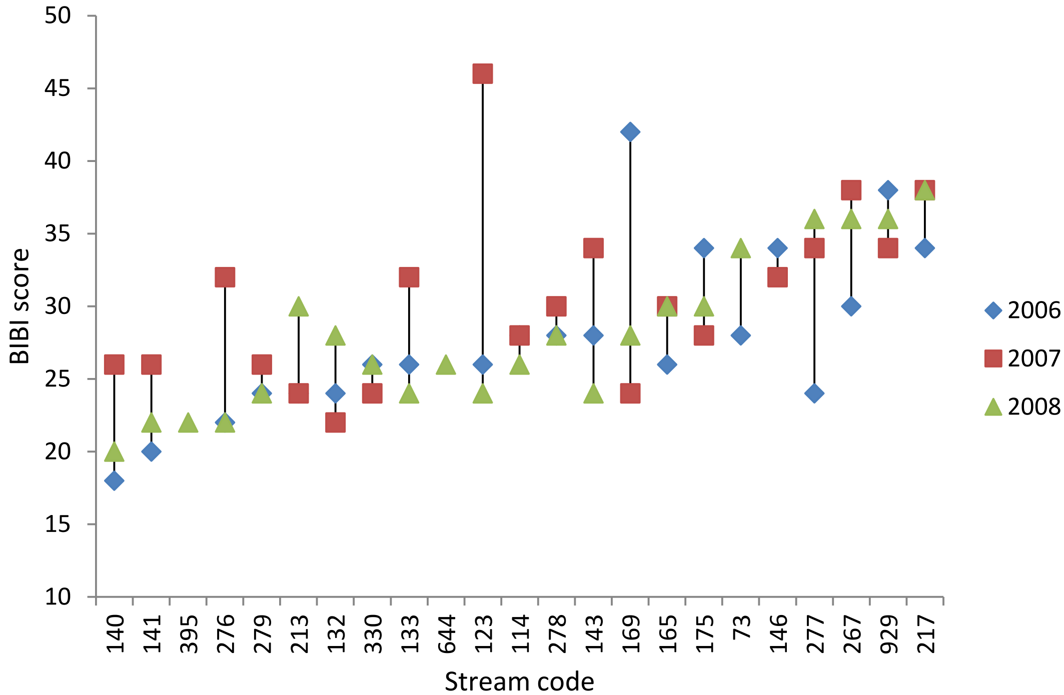

Figure 4: BIBI scores for study streams (obtained from Puget Sound Stream Benthos). Streams are ordered left to right by increasing median BIBI score.

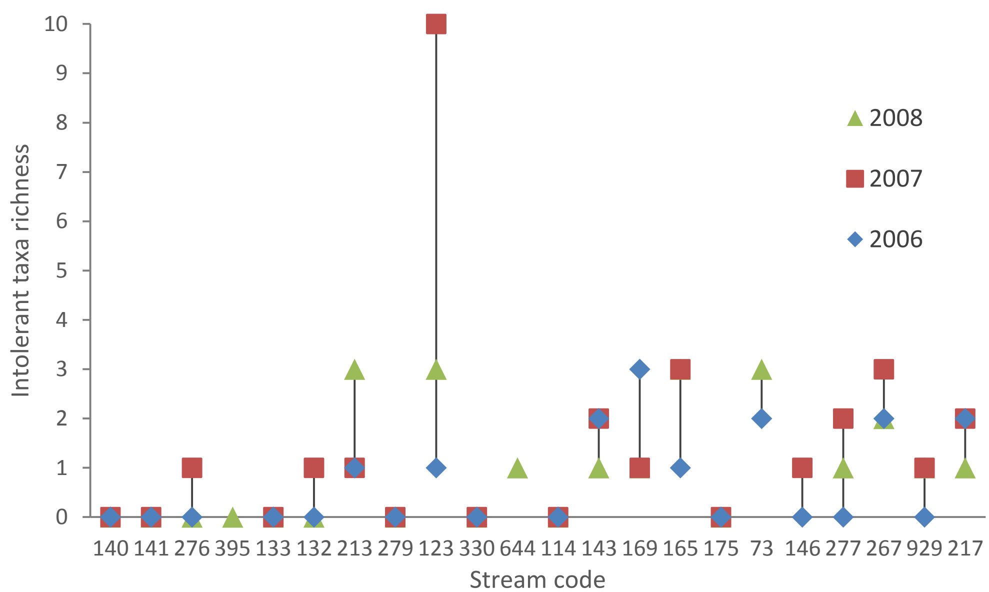

Figure 5: Intolerant taxa richness measured at each stream between 2006 and 2008. Streams are ordered left to right by increasing median BIBI score. See text for analysis of stream 123 with a large value for 2007.

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.

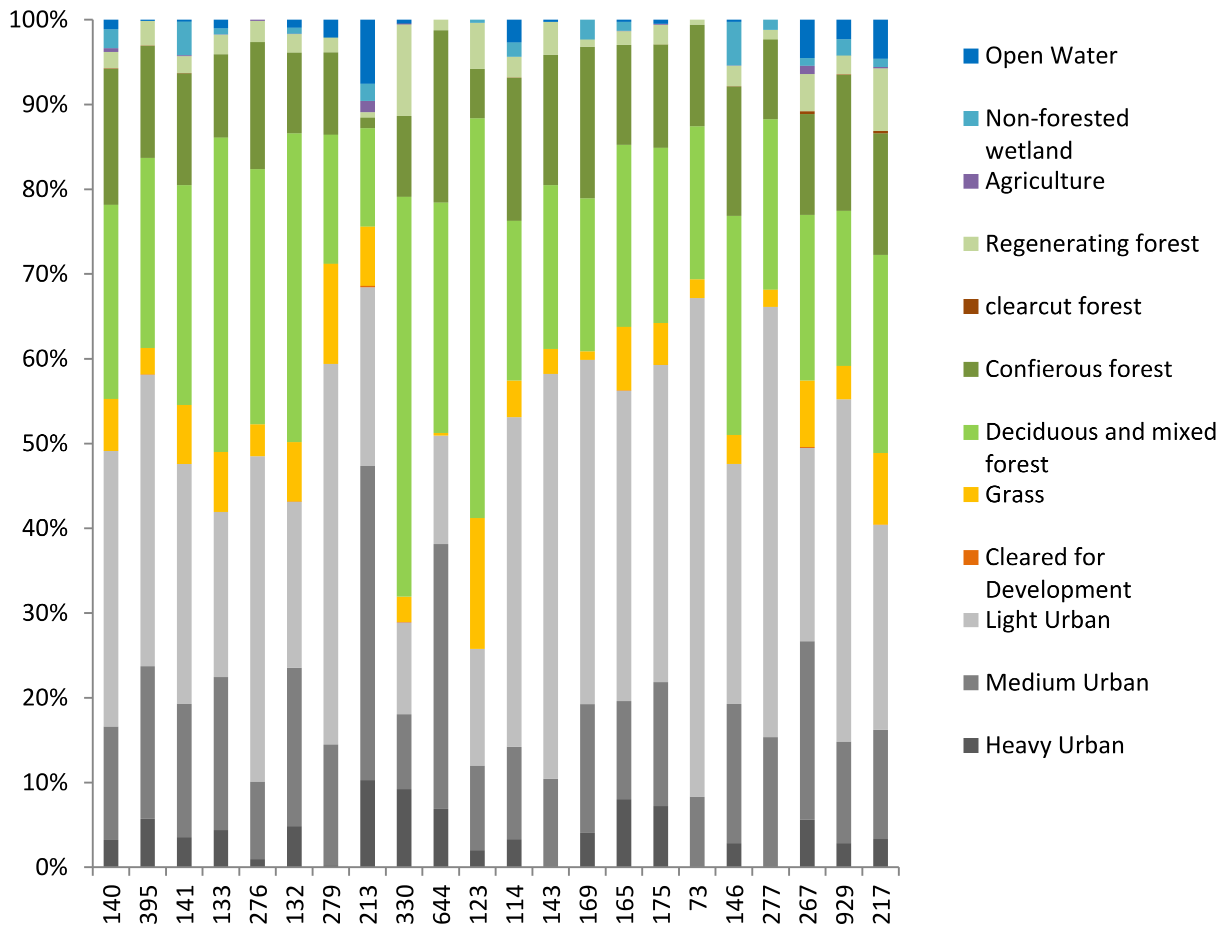

Figure 6: Composition of land covers for each study basin in 2007. Data obtained from the Urban Ecology Laboratory at the University of Washington. Basins are ordered left to right by increasing median BIBI score.

Table 3: 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.

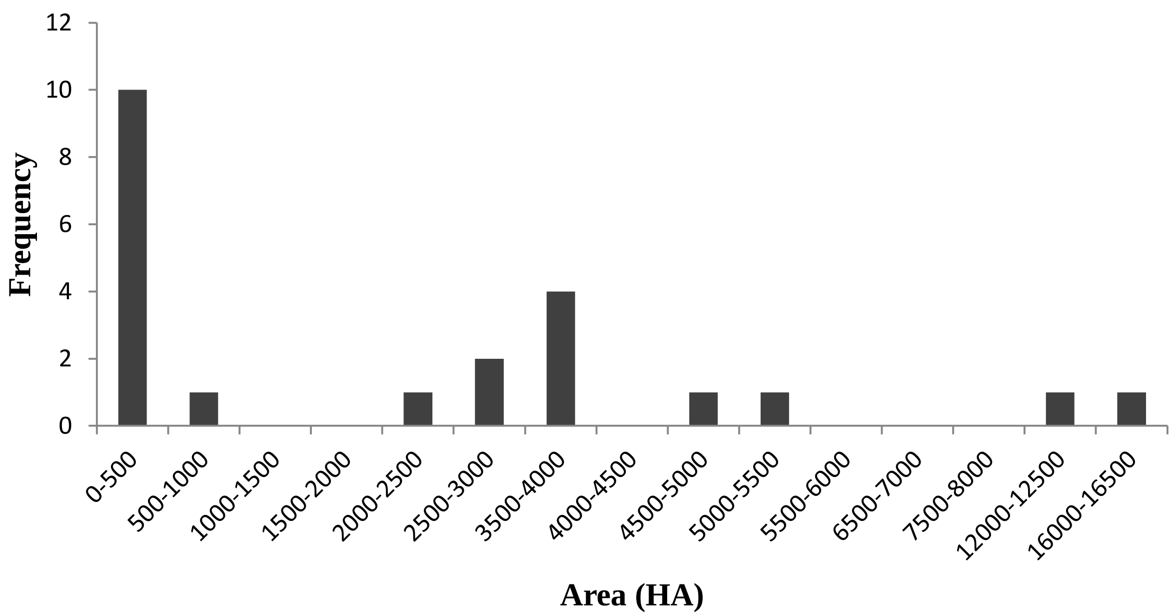

Figure 7: Frequency of basin sizes among study basins.

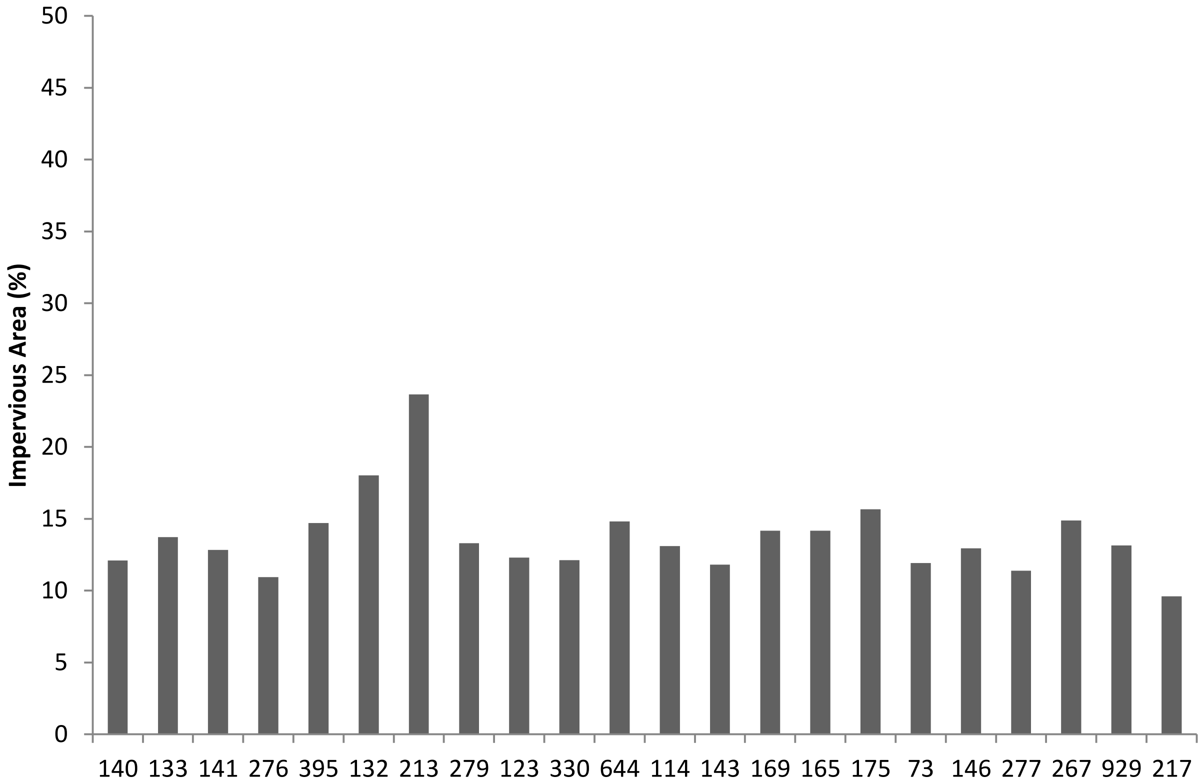

Figure 8: Percent of each study basin covered in impervious surfaces in 2006. Data obtained from the Washington Department of Ecology.

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 4: 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 5: 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 6: Summary statistics for urban infrastructure in study basins.

	Road area per basin area (%)	Road-Stream crossings per km of stream (# km^{-1})
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 7: 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 of basin (%)	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 8: 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 of basin (%)	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 9: Summary statistics for forest patterns within 100m of stream. PD = patch density, LPI = largest patch index, ED = edge density, AI = aggregation index

	Percent of basin (%)	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	2.6	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 (Tables 10 and 11).

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 10: 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 11: 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 suggests 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.

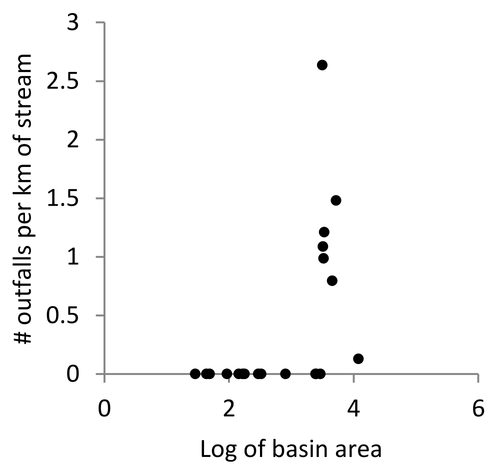

Figure 9: Number of stormwater outfalls per 10 kilometers of stream against area of the drainage basin. Although there is a statistically significant correlation between these variables, the shape of the plot suggests this correlation is not meaningful.

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.

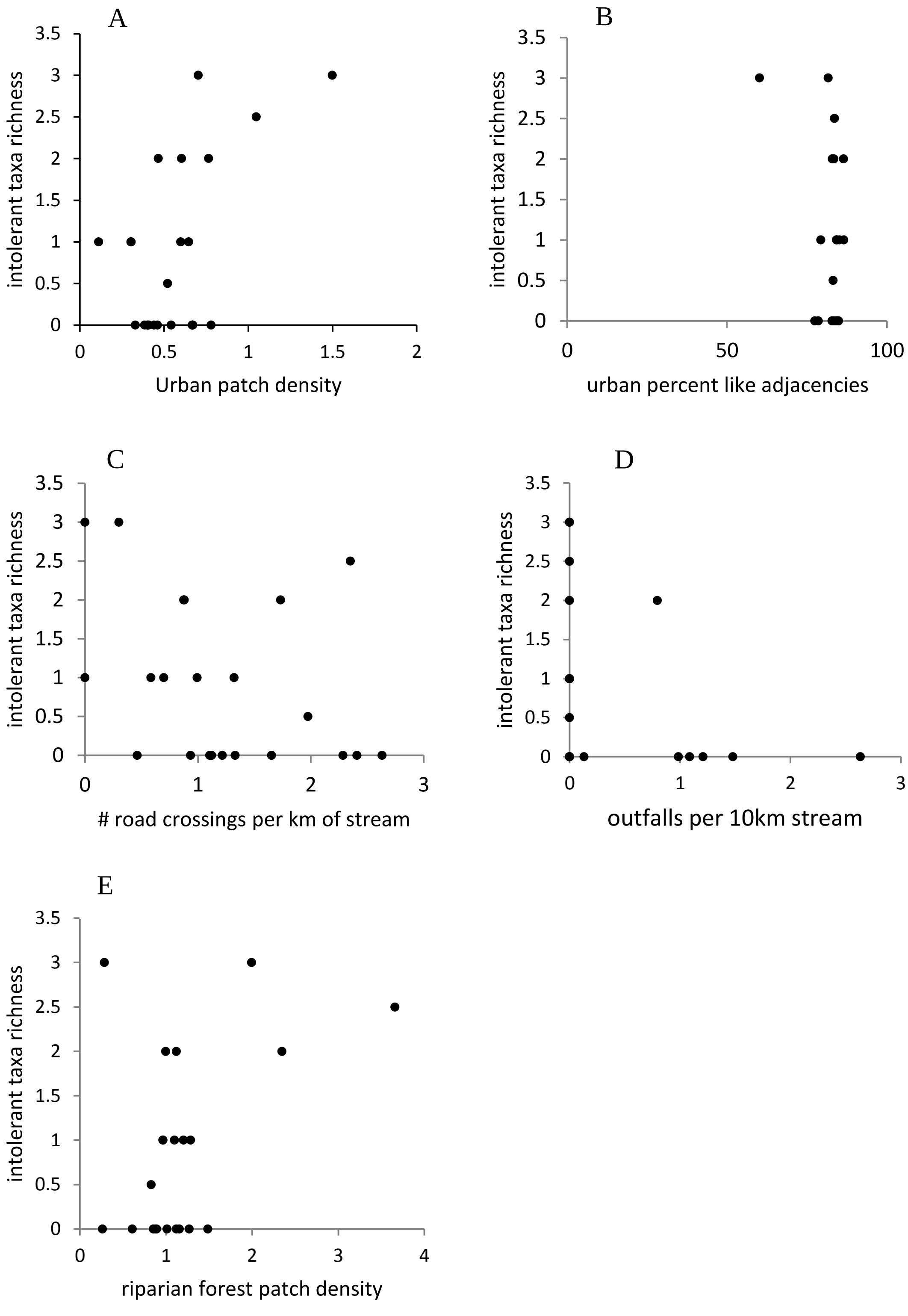

Figure 10: Plots of variables found to be significantly correlated with intolerant taxa richness. (A) Urban patch density (B) Urban percent like adjacencies (C) Number of road

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 12: Channel dimensions and predicted discharge (Q) of extreme-scoring streams.

BIBI score	Name	Bankfull width (m)	Bankfull depth (m)	Q ($m^3 s^{-1}$)
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).

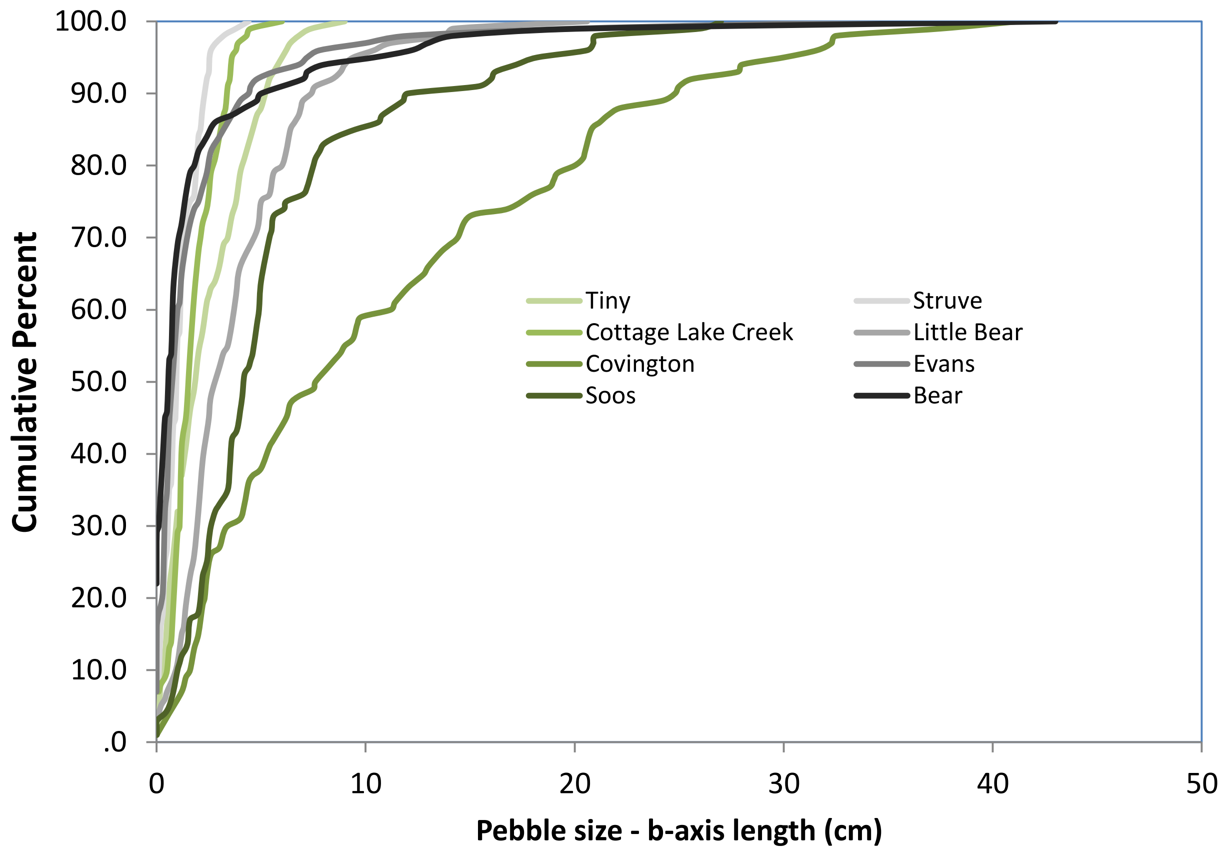

Figure 11: Cumulative percent of pebble size streams with extreme BIBI scores. Green = streams with high BIBI scores; grey = streams with low BIBI scores. Darker shades indicate larger streams.

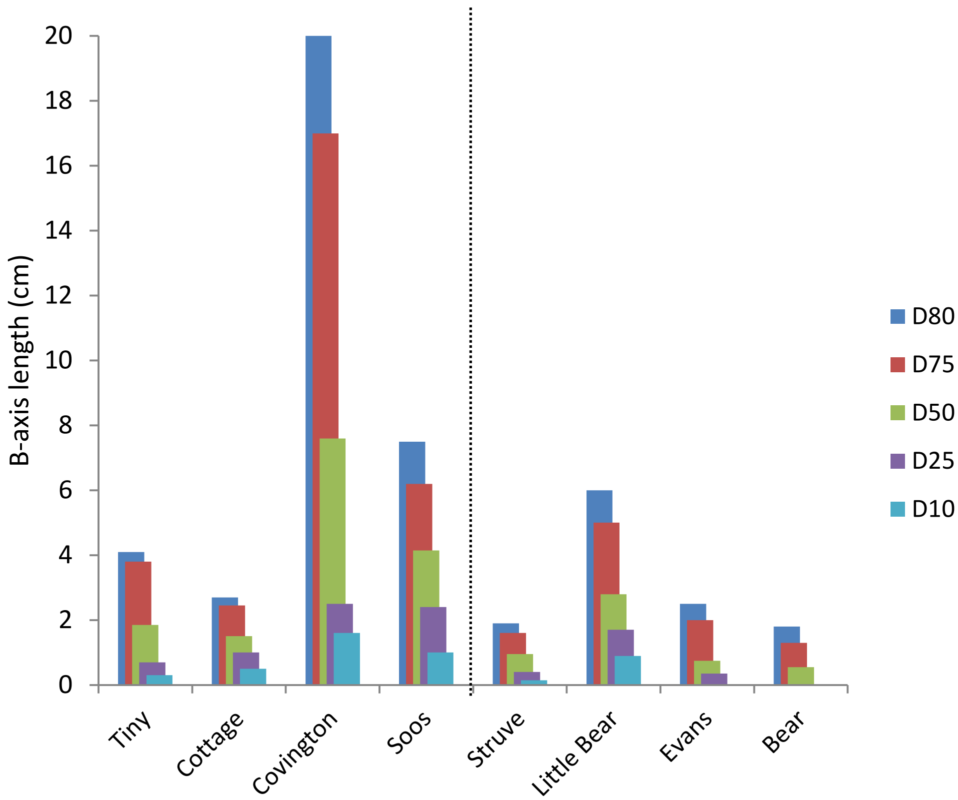

Figure 12: D10, D25, D50, D75 and D80 particles sizes in streams with extreme BIBI scores. Streams on the left half have high BIBI scores; streams on the right have low BIBI scores. The difference between the two groups is significant for D10, D25, and D50.

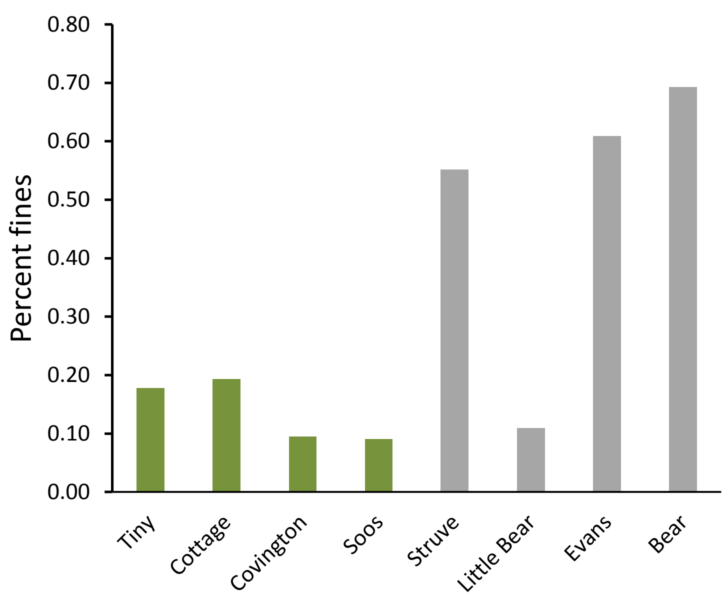

Figure 13: Percent fines (<2mm) in samples from each of the streams with extreme BIBI scores. Green = high-scoring streams, grey = low-scoring streams. The difference between the two groups is significant.

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.

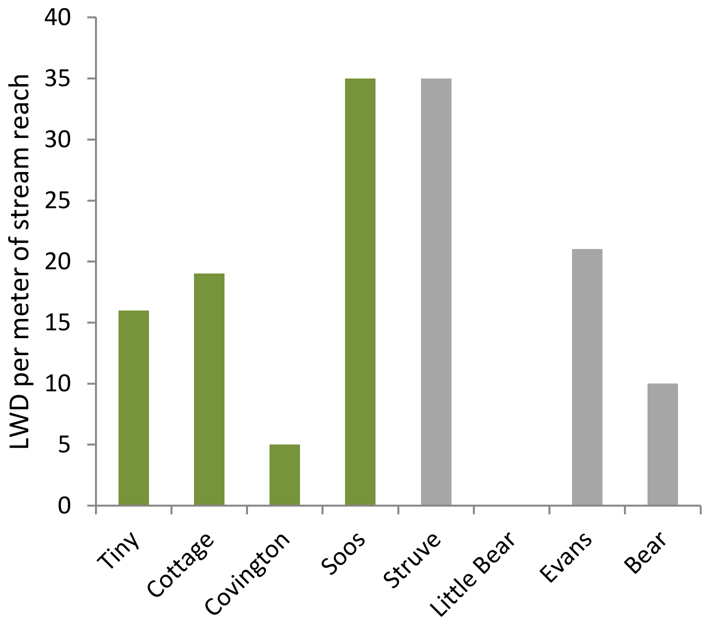

Figure 14: Number of large woody debris (LWD) per meter of stream reach in streams with extreme BIBI scores. Green = high BIBI scores; grey = low BIBI scores. The difference between the two groups is not significant.

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.

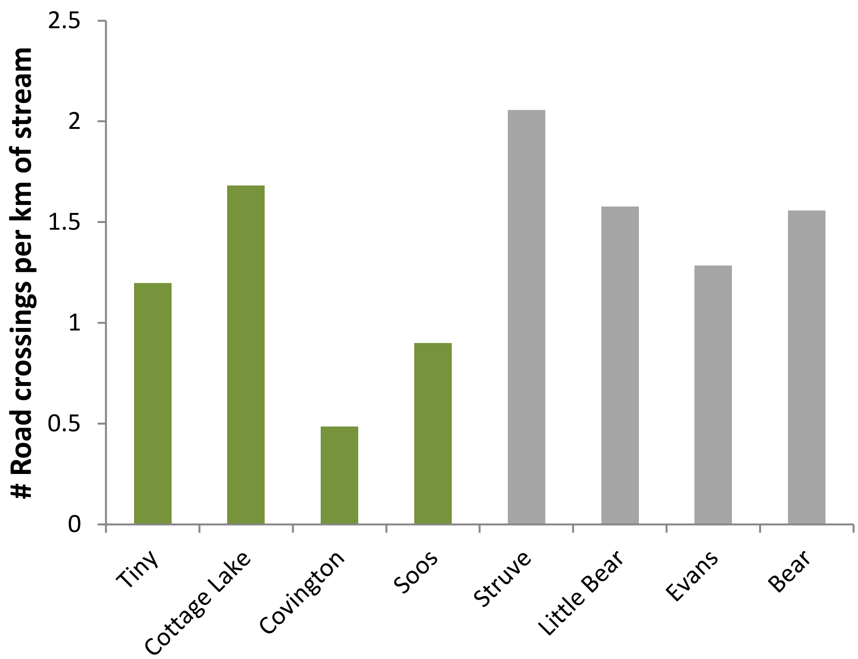

Figure 15: Number of road crossings per kilometer of stream in streams with extreme BIBI scores. Green = high BIBI score; grey = low BIBI score. The difference between the two groups is significant.

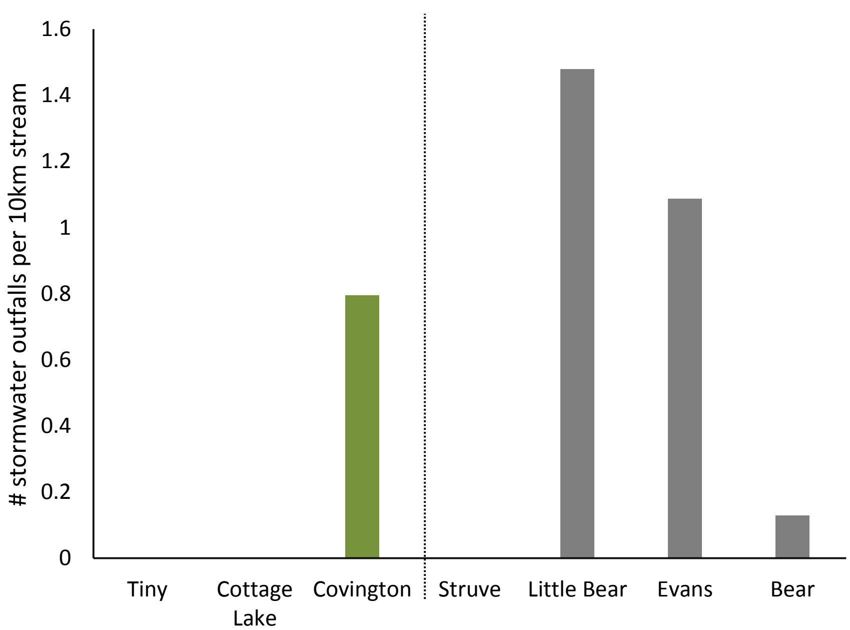

Figure 16: Number of stormwater outfalls per 10km of stream in streams with extreme BIBI scores. Left side of line = high BIBI score; right side = low BIBI score

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 B). 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 possible 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 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 be representative 2006-2008 LWD metrics. 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 Implications and Applications

A big finding in this study was that BIBI score was unrelated to any of the metrics originally hypothesized to affect it. Other studies have found that BIBI scores do decline with urbanization, but possibly the variation seen about the correlations is within the BIBI's natural range of variation. As a result, the 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). For example, Dolph et al. (2009) studied the variability of scores for a fish index of biotic integrity and found it to be quite high given the overall possible range of the score. A similar situation may occur with BIBI. To investigate differences and trends in streams with similar levels of urbanization, researchers may need to use metrics other than the BIBI score as a measure of health. In this study, intolerant taxa richness was useful in identifying trends in the study streams and may be able to provide extra information where the BIBI score cannot.

The results of this study as well as others suggest that road crossings are weakly related to stream health (McBride and Booth 2002, Alberti et al. 2007). They alter the physical stream structure downstream from the crossing (McBride and Booth 2005) and are associated with lower BIBI scores in streams of varying urbanization (Alberti et al. 2007). Local governments should study this phenomenon more closely to understand to what extent road crossings alter stream condition (by using other measures in addition to the BIBI) and what structures have less of an impact.

The positive relationship between patch density of urban land and intolerant taxa richness suggests that developing land in separated chunks with non-urban buffers in between may be associated with less disturbed stream macroinvertebrate communities than developing in aggregated lots (given that the area of development is to remain the same). Interestingly, light urban development (lower density single family houses, usually) still behaves like urban land, according to this study, and should be investigated further to determine if this holds true for other measures of stream condition. Light urban covers more area of land in the Puget Sound region than either heavy or medium urban (8% versus 3.6 and 7%, respectively), so characterizing its impact on streams is important.

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. This also implies that changes in hydrology associated with development may be one of the drivers of decline in macroinvertebrate communities.

Sediment size appears to be important for macroinvertebrate communities. The four streams with the highest BIBI scores in this study had significantly large sediment than the four streams with the lowest BIBI scores, suggesting that changes in sediment size may be a sub-driver in changes to macroinvertebrate communities following urbanization. Larger sediment may be able to provide more habitat options for varying invertebrate taxa, leading to healthier invertebrate communities. Achieving this is tricky, however, because sediment sizes cannot be directly changed. Instead maintaining a more natural hydrological regime through the use of low impact development techniques may be a way to approach this.

4.5 Data Limitations

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, and 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 support 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, but they do allow for the identification of trends that should be further studied.

Extreme-scoring Streams – Physical metrics were compared between the 4 highest-scoring and 4 lowest-scoring streams. However, these streams were ranked based on 2006-2008 data and the physical measurements were taken in 2012. There is likely to be some difference between the conditions in the streams between these times, as well as in the BIBI score. Additionally, a Wolman-pebble count was used to assess sediment size; however, this method tends to over-estimate sediment size (Marcus et al. 1995). It was assumed that while this may have happened, this overestimation was consistent among streams, allowing for comparisons between the groups.

5. Conclusions

5.1 Major Findings

The decline of freshwater macroinvertebrate communities with increasing urbanization is well-documented in the literature. This study provides one of the first assessments of streams with similar levels of urbanization, allowing for the potential to better understand the mechanisms that lead to biological decline and the identification of factors that may lessen the effect of urban development on streams. Streams with high BIBI scores from 2006-2008 had larger sediments than streams with low BIBI scores, implicating changes in sediment regimes as one mechanism leading to reduced biological health.

Evidence here also points to several factors that may be able to mitigate this decline in moderately urbanized drainage basins (approximately 13 to 15 percent imperviousness) in the

Pacific Northwest. Because increased fragmentation of urban land was associated with healthier macroinvertebrate assemblages in this study, local planners should consider the spatial arrangement of urban cover when assessing the condition of local streams. Infrastructure intensity (specifically the number of road crossings and stormwater outfalls) is negatively related macroinvertebrate community health. These results contribute to other research that has found that traditional stormwater infrastructure has an impact on streams. 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. Growing evidence in the literature suggests that riparian forests in urbanized basins are unable to mitigate the drastically altered hydrology. Stormwater infrastructure that pipes water straight to streams, bypassing the riparian area exacerbates this. 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.

5.2 Directions for Future Research

In this study, existing BIBI data were used, but it is recommended that future studies attempt to gather additional BIBI data to reduce limitations associated with existing datasets. This will allow researchers to identify more optimal basins that have more similar sizes. If the same team of researchers gathers the BIBI data, this will also cut down on the some of the variation in BIBI score due to variations in sampling skill.

Future research 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 factors of climatic and land cover change 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 examine different types of road crossings. Knowing how crossing type affects stream health and what potential designs can minimize impacts to the stream would be beneficial for managers and planners. Urban patch density was also found to be related to intolerant richness, but why this relationship exists is still unknown. Future research should investigate this by increasing the stream sample size to better detect the causes of this trend. Watershed modeling is a tool that is potentially able to assess how patch density affects these hydrologic processes.

Ultimately, the research presented here identifies several factors in urban landscapes that potentially contribute to stream health. Local managers and planners can use this list as a starting point for better understanding the ecology of their watersheds and to better prioritize funds for research.

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Appendix A: Study site codes and names

Code from PSSB website	Name	Longitude	Latitude
73	Evans Creek	-122.02232	47.667264
114	Cottage Lake Creek	-122.0887	47.744769
123	Tributary to Cedar River	-122.12568	47.461466
132	May Creek	-122.2006	47.529663
133	May Creek	-122.19466	47.519171
140	Bear Creek	-122.09833	47.677597
141	Evans Creek	-122.08086	47.67462
143	Tributary to Evans Creek	-122.07279	47.674662
146	Evans Creek	-122.05059	47.651634
165	Ebright Creek	-122.07361	47.608613
169	Many Springs Creek	-122.05321	47.569898
175	Little Bear Creek	-122.15494	47.778471
213	Panther Creek	-122.20746	47.442177
217	Covington Creek	-122.1315	47.319338
267	Big Soos Creek	-122.1385	47.317784
276	Struve Creek	-122.05336	47.752179
277	Tributary to Cottage Lake Creek	-122.0796	47.730382
279	Stensland Creek	-122.08238	47.685647
330	Issaquah Creek	-122.02162	47.55039
395	Little Bear Creek	-122.16099	47.8169
644	Perrigo Creek	-122.10416	47.688266
929	Cottage Lake Creek	-122.07959	47.725774

Appendix B: 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	Blephariceridae	Agathon	
Arthropoda	Insecta	Diptera	Blephariceridae	Blephariceridae	Agathon	arizonica
Arthropoda	Insecta	Diptera	Blephariceridae	Blephariceridae	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 C: 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