Stormwater Action Monitoring Status and Trends Study of Puget Lowland Ecoregion Streams: Evaluation of the First Year (2015) of Monitoring Data

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SAM
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ABSTRACT

In 2015, the condition of Puget Lowland streams was evaluated by collecting data for stream benthic invertebrates, periphyton, water quality, sediment quality, instream and riparian habitat, and land cover data at 105 sites across the region. The study was the first large-scale regional assessment of stream condition conducted as part of the Stormwater Action Monitoring (SAM) program, a collaborative, regional stormwater monitoring program funded by more than 90 Western Washington stormwater permittees.

The long-term goal of this study is to monitor how stream health changes over time in Puget Lowland streams as the area urbanizes and stormwater controls are implemented more broadly. The first round of monitoring in 2015 evaluated the current condition of wadeable streams within urban growth areas (UGAs) and outside UGAs representing a range of development conditions and impacts of stormwater runoff on small streams. The study questions were:

- What is the status of Puget Lowland ecoregion stream health within and outside UGAs?
- What are the major natural and human stressors impacting stream health?
- How do the results of this study compare to other stream monitoring programs?
- What monitoring parameters should be carried forward for SAM small stream monitoring in the future, and at what timing and frequency?

Many of the stream health measures, such as fecal coliform bacteria, total phosphorus, benthic index of biotic integrity (B-IBI), indicated poorer condition in streams within UGAs compared to streams outside UGAs. For example, 82 percent of stream length within UGAs was in poor condition based on B-IBI scores, while 31 percent of stream length outside UGAs was found to be in poor condition. Key stressors identified included watershed and riparian canopy cover, stream substrate characteristics, and nutrients. Watershed and riparian canopy cover were found to be the most important stressors to B-IBI at the regional scale. This suggests that canopy cover protection and recovery (reducing impervious surface) could lead to substantial improvements in B-IBI scores.

Comparisons of SAM streams data to other Puget Lowland stream monitoring programs were made for B-IBI scores and parameters representing water and sediment quality and stream habitat measures. Variability in results among programs was attributed primarily to differences in study designs, spatial sampling extent, and differences in methods.

Recommendations for SAM small stream monitoring in the future included two options: a minimum change scenario that maintains the two UGA strata but modifies the list of target parameters (e.g., eliminate monthly water quality sampling; add continuous stage measurement) and a second option that recommends a modification in the design to focus more specifically on the gradient of urbanization (relatively undeveloped to highly urbanized) that is more broadly captured by the two UGA strata.

EXECUTIVE SUMMARY

Introduction

The current condition (status) of Puget Lowland ecoregion streams (PLES) was evaluated by collecting data for stream benthic macroinvertebrates, periphyton, water quality, sediment quality, instream and riparian habitat, and geographic information system derived landscape data. The study was designed and implemented as part of the Stormwater Action Monitoring (SAM) program. SAM is a collaborative, regional stormwater monitoring program that is funded by more than 90 Western Washington cities and counties, the ports of Seattle and Tacoma, and the Washington State Department of Transportation.

The SAM regional sampling occurred from January to December 2015. For water quality measures, 61 sites were targeted for monthly sampling, although due to severe low summer flows in 2015, only 52 sites were considered suitable for use in the status assessment. For biological, sediment, and habitat measures, 85 sites were sampled once in the summer. Additional sites in Pierce County and the City of Redmond were monitored under alternative permit conditions (Option 2). Monthly Option 2 water quality sampling was conducted at 20 sites from October 2014 to September 2015 and biological, sediment, and habitat measures were sampled at the same sites once in summer of 2015. All the sites were randomly selected from small streams within and outside Urban Growth Areas (UGAs).

This report presents a status assessment based on the data collected, identifies key variables that correlate with biological condition, compares SAM to other Puget Sound stream monitoring programs, and makes recommendations for future monitoring to assess status and trends.

Background

Prior to the SAM program, the largest municipal stormwater permittees conducted individual outfall monitoring; most permittees were not required to conduct monitoring. For the 2013 National Pollutant Discharge Elimination (NPDES) permits a regional stormwater monitoring program was proposed by the Stormwater Work Group (SWG), an independent group of stakeholders including permittees, state and federal agencies, tribes, and businesses and environmental groups. The program includes status and trends monitoring in small streams and Puget Sound nearshore receiving waters, effectiveness studies, and source identification, collectively intended to provide feedback to permittees and the region to improve stormwater management and protect beneficial uses.

This first round study is in some ways a pilot, with the findings and outcome intended to inform the long-term design for future SAM status and trends monitoring in streams. The SWG established the goals and level of effort for the first round of monitoring and asked the following specific questions to guide the receiving water status assessment:

- What is the status of biology, water and sediment chemistry, and habitat in Puget Lowland ecoregion streams within and outside UGAs?
- What percent of Puget Lowland ecoregion streams are in "poor" and "good" condition within and outside UGAs?
- How do stream conditions correlate with natural and human variables?
- How do SAM results compare with those of other probabilistic or targeted regional and local Puget Sound stream monitoring programs?
- Which measures should be carried forward to assess status and trends, and at what frequency?

The long-term goals of the study are: to evaluate whether stream conditions are getting better or worse; to help us understand whether management actions are collectively adequate at the regional scale to protect and restore stream health; and to identify other patterns in healthy and impaired stream sites that might provide insight into improving stormwater management approaches.

Approach

A random probabilistic design was used to select the SAM sites. The design allows spatial characterization of large areas across the region that would not otherwise be possible and uses modest funding and resources. The conditions observed at the SAM sampling sites reliably represent all the Puget Lowland ecoregion streams, including streams in areas that were not sampled. The design makes it possible to report the current status for all stream miles rather than simply reporting the number of sample sites that are in good or poor condition.

At the time sites were selected for this study, 47% of the Puget Lowland ecoregion area was within UGAs and 53% was outside. Figure ES-1 shows the sites sampled in this assessment. Approximately half of the sites selected were within UGAs and half outside.

Watershed Health measurements were made once during the summer at all sites. These measurements included sampling of stream benthic invertebrates and periphyton, physical habitat, and selected water and sediment quality parameters.

Due to drought conditions in 2015, streams at some of the SAM regional monthly water quality sites went dry and sampling was discontinued. Water quality data from 52 SAM regional sites with the most complete data were used in the status assessment. The water quality samples were analyzed for nutrients, metals, organic contaminants (polycyclic aromatic hydrocarbons or PAHs), and fecal bacteria. Temperature, dissolved oxygen, pH, turbidity, and other measurements were made in the field.

The Water Quality Index (WQI) was calculated for the SAM regional sites using data for eight constituents measured during the 2015 monthly sampling program: temperature, dissolved oxygen, pH, fecal coliform bacteria, total nitrogen and phosphorus, total suspended solids, and turbidity. The WQI was not calculated for the Option 2 sites due to the difference in sampling period (October 2014 to September 2015). The WQI was included in the status assessment, while water quality data from the Option 2 sites were

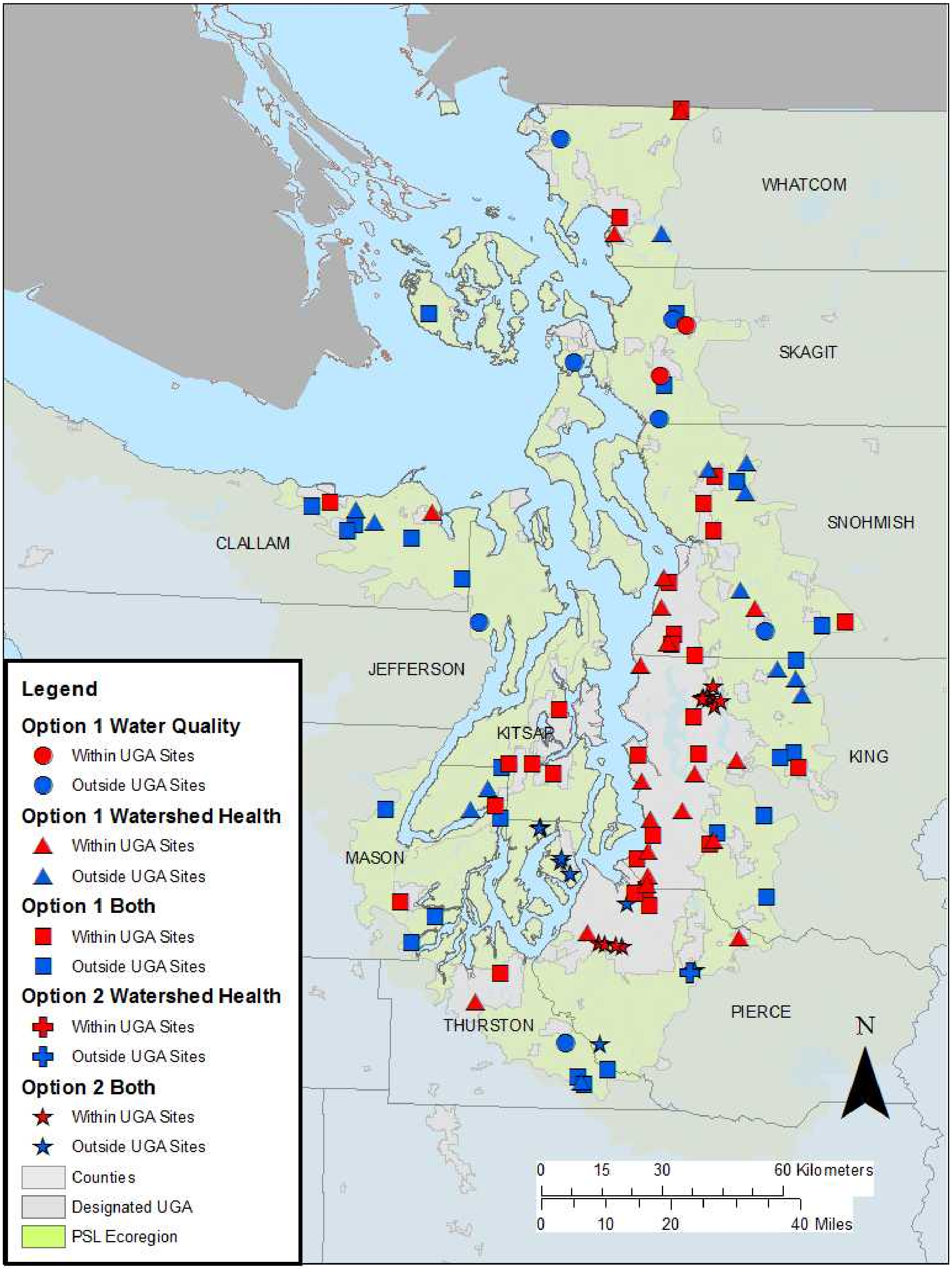

Figure ES-1. Puget Lowland ecoregion streams sites sampled in 2015 under Stormwater Action Monitoring (SAM Option 1) and Option 2 (permit alternative) monitoring. Sediment quality, biota, and habitat measures were collected at Watershed Health sites.

included with the SAM regional data to address questions regarding correlation of stream condition with natural and human stressors.

Streambed sediment samples were collected once during the summer from the stream substrate at all of the sites. Sieved sediment samples were analyzed for metals and organic contaminants, including PAHs, polychlorinated biphenyl compounds (PCBs), polybrominated diphenyl ethers (PBDEs), and common roadside use pesticides.

For the status assessment, values were compared to data from "least-disturbed" reference sites. For parameters for which regulatory standards exist, the data were also compared to state water and sediment quality standards. Additional non-regulatory screening thresholds were developed based on literature and with input from local, state, and federal experts on impacts of contaminants on stream ecology and biota. The reference conditions, standards, and additional thresholds were used to assign each site to a category of good, fair, or poor condition for each particular measure.

Per the statistical analysis, each site represented a proportion of the total length of small Puget Lowland ecoregion streams. These values were used to estimate the distribution of good, fair, or poor condition over the entire extent of small Puget Lowland ecoregion streams. A separate site-by-site comparison resulted in findings that were very consistent with the results of the status assessment.

Summary of Major Findings

Status Assessment

Table ES-1 presents a summary of the status assessment of conditions of streams within and outside UGAs. For nearly all parameters where differences were measured, Puget Lowland ecoregion streams are in better condition outside of UGAs. The following sections explain more of the meaning of these findings, and some additional findings. Data from SAM regional sites and Option 2 jurisdictions are combined unless otherwise specified.

Status of biological condition

Biological condition status focused on two indices that characterize the health of the instream communities at each site.

The Benthic Index of Biotic Integrity (B-IBI) measures the condition of stream invertebrates and is sometimes called the "stream bug index." B-IBI scores were significantly better outside UGAs with 46% of stream miles in good condition. In contrast, 82% of stream miles within UGAs had poor B-IBI scores. Of the 20 other measures of the biological health of stream invertebrates, 18 indicated better biological conditions outside of UGAs.

The Trophic Diatom Index (TDI) measures the condition of the periphyton diatom community sensitive to nutrient pollution in streams. TDI scores were significantly better outside UGAs. 71% of stream miles outside UGAs were in good condition and 66% within UGAs were in poor condition. Of the 44 other measures of the biological health of the periphyton community, 29 measures indicated better biological condition outside of UGAs.

Status of water quality

Water quality status was assessed using several measures of pollutant levels and basic water quality.

The Water Quality Index (WQI) combines and summarizes eight measures of conventional water quality for each site into a single numeric index. No streams within or outside UGAs were in poor condition based on the WQI; 67% of stream miles outside UGAs and 43% within UGAs were in good condition. Annual WQI scores were statistically different within and outside UGAs.

When disaggregated from the WQI, some individual measures showed stronger differences in status within and outside UGAs. Fecal bacteria and nutrients indicated poorer condition within UGAs. Minimum dissolved oxygen, maximum temperature, and maximum pH indicated similar condition within and outside UGAs. Minimum pH was lower at sites located outside UGAs.

Fecal bacteria were detected at all sites and in most monthly samples. Values were significantly higher within UGAs. 100% of stream miles outside UGAs were in good condition and 32% of stream miles within UGAs were in poor condition. 69% of all sites satisfied the criteria for safe human contact.

Nutrients: Nitrogen and phosphorus were measured in nearly all monthly samples and were significantly higher within UGAs. For total phosphorus, 80% of stream miles outside UGAs were in good condition; 46% of stream miles within UGAs were in poor condition. For total nitrogen, 68% of stream miles outside UGAs were in good condition; 43% within UGAs were in poor condition.

Metals: Total and dissolved cadmium and silver and dissolved lead were rarely detected; dissolved chromium, total lead and total and dissolved zinc were detected infrequently. Total chromium and total and dissolved arsenic and copper were often detected in monthly samples. Total and dissolved arsenic and total chromium values were significantly lower outside UGAs. Total and dissolved copper were not different within and outside of UGAs. Metals concentrations in more than 99% of all monthly samples were below state standards.

Organic contaminants: PAHs and pesticides were detected in fewer than 5% of monthly water samples with one exception: naphthalene (a PAH), was found in 24% of monthly samples. Because organic contaminants were detected so infrequently, it was not possible to evaluate differences within and outside UGAs, although naphthalene was detected more frequently at sites within UGAs.

Table ES-1. Numbers of sites assessed within and outside Urban Growth Areas (UGAs) for particular parameters; total percentages of Puget Lowland ecoregion stream length found to be in poor and good condition within and outside UGAs; and whether there was a statistically significant difference between conditions within and outside UGAs.

Parameter	Number of sites with data assessed		Percent of stream length in "poor" condition		Percent of stream length in "good" condition		Difference between OUGA and WUGA?
	OUGA	WUGA	OUGA	WUGA	OUGA	WUGA	
Biological:							
B-IBI	45	59	31	82	46	12	Yes
TDI	45	59	29	66	71	26	Yes
Water quality:							
WQI	24	28	0	0	67	43	Yes
Fecal bacteria	24	28	0	32	100	68	Yes
Minimum DO	24	28	63	64	38	36	No
Minimum pH	24	28	29	11	71	89	Yes
Maximum pH	24	28	13	7	88	93	No
Max.Temperature	24	28	54	54	46	46	No
Total phosphorus (Aug-Oct mean)	24	28	8	46	80	36	Yes
Total nitrogen (Aug-Oct mean)	24	28	12	43	68	39	Yes
Sediment quality:							
Arsenic	46	59	1	10	72	51	Yes
Cadmium	46	59	3	0	97	92	Yes
Chromium	46	59	3	2	48	43	No
Copper	46	59	3	6	35	40	No
Lead	46	59	3	4	92	70	Yes
Zinc	46	59	0	2	85	39	Yes
Total PAHs	46	59	0	2	100	89	NA
Total PCBs	46	59	0	0	100	95	Yes
Total PBDEs	46	59	0	0	100	93	Yes
Dichlobenil	46	59	NA	NA	NA	NA	No
Habitat:							
Canopy closure	46	59	20	20	61	74	No
Wood volume	46	59	44	61	39	31	No
Residual pool area	46	59	16	29	44	53	No
Stream Substrate	46	59	20	25	45	13	Yes
Bed Stability	46	59	25	36	64	26	Yes
Landscape:							
WS %urban	46	59	17	86	72	10	Yes
WS canopy cover	46	59	41	94	46	4	Yes
Riparian canopy cover	46	59	29	56	57	21	Yes
Areal nitrogen loading rate	46	59	16	76	56	12	Yes

OUGA is outside Urban Growth Areas; **WUGA** is within Urban Growth Areas; **B-IBI** is benthic index of biotic integrity; **TDI** is trophic diatom index; **WQI** is Water Quality Index; **PAHs** are polycyclic aromatic hydrocarbons; **PCBs** are polychlorinated biphenyls; **PBDEs** are polybrominated diphenyl ethers. **WS** is Watershed. **NA** is Not Assessed due to limited frequency of detection outside UGAs (Total PAH) or due to lack of screening level (dichlobenil).

Status of sediment quality

The sediment quality status assessment focused on metals and organic contaminants that were frequently detected and that have established ecologically-relevant thresholds, specifically adverse effects on benthic invertebrates.

Metals were detected at almost every site with the exception of silver which was detected in 57% of the samples. Concentrations of arsenic, cadmium, lead, and zinc were significantly higher within UGAs. Overall, only small percentages of stream length were in poor condition based on sediment metals concentrations. Sediment chromium (five sites) and cadmium (1 site) concentrations were the only metals that exceeded the state Sediment Screening Levels (a threshold exceedance would indicate a high potential for adverse effects to benthic invertebrates); levels above which indicate a high potential for adverse effects to benthic invertebrates. A number of sites exceeded the lower Sediment Cleanup Objective (a "no-effects" threshold concentration) for arsenic (28 sites within and outside UGAs), copper (1 within UGA site), chromium (6 within and outside UGA sites), and silver (3 within UGA sites).

Total PAHs were detected in 43% of the samples. Because PAHs were detected so infrequently, it was not possible to evaluate differences within and outside UGAs, although Total PAH and many individual PAH compounds were detected more frequently at sites within UGAs. All stream miles outside UGAs and 89% within UGAs were in good condition based on Total PAH; 2% of stream miles within UGAs and no stream miles outside UGAs were in poor condition. Total PAH did not exceed Sediment Screening Levels. One site did exceed the Sediment Cleanup Objective.

Total polychlorinated biphenyls (PCBs) were quantified from detected PCB congeners at every site. Concentrations of Total PCBs were higher within UGAs. No stream miles within or outside UGAs were considered to be in poor condition based on ecologically relevant screening thresholds. One site did exceed the no-effects Sediment Cleanup Objective.

Total polybrominated diethyl ethers (PBDEs, used as flame retardants) were quantified from detected PBDE congeners at every site. Concentrations of PBDEs were higher within UGAs. No stream miles within or outside UGAs were considered to be in poor condition based on ecologically relevant screening thresholds. There are currently no regulatory sediment quality thresholds for PBDEs.

Total phthalates (plasticizers) were detected infrequently except for bis(2-ethylhexyl) phthalate which was detected with greater frequency within UGAs. No sites exceeded Sediment Screening Level above which adverse effects to benthic invertebrates might be expected. Two sites exceeded the no-effects threshold Sediment Cleanup Objective.

Common roadside use pesticides were detected infrequently, with the exception of the herbicide dichlobenil which was detected in over 70% of the samples. However, there was not a statistically significant difference in the distribution of dichlobenil concentrations within and outside UGAs, and no screening levels or sediment standards were available to evaluate the ecological importance of the observed dichlobenil concentrations.

Status of habitat condition

Over 100 stream habitat metrics were calculated from the stream reach field surveys conducted as part of the watershed health monitoring. Five were chosen for assessment that represent measures of tree cover (canopy closure), habitat complexity (wood volume and residual pool area), stream substrate (medium substrate particle diameter), and stream bed stability (relative bed stability). Values were compared to reference site conditions; no regulatory standards exist.

Stream substrate and bed stability were in significantly better condition outside UGAs. Conditions within and outside of UGAs were not significantly different as measured by riparian canopy closure, wood volume, or residual pool area.

Status of landscape condition

Over 100 watershed and riparian physical landscape and land cover metrics were calculated from available geographic information system layers by the U.S. Geological Survey (USGS) following methods developed as part of the National Water Quality Assessment. Four metrics were chosen for assessment because these metrics were identified as predominant stressors presenting risk to benthic invertebrate and/or periphyton communities. These metrics were watershed percent urban development, watershed percent tree canopy cover, riparian canopy cover, and watershed areal nitrogen loading. Values were compared to reference site conditions.

There were statistically significant differences (less urban development and more tree cover) for watersheds draining to sites outside versus inside UGAs for all four land cover metrics. Based on comparisons to reference site watershed and riparian conditions, more than 50 to almost 100 percent of within UGA stream miles were in poor condition with respect to land cover and about 5 to 20 percent were in good condition. Streams outside UGAs were generally in better condition with respect to land cover – less stream length in poor condition and more stream length in good condition relative to streams within UGAs.

How does stream condition correlate with natural and human variables?

To be considered a reliable biological response indicator, the indicator should respond to natural and human stressors, but be relatively insensitive to natural landscape variables that are largely beyond human control (e.g., watershed area or basin elevation). Based on analyses presented in this report, we conclude that B-IBI is not significantly affected by natural landscape variables, but significant relationship with natural and human stressors were identified. As shown in Figure ES-2, the relative risk/attributable risk analysis for B-IBI showed that risk of poor B-IBI scores was associated with several measures of human disturbance. The highest attributable risk of poor B-IBI condition was determined to be watershed canopy cover (59%) followed by riparian canopy cover (34%) and watershed percent urban development (29%). As an example, the results suggest that as a best-case scenario a 34 percent reduction in the extent of stream reaches classified in poor B-IBI condition would result if poor riparian conditions were substantially improved.

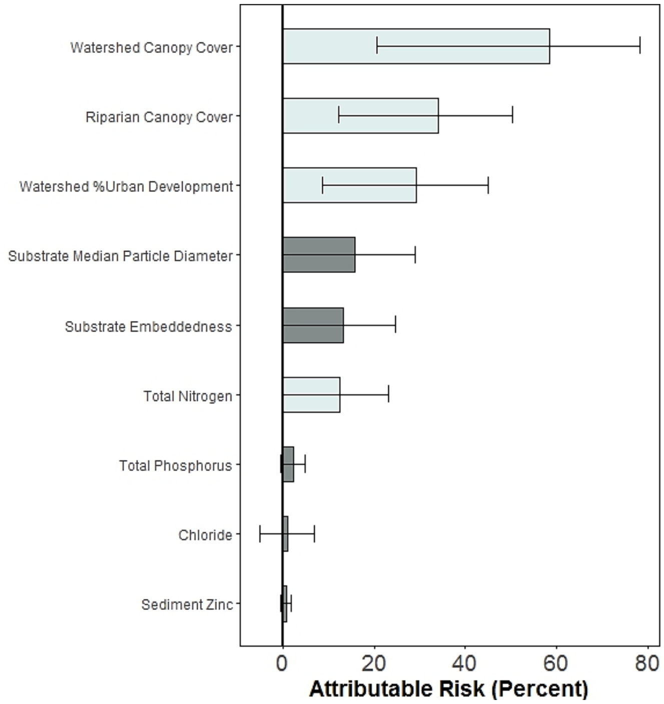

Figure ES-2. Attributable risks to B-IBI scores and their 95 percent confidence intervals. Stressors shown with dark shaded bars are insignificant because the error bars include 0 percent attributable risk.

The risk of poor B-IBI scores was also related to stream particle size and embeddedness, which in the Puget Lowland ecoregion are also associated with urban development; however, these factors were not found to be statistically significant in the attributable risk analysis. The attributable risk for total nitrogen (12 percent) was statistically significant.

Based on analyses presented in this report, we conclude that TDI is not significantly affected by natural landscape variables, but significant relationship with natural and human stressors were identified. TDI, the other biological assessment metric assessed, was not affected by natural landscape variables. The highest attributable risk (34%) for poor TDI scores was stream total phosphorus concentrations.

How do SAM's stream assessment results compare with other monitoring programs?

To inform our recommendations for future SAM streams sampling and analysis, SAM's findings were compared with those of other monitoring programs of various designs. Overall, variations in results among programs were likely due to differences in study designs (targeted versus probabilistic), the numbers of sites sampled, the types of streams targeted for sampling, and the geographic extent of the sampling. Some variations were due to differences in methods used.

The range and medians of B-IBI scores, fecal bacteria values, total phosphorus, sediment copper and zinc concentrations, and canopy closure values all differed among the various programs included in this assessment. However there was a general pattern of lower B-IBI scores and canopy closure values, similar sediment copper concentrations, and higher fecal bacteria values and total phosphorus and sediment zinc concentrations within UGAs compared to outside UGAs.

Although protocols for collecting B-IBI data across programs differ to some degree, the data are comparable. B-IBI results are consistent across programs with relatively large sample sizes (more than 30 sites within or outside UGAs). Option 2 represented only a small geographic portion of Puget Lowland ecoregion streams so results were not similar to SAM regional results.

SAM and Ecology's 2009 Watershed Health and Salmon Recovery (WHSR) monitoring found less embeddedness in streams sites outside UGAs, while Ecology's 2013 WHSR, Kitsap County's Watershed Health, and Water Resource Inventory Area (WRIA) 8 Status and Trends programs found the opposite.

Sediment copper results were not consistent among programs. SAM and the U.S. Geological Survey's (USGS) Pacific Northwest Stream Quality Assessment (PNSQA) results were most consistent; both programs sieved their sediment samples while King County's sediment and Ecology's Watershed Health and Salmon Recovery monitoring did not. Lower sediment zinc concentrations were also observed by programs that did not sieve samples.

In this assessment, each SAM regional site and each Option 2 site represented much different lengths of total stream miles in the analysis, with Option 2 sites typically accounting for about 6 percent or less of the portion of the total Puget Lowland ecosystem stream length represented by the SAM regional sites. Relatively significant errors in estimates of regional statistics could occur if local targeted or probabilistic study data were used to infer regional conditions. However, these differences become smaller as the number of targeted or probabilistic sites increases and the targeted sites are selected to represent the same strata or stressor gradient that is the focus of the regional probabilistic design.

There are several ways to approach the question of how SAM compares to local monitoring programs. One way is to compare the questions asked and answered by the programs; they have different purposes, and so the findings are not meant to be comparable. Most local monitoring programs are targeted, and by definition all of them are geographically limited, so the findings above are relevant to this question.

Discussion

These results confirm that overall, biological conditions—along with water and sediment quality and habitat measures—are better outside UGAs than within UGAs.

The biological condition assessment found that measures of urbanization, substrate condition, habitat, and nutrients explained the variability in biologic indices. These findings are consistent with other previous studies in Puget Lowland ecoregion streams. The B-IBI and TDI metrics are strong candidates for continued monitoring for trends assessment.

The water quality assessment found that individual metrics provide a better understanding of variation than the WQI. PAHs and metals were rarely detected in water. Monthly water quality sampling is probably more suited to targeted monitoring programs that sustain long-term monthly (and sometime targeted storm event) sampling aimed at identifying sources of pollution and evaluating the effectiveness of management practices focused on resolving specific problems.

The sediment quality assessment found that the limited number of sites adversely affected by contaminants most typically occurs within UGAs. These findings are consistent with monitoring elsewhere: concentrations of metals and other organic contaminants in sediment are generally low, although often found at elevated levels in urban areas, and below probable toxicity thresholds for benthic invertebrates. The Washington State Department of Agriculture (WSDA) collaborated with SAM to analyze sediment samples for an additional 100 pesticides at most the stream sites and those findings, including a toxicity analysis, will be published in a separate report. WSDA's report will help SAM identify future pesticide sampling priorities.

The habitat assessment found that stream substrate and bed stability were in poorer condition within UGAs and in better condition outside UGAs, consistent with a body of research on Puget Lowland streams that has linked urbanization to changes in substrate and bed stability. However, the degree of substrate alteration also depends on the local hydrogeology in which forest clearing and development occurred. The small differences in the condition of riparian canopy closure, wood volume, and residual pool areas within and outside UGAs may be due in part to historical management practices – now prevented by protective ordinances – that included clearing of riparian vegetation and removal of wood from streams and riparian areas. These data should continue to be collected as part of the B-IBI and TDI sampling program.

The correlation assessment found that biological metrics are not sensitive to natural physical variables. Biological response metrics that are not sensitive to natural physical variables are more precise measures of human influence. The greatest attributable risk for low B-IBI scores in the Puget Lowland ecoregion streams was found to be watershed canopy cover with an attributable risk of 59%.

The B-IBI relative risk/attributable assessment findings are generally consistent with previous studies. Low B-IBI scores were most associated with land use/land cover factors indicating various aspects of urbanization and clearing. Substrate measures were nominally insignificant, despite these parameters previously being identified in other studies as having the highest attributable risk to B-IBI scores. Total nitrogen was the only other statistically significant stressor identified by this assessment.

The comparisons of SAM to other Puget Lowland monitoring programs found that SAM is a comprehensive program compared to most, with many more sites and parameters. However, other programs include measures of streamflow or stage measurements that SAM is lacking. Most sampling protocols were the same or similar, but where differences are known to occur they are meaningful. Probabilistic and targeted monitoring programs are answering different questions. To continue to understand whether conditions in Puget Lowland ecoregion streams are improving or deteriorating, we recommend continuing the probabilistic design to provide unbiased estimates of status and trends. Local programs are conducted on a limited scale and their status and particularly their trend findings cannot be extrapolated to the regional Puget Lowland scale. The regional probabilistic sampling framework of the SAM small streams status and trends study provides estimates of population statistics that can be used to put more local sampling information into a regional context. This can be explored further in advance of future sampling.

Moving forward, an approach of only minimal changes to the initial design can be considered. The first round of monitoring was a general success and achieved several goals: (1) it established the status of small streams in the Puget Lowlands, (2) it covered a range of urbanization across Puget Lowland ecoregion stream watersheds, and (3) it supported identification of factors associated with poor and good biological condition. Since 2015 was a drought year, repeating essentially the same study a second time may be important. With some small changes to the existing program, such as dropping monthly water quality sampling and the WQI, and adding continuous monitoring of stage, the current program will address the study questions more effectively.

On the other hand, because the primary purpose of this program is to assess trends, strategic increases in the frequency of sampling are needed to evaluate trends as soon as possible. Furthermore, adjustments to the strata adding categories that span the range of developed conditions would provide discrete information about stream conditions that is more useful to stormwater managers than simply tracking changes and comparing conditions within and outside UGAs. These modifications to the design should be included to the extent possible within budgetary constraints.

Scientific Recommendations

This section addresses the SWG's question: Which measures should be carried forward to assess trend, and at what frequency? Furthermore, this section articulates which aspects of the 2015 study design have provided the most meaningful information, the components that do not seem to work well to address the main study questions, the outstanding information needs, and some discrete methods that can be used to address those needs. Further discussion will be needed to finalize the study design in advance of the next sampling effort.

<u>Overall design</u>: Continue the regional probabilistic sampling design and approach. Consider a rotating sampling approach. Look for opportunities to include existing targeted sites or a subset of the probabilistic sites to answer special questions. In near future rounds of monitoring, focus on answering the following questions:

- What is the condition of small Puget Lowland ecoregion streams over the full spectrum of development?
- What measured stressors or other indicators are associated with poor or good biological conditions?
- Are conditions measurably changing over time?

The SWG will determine whether adjustments made to the design for status and trends monitoring will be minimal or whether SAM will pursue additional strategic modifications for more frequent sampling and for redefining the sampling strata.

<u>Site selection</u>: The final number of sites to sample and their frequency of sampling should be determined following additional power analysis. Revised budget estimates will help inform final recommendations, but plan for 100 to 150 sites. Specifically:

- Maintain as many of the 2015 sites as is practicable to improve power to detect trends while ensuring all of the sites meet new target criteria. Eliminate the occurrence of nested basins. Identify SAM reference sites and ensure they are sampled following SAM streams criteria and approach.
- Consider a new approach to stratification that captures the entire range of development: in place of within and outside UGAs, stratify the sites by a highly correlating urbanization characteristic (*i.e.*, percent imperviousness or watershed canopy) and fill in sites as needed.
- Consider using the approach proposed for the Lower Columbia urban streams monitoring: convert probabilistic sampling points into probabilistic stream reaches and target a certain range of watershed size.
- If the current strata, defined as within and outside UGAs, continues in future rounds:
    - Reassign sites located at the outer edges of the UGA to the outside UGA stratum that the sites actually represent with their watershed drainage areas.
    - Define how to address UGA boundaries changing over time, both the process to adaptively balance the number of streams in each strata and how to approach the data analysis.
- Engage statistical expertise in determining the data analysis approach to be followed with each of these changes.

<u>Parameters measured and sampling frequency</u>: Drop monthly water quality sampling and add an assessment of hydrology. Continue using similar methods. Specifically:

- Hydrology parameters: Add continuous monitoring for stage at all watershed health sites where existing stream gaging is not providing this data. Plan for management of these data to allow for easy analysis and sharing.
- Biological parameters: Continue using B-IBI and periphyton to assess biological condition. Because the primary objective of this program is to detect trends, to detect trends sooner conduct the watershed health sampling more frequently than once per five years. Increase the number of replicate samples to improve understanding of variance.
- Water quality parameters: Discontinue monthly water quality sampling. Add continuous monitoring for temperature with the stage monitoring; also consider adding turbidity and/or conductivity. Sample metals, nutrients, fecal coliform, turbidity and total suspended solids only annually during the watershed health sampling or add an index period for sampling. Consider a spring season sampling for pesticides that were detected in the recent USGS PNSQA study.
- Sediment quality parameters: Continue analyzing for TOC, selected metals, PAHs (using a more sensitive method), and – if budget allows – PBDEs. Keep/add selected pesticides based on the findings of the Washington State Department of Agriculture (WSDA) analysis. Continue sieving samples prior to analysis.
- Habitat parameters: Continue to collect these data as part of the Watershed Health sampling.

Data analysis and reporting: Continue the detailed analytical approaches used in this assessment, and in particular the relative risk assessment. The analysis of land cover and other GIS metrics should be repeated every permit cycle to look at change over time.

Special studies: Prioritize and conduct additional investigations that improve our collective understanding of stormwater impacts on Puget Lowland ecoregion streams. For instance, conduct some water sampling during storms and base flow. Continue to identify key stressors for biological communities.

1.0 OVERVIEW OF THE STORMWATER ACTION MONITORING PUGET LOWLAND STREAM STUDY

The Stormwater Action Monitoring (SAM) program is a collaborative monitoring program funded directly by Western Washington municipal stormwater permittees, with additional in-kind funds from Washington state agencies and cities, federal agencies, and local business. The Stormwater Work Group (SWG), a formal stakeholder group, developed a regional stormwater monitoring strategy (SWG, 2010a) to improve our understanding of the effects of stormwater in the Puget Sound region; SAM is the resulting regional monitoring program formed by this effort. The SWG provides leadership and oversight on SAM, and Ecology is the service provider for these funds.

SAM's scientific framework has three primary parts; stormwater management effectiveness studies, source identification, and receiving water monitoring. The latter has two sensitive environments where SAM status and trend monitoring efforts are focused: the nearshore marine environment and small Puget Sound streams potentially impacted by stormwater runoff. This is the first status assessment of Puget Lowland ecoregion streams (PLES) by the SAM program.

The SAM PLES study frame is small, low order (1^{st} - 3^{rd} order), perennial streams of the Puget Lowland ecoregion (Omernik and Griffith, 2014) draining to Puget Sound. To provide unbiased estimates of status and trend of streams conditions, SWG recommended a probabilistic survey design, which can be compatible with the current statewide status and trends monitoring program. The study characterized approximately 1,670 stream miles in 2015. Study findings are shareable across the region, across watershed boundaries, and across jurisdictional lines because of the random probabilistic study design on an ecologically important aquatic environment. Significance of these findings is described in this report and represents the initial assessment of current condition or "status."

Recommendations from this effort will help shape the future SAM PLES status and trend monitoring program. The SAM PLES program is intended to be a long-term effort to improve our collective understanding of whether stormwater management programs are helping to achieve the larger goal of restoring the Puget Sound ecosystem. Observed impacts and changes over time are expected to reflect multiple stressors and the effect of multiple management actions (SWG, 2010a). The SAM PLES status and trends program provides a platform for measuring small streams in urban and rural areas with active stormwater management. The goal is to measure changes of receiving water conditions as a result of stormwater management. The results of the SAM PLES status and trends program can inform and improve stormwater management strategies and support effective management approaches.

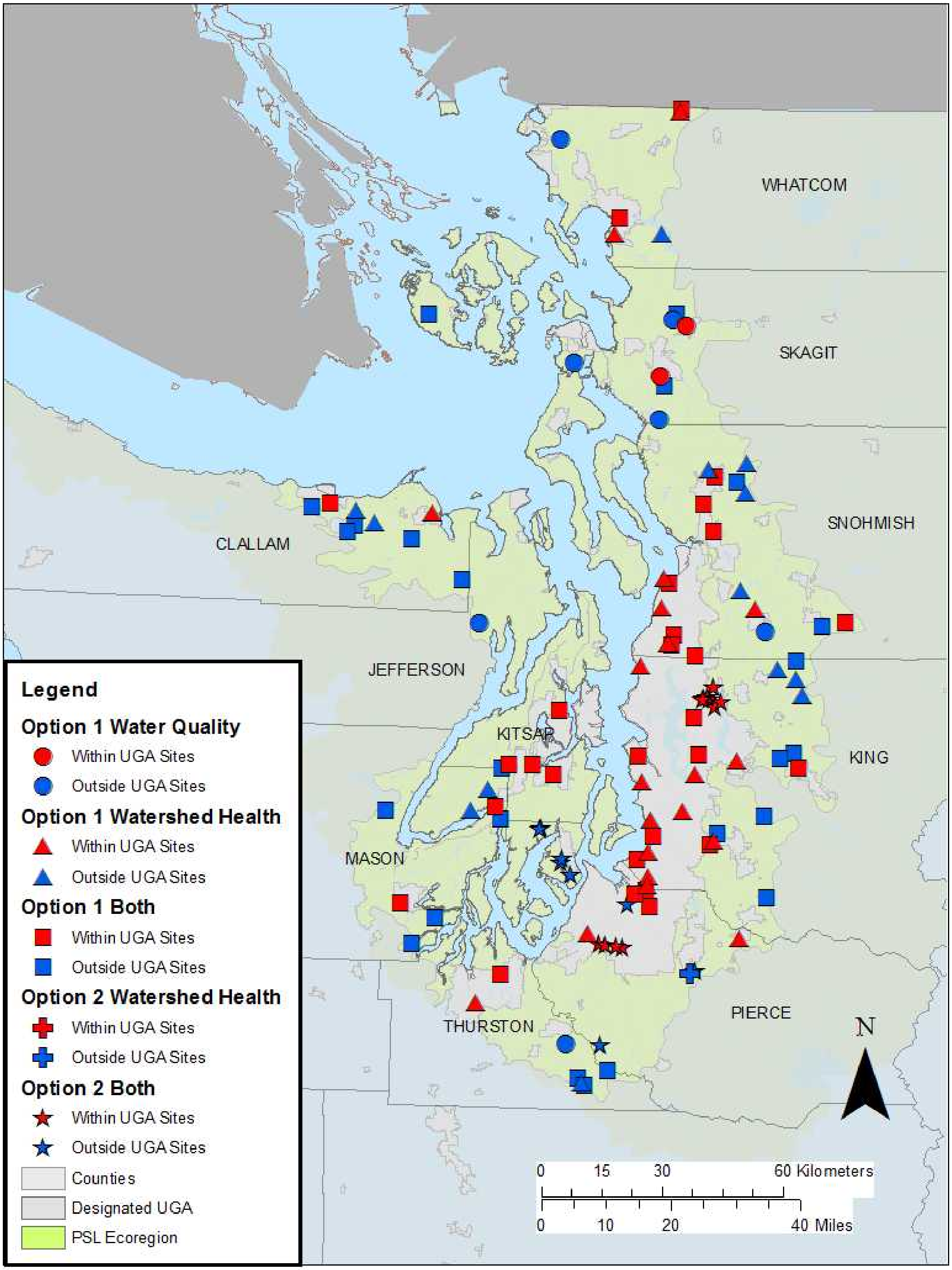

Figure 1. Puget Lowland ecoregion streams sites sampled in 2015 under Stormwater Action Monitoring (SAM Option 1) and Option 2 (permit alternative) monitoring. Sediment quality, biota, and habitat measures were collected at watershed health sites.

This report summarizes findings from one year, 2015, sampling of small streams in the SAM PLES study area (Figure 1). Sampling was paid for by pooling funds for regional monitoring (Option 1) from municipal stormwater permittees. In addition, we also use Option 2 results for comparison or in-combination with Option 1 results to describe the findings.

> Note: Most Western Washington Phase I and II Municipal Stormwater Permittees chose to participate via the cost sharing program ("Option 1" in special condition S8.B in the 2013 Phase I and Phase II western Washington municipal stormwater permits). Washington State Department of Transportation opted to contribute funds for SAM receiving water monitoring in 2014 to comply with their separate permit requirements.

We use "SAM PLES" to refer to results from the full sampling frame, which represent the combination of data from the SAM regional sites (Option 1) and Option 2 sites. Incorporating the Option 2 data is accommodated in the probabilistic framework of the survey design (Larsen et al., 2008), by assigning a spatial weight factor to each stream site representing the extent of stream length that the site represents. As part of analysis, the spatial weights must be adjusted based on the successful number of sites sampled in the study. A more in-depth discussion of the spatial weights is covered in Section 2.3 and Section 3.1.

1.1 Goals and Objectives

The study aimed to characterize the status of small streams impacted by stormwater and measure whether stream conditions are getting better or worse over time. Given that development is focused inside Urban Growth Area (UGAs), the PLES study differentiated Puget Lowland streams into two strata: within UGAs and outside of UGAs. These objectives can be summarized as two questions (Ecology, 2014):

- What are the status and trends of instream water quality, biological, and habitat conditions for 1st, 2nd, and 3rd order (small) streams in Puget Lowland region?
- What are the status and trends of the water quality, biological, and habitat conditions for 1st, 2nd, and 3rd order (small) streams in Puget Lowland, both inside and outside of Cities/UGAs?

The scope of this project and report are larger than just a scientific status assessment of small streams. In addition, the SWG wanted the results of the first year of monitoring to be placed in context to other types of existing monitoring program, and also provide scientific recommendations for the design of the long-term status and trend program. To that end, this report consists of five components: (1) streams status assessment of Puget Lowland stream quality and health, (2) comparison of data to other Puget Sound stream monitoring programs, (3) review of regional monitoring programs in other parts of the country, (4) an initial evaluation of data and methods that can be used to evaluate trend detection power of particular monitoring designs, and (5) scientific recommendations for future SAM PLES status and trend monitoring.

1.1.1 Streams Status Year of Monitoring

The questions above were further refined into the following to guide the analyses conducted for this report:

- **Q1:** What percent of streams meet biological, water, and sediment quality standards for beneficial uses within and outside urban growth areas (UGAs)?
- **Q2:** What natural variables correlate with the status of streams within and outside UGAs?
- **Q3:** What human variables correlate with the status of streams within and outside UGAs?

1.1.2 Data Comparisons to Other Local/Regional Programs

The SWG recommended we also compare results of this status assessment survey to other Puget Lowland stream monitoring programs. Specifically we describe these questions as:

- **Q4:** How do SAM PLES results compare to other stream monitoring programs in the Puget Sound lowlands?
  - **Q4.1:** Compare to other probabilistic sampling programs.
  - **Q4.2:** Compare to targeted sampling programs.

1.1.3 Considerations for SAM PLES Status and Trend study Design

Lastly, the SWG wanted scientific recommendations from the first year's status assessment to inform the SAM PLES status and trend monitoring effort. Specifically:

- **Q5:** What water, sediment, biological and habitat parameters should be carried forward for SAM PLES monitoring in the future, and at what timing and frequency?

1.2 How to Navigate this Report

An overview of the study methods (study area, spatial study design, landscape/land cover data, parameters measured, development of screening thresholds, and statistical data analysis methods) is provided in Section 2.0. Section 3.0 of this report provides the results for the three status assessment questions. The results of the implementation of the study design are provided in Section 3.1. A summary of the results of the geographic information system (GIS) landscape/land cover analyses conducted by the U.S. Geological Survey (USGS) is provided in Section 3.2. A complete status assessment is provided in Section 3.3. The results of the statistical analyses performed to identify what natural and human variables correlate with the observed status of streams is provided in Section 3.4.

The 2015 SAM PLES results are compared to other Puget Lowland monitoring programs (Section 4.0) and other regional status and trends monitoring programs are summarized in Section 5.0. The development of statistical tools to evaluate the power of various survey designs to detect trends is described in Section 6.0. Recommendations for future SAM PLES status and trend monitoring are provided in Section 7.0.

A summary of laboratory detection frequency is provided in Appendix A of this report. Statistical summaries of the 2015 study data within and outside UGAs are provided in Appendix B for each data type as indicated below:

- Benthic invertebrate metrics
- Periphyton metrics
- Water Quality Index (annual and monthly)
- In situ water quality data
- Laboratory water quality data
- Laboratory sediment quality data
- Stream habitat metrics
- Landscape GIS data

The results of all cumulative distribution frequency analyses (a component of the status assessment), including the results of statistical comparisons of streams within versus outside UGAs are provided in Appendix C for the same data types as Appendix B.

2.0 METHODS

2.1 Study Area

The 2015 SAM PLES study area encompassed the Puget Lowland ecoregion covering approximately 13,900 km^2 (5,361 mi^2) with 6,479 km^2 (2,501 mi^2) within UGAs and 7,406 km^2 (2,859 mi^2) outside of UGAs. The Puget Sound region continues to experience pressures associated with development and a growing population. The Puget Sound Regional Council compiles data from the U.S. Census Bureau which indicates that the population in Central Puget Sound (King, Kitsap, Pierce, Snohomish counties) in 2015 was almost 3.9 million with about 70 percent living within incorporated areas.[1] Population in the same region is forecasted to reach almost 5 million by 2040[1] which is a 22 percent growth rate over 25 years. The continued influx of people and associated development will continue to exert pressure on ecosystems. Land use policies surrounding the state's Growth Management Act aim to focus population growth within UGAs in order to minimize forest loss and development pressures on rural and resource lands outside of urban areas.[2]

2.2 Field Sampling and Laboratories

The plan was to collect Watershed Health (WH) parameters (biological, sediment, and stream habitat parameters) between July 1 and October 15, 2015 and monthly water quality samples from January-December, 2015. SAM regional sites (Option 1) targeted 100 sites total for WH parameters; 50 sites within and 50 sites outside UGAs, and a subset of 60 sites was targeted for the monthly water quality sampling; 30 within and 30 outside UGAs.

For Option 2 jurisdictions, the number of sites were for both WH and monthly water quality; 12 for Pierce County and 8 for City of Redmond. The timeframe was the same for WH but not monthly water quality sampling; the permit required monthly sampling between October 2014 and September 2015.

For the Option 1 pooled funds, field sampling was done by four separate field crews (San Juan Island Conservation District, Skagit County, King County and USGS and samples were sent to two primary laboratories: King County Environmental Lab (KCEL) and Manchester Environmental Lab (MEL). Due to short holding times for bacteria the SAM coordinator arranged for samples collected in Whatcom County and San Juan County to go to Edge Analytical in Bellingham, and bacteria samples from the Kitsap peninsula and further west to go to Clallam County laboratory. MEL subcontracted for high resolution organic contaminants. In February 2015, duplicates of 6 water and 10 sediment samples were collected for an inter-laboratory study to compare KCEL and MEL results for a select number of water parameters (ammonia, nitrate+nitrite, turbidity, dissolved metals and polycyclic aromatic hydrocarbons (PAHs)) and sediment parameters (metals and PAHs).

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[1] Puget Sound Regional Council: https://www.psrc.org/sites/default/files/trend-population-201607.pdf , https://www.psrc.org/regional-macroeconomic-forecast

[2] Puget Sound Partnership Vital Signs: Land Development and Cover: http://www.psp.wa.gov/vitalsigns/land_cover_and_development.php

For Option 2, City of Redmond and Pierce County conducted their own field sampling, and sent samples to the same laboratories for analysis.

2.3 Spatial Study Design

The SAM status and trends in receiving waters studies (this SAM PLES study and the marine nearshore mussels and marine nearshore sediment study) used a spatially-balanced probabilistic design to identify potential sampling locations called Generalized Random Tessellation Stratified (GRTS) survey design. The Puget Lowland ecoregion contains a total of 7,329 master sampling points, each representing 1 km of stream length, drawn by the GRTS technique.

The calendar year of 2015 turned out to be an unusual drought year in the Pacific Northwest[3] and some sites were too dry to sample during the year. A re-sampling was attempted, but at a few sites, it was often not possible to collect samples in every month. If more than three months of data were missed or if a site was dry in summer, then SAM WH sampling at that site was discontinued and the crews sampled the next site from the candidate list for WH. Given the dry conditions, the total number of sites sampled for summer WH and monthly water quality varied substantially from the intentions, and are listed in Table 1 and shown in Figure 1.

For WH a total of 105 sites were sampled; 48 sites within UGAs and 37 sites outside of UGAs for SAM. Pierce County and the City of Redmond sampled 20 of those sites for WH and water quality (Pierce County: 4 within UGAs and 9 outside UGAs; Redmond: 7 within UGAs). For monthly water quality a total of 61 SAM sites were initially sampled (Table 1). However, due to drought conditions, 24 outside UGA sites and 28 within UGA sites were considered complete enough for use in the status assessment. Of the 105 sites sampled for WH biological parameters during the study; 59 sites were within UGAs and 45 sites were outside of UGAs. For sediment quality and physical habitat, one more site was sampled outside UGAs.

Because of the spatially balanced, probabilistic framework used in the SAM PLES study, adjusted spatial weights could be calculated and used in the status assessment to allow inferences from site specific results to the length of small Puget Lowland ecoregion streams within and outside UGAs (for WH parameters). Combining Option 1 and Option 2 data for the WH status assessment is accommodated in the probabilistic framework of the survey design by assigning a spatial weight factor to each stream site representing the extent of stream length that the site represents. Because weights for water quality data analysis were not calculated for Option 2 sites (due to comparability of calendar and water year data collection periods), inferences regarding water quality status are for small streams within and outside UGAs, excluding streams within the City of Redmond or unincorporated

[3] A statewide drought emergency was declared by the governor in May 2015 due to historic low snowpack and air temperatures were frequently above normal during the winter of 2014-2015, spring and early summer of 2015 due to unusually warm surface water conditions in the Northeast Pacific ocean (Bond et al., 2015; Ecology, 2016).

Pierce County. Due to the different number of WH and monthly water quality sampling sites and sites that were determined to be non-target sites during initial reconnaissance and during the monitoring period, initial spatial weights were adjusted to reflect the estimated extent of target stream length sampled. The adjusted spatial weight of each site and the total stream length represented by this monitoring effort was calculated using the spsurvey package in R (Kincaid and Olsen, 2015).

Table 1. Number of Watershed Health and Monthly Water Quality sties sampled by Option and Strata.

	SAM regional sites (Option 1)		Pierce County (Option 2)		City of Redmond (Option 2)	
Strata	OUGA	WUGA	OUGA	WUGA	OUGA	WUGA
WH	13	20	1	-	-	-
WQ + WH	24	28	8	4	-	7
WQ only	7	2		-	-	-

WH = Watershed Health Monitoring; WQ = Monthly Water Quality Monitoring
Study ID in Ecology EIM database: SAM_PLES for SAM, RSMP_PC_PLES2015 for Pierce County, and RSMP_RD_PLES2015 for City of Redmond

2.4 Landscape Data

The goal of landscape data collection was to provide a basis for identifying natural landscape and anthropogenic factors influencing stream health in the region. To meet this goal, a framework of fundamental geospatial data was required to develop physical and anthropogenic characteristics of the sampled sites and corresponding watersheds and riparian zones. Landscape data were compiled for 105 SAM PLES basins and 16 Puget Lowland reference (least-disturbed) basins monitored by Ecology (more information regarding reference conditions follows below). The basins represent the sites sampled for WH parameters. Landscape data were derived through GIS analyses conducted by the USGS. The resulting GIS data are provided on ScienceBase at https://doi.org/10.5066/F7JQ0Z80 (Sheibley et al., 2017a).

Watershed boundaries were delineated using the online interactive map application from the StreamStats Program and compared to the national Watershed Boundary Dataset.[4] In cases where there were significant differences between the two sets of boundaries, manual corrections to the watersheds were made. The riparian-zone boundaries were created from stream centerlines digitized from imagery that were buffered by 50 m on each side of the stream centerline. The along stream length of the digitized riparian reach upstream from the sampling point was calculated as the distance in kilometers equal to the base-10 logarithm of the total watershed area in kilometers squared (Johnson and Zelt, 2005).

[4] StreamStats: https://water.usgs.gov/osw/streamstats/

Environmental characteristics describing physical and anthropogenic characteristics of the study region were identified in the watershed and riparian zones upstream of the sampling sites. A total set of 116 environmental characteristics that included both natural physical and human disturbance variables was calculated using ArcGIS. These variables include data on basin geology, precipitation, soils, land cover, basin slope and elevation, drainage area, measures of urbanization (population density, imperviousness, etc.), and nutrient loading. The tools that were used to geoprocess the watershed and riparian characteristics were elements of either ArcToolbox version 10.1, a component of ArcGIS for Desktop (Esri, 2014) or the National Water Quality Assessment (NAWQA) Area-Characterization Toolbox ("NACT Toolbox") version 2.0 originally published by Price et al. (2010).

All of the landscape data were included in statistical analyses (described in Sections 2.6.1 and 2.7 below), but results presented in this report are limited to final results or to highlight specific points. The landscape and land cover metrics presented in the report are summarized in Table 2.

Table 2. Selected landscape data presented in this report.

Variable Name	Variable Abbreviation	Description	Units
Watershed Drainage Area [a]	GISAreaKm2	Watershed area for site, derived in a GIS	square kilometers
Site Longitude	lonECO2use	Geographic longitude coordinate of the sample site	decimal degrees
Watershed Mean December Precipitation [b]	PPT_Dec_1982_2014	Mean of mean December precipitation in the watershed for 33 years of record, 1982-2014	millimeters
Watershed Areal Nitrogen Loading Rate [c]	N_Tot2002_kgkm2	Estimated total nitrogen application rate from fertilizer, manure, and wastewater in watershed, 2002	kilograms per square kilometer
Watershed %Urban Development [d]	UrbanLMH2011	Sum of "DevelopedLow2011","DevelopedMed2011", and "DevelopedHigh2011".	percentage
Watershed %Agriculture [d]	AgricultureTotal2011	Sum of "PastureHay2011" and "CultivatedCrops2011".	percentage
Watershed %Forest [d]	ForestTotal2011	Sum of "DeciduousForest2011", "EvergreenForest2011", and "MixedForest2011".	percentage
Watershed %Wetlands [d]	WetlandsTotal2011	Sum of "WoodyWetlands2011" and "EmerHerbWetlands2011".	percentage
Watershed %Canopy Cover [e]	Canopy2011	Mean percentage of tree canopy area in watershed, from the National Land Cover Dataset 2011	percentage
Watershed High Intensity Development [d]	DevelopedHigh2011	Percentage of developed-high intensity in watershed, from the National Land Cover Dataset 2011-class 24	percentage

Variable Name	Variable Abbreviation	Description	Units
Watershed House Density [e]	HouseDensityBLK2010	Mean housing unit density (from Census block geography) in watershed, 2010	housing units per square kilometer
Riparian %Urban Development [d]	r50_UrbanLMH2011	Sum of "r50_DevelopedLow2011", "r50_DevelopedMed2011", and "r50_DevelopedHigh2011"	percentage
Riparian %Agriculture [d]	r50_AgricultureTotal2011	Sum of "r50_Pasture_Hay2011" and "r50_CultivatedCrops2011"	percentage
Riparian %Forest [d]	r50_ForestTotal2011	Sum of "r50_DeciduousForest2011", "r50_EvergreenForest2011", and "r50_MixedForest2011"	percentage
Riparian %Wetlands [d]	r50_WetlandsTotal2011	Sum of "r50_WoodyWetlands2011" and "r50_EmerHerbWetlands2011"	percentage
Riparian %Canopy Cover [f]	r50_CanopyCover2011	Mean percentage of tree canopy area in riparian zone (50-m buffer of the digitized ecological reach), from the National Land Cover Dataset 2011	percentage

[a] *Sheibley et al. (2017a): https://doi.org/10.5066/F7JQ0Z80*
[b] *Prism Climate Group (2014)*
[c] *Homer et al. (2007); Maupin and Ivahnenko (2011); Gronberg (2012); Gronberg and Spahr (2012); Mueller and Gronberg (2013); Wise and Johnson (2013); USGS (2014a)*
[d] *Homer et al. (2015); USGS (2014b)*
[e] *U.S. Census Bureau (2012)*
[f] *USGS (2014c)*

2.5 Parameters

The SAM PLES study included monitoring of water and sediment quality, physical stream habitat, and biological communities (benthic macroinvertebrates and periphyton) in small streams of the Puget Lowland ecoregion of the Puget Sound basin. Stream monitoring followed the Quality Assurance Project Plan (QAPP), Appendices, and QAPP addendum (Ecology, 2014).

Water quality sampling was conducted monthly for the parameters listed in Table 3. Water quality parameters measured included instantaneous stream discharge (flow), field instrument measurements, and laboratory measurements for some parameters (Table 3). Overall, over 50 parameters were measured, including:

- Conventional parameters: fecal coliform bacteria, turbidity, hardness, chloride, total suspended solids, dissolved organic carbon, and in situ measurements of dissolved oxygen, pH, temperature, and specific conductance.
- Nutrients: ammonia, nitrate+nitrite, total nitrogen, orthophosphate, and total phosphorus.
- Metals: (total and dissolved) arsenic, cadmium, chromium, copper, lead, silver, and zinc.
- Organic contaminants: PAHs.

One of the reasons monthly water quality sampling was conducted was to evaluate water quality status with respect to the Water Quality Index (WQI). The WQI is a unitless index developed by Ecology to provide a broad measure of overall water quality status of monitored streams across the entire state (Hallock, 2002). The index represents an aggregation of eight parameters (temperature, dissolved oxygen, pH, fecal coliform bacteria, total nitrogen, total phosphorus, total suspended solids and turbidity) and the resulting index can range from 0-100. Higher values are indicative of better water quality. Scores above 80 meet expectations for water quality and below 40 is indicative of a stream of high concern. The WQIs for each constituent are calculated for each month and then aggregated by calculating a simple average and subtracting a penalty factor for constituent monthly scores less than 80. The penalty factor is intended to weight low-scoring constituents more heavily so they are not masked by the averaging process. The overall annual WQI score (evaluated in this study) is the average of the three lowest scoring months during the year.

Table 3. Monthly water quality parameters measured during the study.

Monthly water quality monitoring	Parameters
Field in-situ measurement	temperature, pH, dissolved oxygen, specific conductance, stream discharge
Laboratory analysis	ammonia, chloride, dissolved organic carbon, hardness, fecal coliform, metals (total and dissolved arsenic, cadmium, chromium, copper, lead, silver, zinc), nitrate+nitrite nitrogen, polycyclic aromatic hydrocarbons (PAHs), orthophosphate phosphorus, total nitrogen, total phosphorus, total suspended solids, turbidity

The ecological condition of study streams was determined by analyzing the composition and relative abundance of two key biological assemblages calculated using WH parameters –benthic macroinvertebrates and periphyton (Table 4). The physical habitat parameters are described in Janisch (2013) and include metrics describing bank quality and stability, channel dimensions, fish cover, habitat dimensions, large woody debris, riparian cover, disturbance, and vegetation structure, stream sinuosity, and bottom substrate. WH parameters were measured once in summer between July 1 and October 15, 2015 at each site.

Table 4. Watershed Health parameters measured during the study.

Watershed Health Parameters
Benthic macroinvertebrates (identification and enumeration of taxa)
Periphyton (identification of taxa and analysis of chlorophyll a as a surrogate for areal periphyton biomass)
Physical habitat (discharge, slope and bearing, wetted width, bankfull width, bar width, substrate size, substrate depth, shade, human influence (e.g. presence of dike, pipes, or trash, any human activity), riparian vegetation, large woody debris)
*Water quality (field and laboratory measurements: pH, temperature, dissolved oxygen, specific conductance, total nitrogen, total phosphorus, chloride, turbidity)
Instantaneous stream discharge

**Water quality parameters in Table 4 were sampled under the WH protocol when a site was a WH only site. Otherwise, monthly water quality sampling suite in Table 3 was sampled.*

For stream benthic macroinvertebrates, the metrics selected for use in this study were:

- Benthic Index of Biotic Integrity (B-IBI; 0-100 scale) (King County, 2014a)
- Hilsenhoff Biotic Tolerance Index (HBTI) (Hilsenhoff, 1988)
- Fine Sediment Sensitivity Index (FSSI) (Reylea et al., 2012)
- Metals Tolerance Index (MTI) (McGuire, 1999)

The Puget Lowland 0 to100 scale Benthic Index of Biotic Integrity (B-IBI) (King County, 2014a) was selected because it has been shown to respond predictably to changes in environmental conditions of wadeable streams related to human disturbance. This makes B-IBI ideal for use as an ecological response indicator (Fore et al., 1996; Morley and Karr, 2002; Booth et al., 2004).

The three other metrics were chosen because of their potential diagnostic value (King County, 2015a). HBTI is an indicator of easily degraded organic matter pollution. The FSSI is an indicator of fine sediment problems). The MTI is an indicator of metal pollution. Higher HBTI scores indicate poorer biological condition due to easily oxidizable organic matter pollution (Hilsenhoff, 1988). Lower FSSI scores indicate a lack of invertebrate taxa sensitive to fine sediment (Reylea et al., 2012). Higher MTI scores (range 0-10) indicate poorer biological condition resulting from metal pollution (McGuire, 2009).

For stream periphyton, the biological response metric selected for use in this study was the 0-100 Trophic Diatom Index (TDI) developed as a tool for assessing stream and river eutrophication (Kelly and Whitton, 1995; Kelly, 1998). Lower Trophic Diatom Index values indicate generally less eutrophic (i.e., lower nutrient concentrations) stream conditions (Kelly and Whitton, 1995; Kelly, 1998).

Stream sediments (Table 5) were sampled during the same time window as the watershed health parameters (and are considered part of the WH monitoring component of this study). SAM sediment protocols followed USGS protocols that sieve sediment samples for the contaminant analysis (<2mm for organics and <63um for metals), which is a deviation from Ecology's WH protocols. Sieved sediment analyses are generally considered to result in higher concentrations, improved frequency of detection, less variance, and better represent exposure to aquatic organisms (Rickert et al., 1977; Horowitz, 1991). The sediment parameters measured included grain size, total organic carbon, percent solids, metals, PAHs, pesticides, phthalates, polychlorinated biphenyl (PCB) and polychlorinated biphenyl ether (PBDE) congeners.

Table 5. Sediment chemistry parameters measured during the study.

Sediment Parameters
Grain size (sieved < 2mm)
Total organic carbon (TOC) (both sieved fractions)[a,b]
Percent solids (both sieved fractions)[a,b]
Metals (arsenic, cadmium, chromium, copper, lead, silver, zinc) [a]
Polycyclic aromatic hydrocarbons (PAHs)[b]
Pesticides (2-4 D, carbaryl, chlorpyrifos, dichlobenil, diuron, triclopyr) [b]
Phthalates [b]
Polychlorinated biphenyls (PCBs)[b]
Polybrominated diphenyl ethers (PBDEs)[b]
Hormone disrupting chemicals: Pharmaceuticals and personal care products (PPCPs) and hormones and steroids (H/S) [c]

[a] *Metals were analyzed on a fine sample fraction that passed through a 0.063-mm sieve.*
[b] *Organic compounds were analyzed on a fine sample fraction that passed through a 2-mm sieve.*
[c] *SAM streams sediments were not analyzed for PPCPs and H/S based on budget decisions made by SWG.*

Washington State Department of Agriculture (WSDA) also analyzed sediment from 81 SAM PLES stream sites for 100 additional pesticides, pesticide degradates, and legacy pesticide compounds. A complimentary report on the results of these extra analyses is expected in the near future from WSDA.

2.6 Status Assessment

The assessment of stream condition, that is status assessment, followed the approach recommended by the U.S. Environmental Protection Agency (U.S. EPA) (Stoddard et al., 2005; U.S. EPA, 2006; U.S. EPA, 2016). This requires the development of thresholds based on regulatory standards, literature values, or data from "least-disturbed" sites as judged most appropriate for each measure (DeGasperi, 2016). These screening thresholds and their application were developed with input from local, state, and federal experts in freshwater and sediment contamination impacts on stream ecology including the Puget Sound Ecosystem Monitoring Program Toxics Workgroup and are summarized below for each metric analyzed in this report.

2.6.1 Least-Disturbed Reference Site Data

In order to assess the status or ecological stream condition, measures of what conditions would look like under relatively little human influence are needed. The "least-disturbed reference approach" to establish condition thresholds is commonly used by the U.S. EPA in similar studies across the United States, especially for measured parameters with no established regulatory criteria (specifically biological, habitat, and landscape GIS metrics).

We chose 16 Puget Lowland stream sites sampled annually since 2009 by Ecology (Ecology, 2010) and the U.S. EPA Region 10 (Herger et al., 2012) as our reference sites representing least disturbed conditions in the Puget Lowland ecoregion (Figure 2). GIS data were compiled for all 16 sites to provide reference condition levels for land cover data (available at ScienceBase at https://doi.org/10.5066/F7JQ0Z80; Sheibley et al., 2017a). Urban and agricultural area in the watersheds of reference sites were less than 10 percent of total land cover (Table 6).

Ecology and U.S. EPA generally followed the same protocols as used in the SAM small streams study for collecting and analyzing stream benthic invertebrates and stream habitat. However, the U.S EPA has not reported stream benthos data to Ecology's Environmental Information Management (EIM) database or to the Puget Sound Stream Benthos (PSSB) database and they did not sample their Sentinel sites for periphyton.[5] This reduced the total number of sites with biological reference data (for benthic invertebrates and periphyton) to 12 sites for the period 2009-2015. In 2014, Ecology began more consistent sampling of EPA Puget Lowland reference sites and biological data from those four sites will be readily available in the future.

[5] Ecology EIM: fortress.wa.gov/ecy/eimreporting/Default.aspx; PSSB: www.pugetsoundstreambenthos.org/

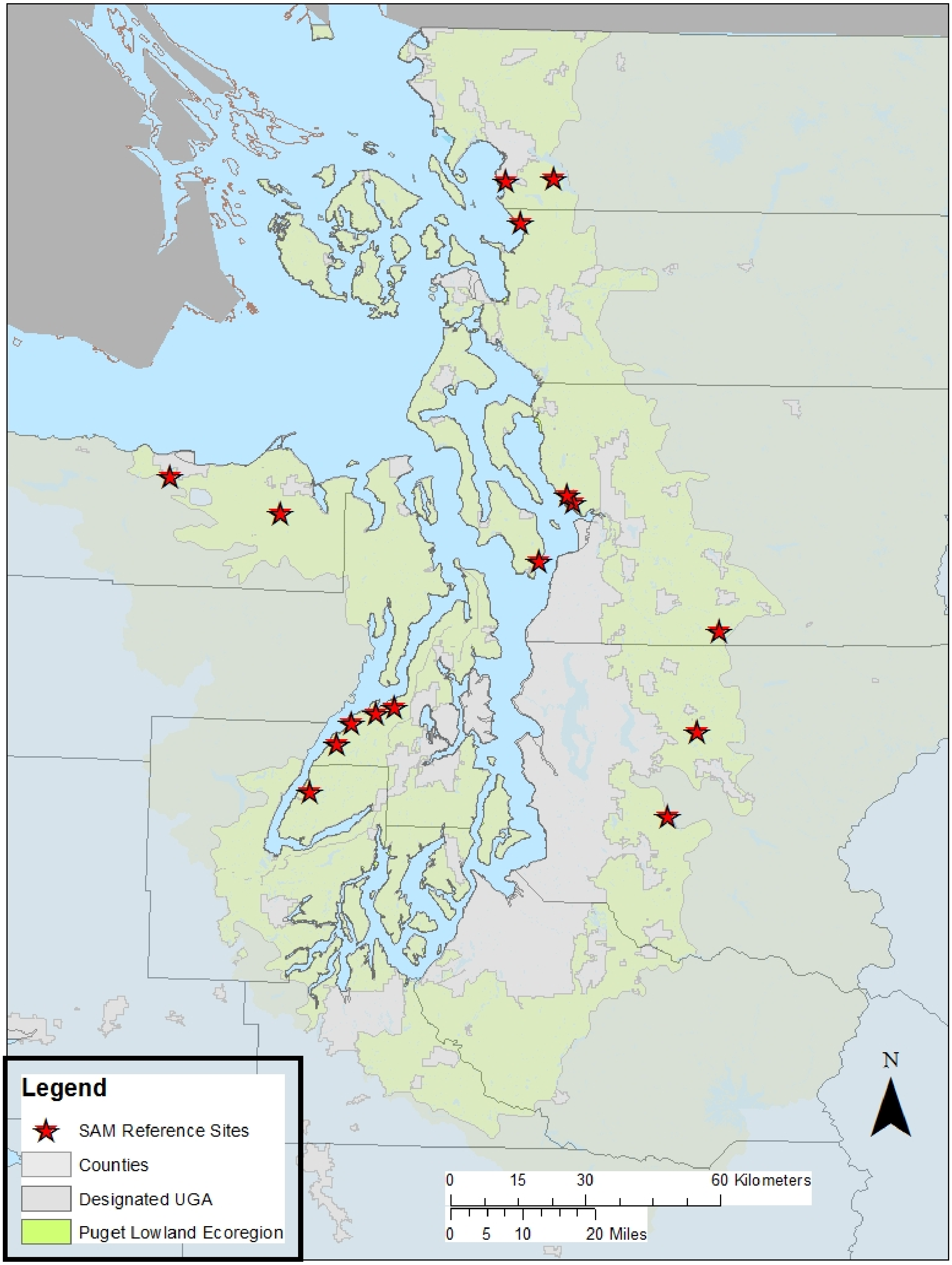


Figure 2. Map showing the locations of the 16 least-disturbed reference sites used to establish thresholds for use in the status assessment.

Table 6. Selected land cover characteristics of the 16 least-disturbed reference sites used in to establish thresholds for use in the status assessment.

Site	Riparian Buffer			Watershed		
	Urban	Agriculture	Forest	Urban	Agriculture	Forest
BIO06600-AUST02 (Austin Creek)	2.06	0.0	96.91	1.15	0.0	91.25
EPA06600-BATT01 (Battle Creek)	0.0	0.0	36.60	3.33	0.0	71.35
EPA06600-BEEF01 (Big Beef Creek)	0.0	0.0	97.22	6.23	0.0	71.63
BIO06600-BIGA02 (Anderson Creek)	0.0	0.0	76.38	0.65	0.0	51.59
BIO06600-BOYC02 (Boyce Creek)	0.0	0.0	100.0	2.30	0.0	81.42
BIO06600-CANY02 (Canyon Creek)	0.0	0.0	92.73	0.29	0.0	83.12
BIO06600-CHUD02 (Chuckanut Creek)	0.0	0.0	52.90	5.06	0.0	81.18
EPA06600-DEWA01 (Dewatto River)	2.65	0.0	62.43	0.72	0.0	62.79
EPA06600-GLEN01 (Glendale Creek)	3.57	0.0	77.38	2.63	9.58	71.12
SEN06600-GRIF09 (Griffin Creek)	0.0	0.0	81.52	0.33	0.0	62.69
BIO06600-HOLD02 (Holder Creek)	0.0	0.0	98.61	1.42	0.14	84.68
EPA06600-OYST01 (Oyster Creek)	0.0	0.0	51.85	0.02	0.0	91.18
BIO06600-SEAB02 (Seabeck Creek)	4.72	0.0	74.80	5.72	0.0	82.50
EPA06600-TULA01 (Tulalip Creek)	0.0	0.0	7.41	8.37	0.77	57.80
BIO06600-TUMW02 (Tumwater Creek)	0.0	0.0	99.08	0.70	2.77	76.89
BIO06600-YOUN02 (Youngs Creek)	0.0	0.0	75.00	0.42	0.0	65.41

Note: Urban land cover represents the sum of low, medium, and high intensity development categories.

Data for the habitat metrics and conventional water quality parameters (e.g., temperature, dissolved oxygen, pH, conductivity, total suspended solids, and turbidity) measured as part of the SAM streams study were available from all 16 sites. More limited data were available for nutrients. Sediment chemistry data (metals and PAHs) based on whole sediment analysis was available from only four of the Ecology Ambient monitoring sites and many of the PAHs were very infrequently detected. Ecologically relevant sediment chemistry thresholds were developed from other information sources as described in Section 2.6.3.

The reference levels developed from this data set are summarized in DeGasperi (2016). The development and application of least-disturbed reference site data to the SAM streams status assessment is described below in Section 2.6.3.

2.6.2 State Water and Sediment Quality Standards

In addition to the least-disturbed reference conditions for the ecological status assessment, the water and sediment quality data were compared to relevant state standards.[6] In this study, we used water quality standards for conventional parameters (temperature, dissolved oxygen, and pH), fecal coliform bacteria, and metals, and sediment quality standards for metals and organic contaminants.

[6] Water quality standards can be found in Chapter 173-201A Washington Administrative Code (WAC) and Sediment Quality Standards can be found in Chapter 173-204 WAC.

The water quality standards for temperature, pH, dissolved oxygen, and fecal coliform differ depending on the designated use of any particular stream reach. Designated uses are also different for specific parameters. For example, use designations for fecal coliform are based on protection of human health and different levels of exposure – primary vs secondary contact recreational exposure. Use designations for temperature, dissolved oxygen, and pH are based on the protection of aquatic life – specifically different salmonid spawning, rearing and migration requirements. Unfortunately, beneficial use designations are not part of the Washington Master Sample set of stream reach attributes. Therefore, a single threshold was chosen for each these water quality parameters in order to provide an assessment of status using the adjusted spatial weights calculated for use with the WQI.

The SAM PLES sampling sites were joined to Ecology's designated beneficial use layer using ESRI ArcMap geographic information system (GIS) tools. This exercise provided the site specific use designations for salmonids and human recreation use so that comparisons could be made explicitly, but did not allow for extrapolation to the entire represented stream lengths within and outside UGAs. We then tabulated the frequency of exceedance of standards based on site specific designated uses. These results were then compared to the results using a single threshold to evaluate if conclusions regarding status might be compromised if only a single threshold were used. If a more explicit evaluation of these water quality standards is desired in the next round of sampling, the Washington Master Sample will need to be populated with state designated salmonid and human use designations and incorporation of these designations into the design strata may be needed.

2.6.2.1 Water quality standards

Fecal coliform bacteria, water temperature, dissolved oxygen, and pH were compared to state water quality standards (WAC Chapter 173-201A). As noted above, these water quality standards differ depending on the designated use of any particular stream reach.

The state freshwater fecal coliform standards are summarized in Table 7. In this study, the primary contact recreation standard that fecal coliform levels must not exceed a geometric mean value of 100 colonies per 100 mL was selected as the threshold for use in the status assessment (see Section 2.6.3.2 below).

Table 7. Freshwater fecal coliform standards used in this study's ecological assessment.

Beneficial Use Category	Bacteria Indicator
Extraordinary Primary Contact Recreation	Fecal coliform organism levels must not exceed a geometric mean value of 50 colonies/100 mL, with not more than 10 percent of all samples (or any single sample when less than ten sample points exist) obtained for calculating the geometric mean value exceeding 100 colonies/100 mL.
Primary Contact Recreation	**Fecal coliform organism levels must not exceed a geometric mean value of 100 colonies /100 mL**, with not more than 10 percent of all samples (or any single sample when less

	than ten sample points exist) obtained for calculating the geometric mean value exceeding 200 colonies /100 mL.
Secondary Contact Recreation	Fecal coliform organism levels must not exceed a geometric mean value of 200 colonies/100 mL, with not more than 10 percent of all samples (or any single sample when less than ten sample points exist) obtained for calculating the geometric mean value exceeding 400 colonies /100 mL.

Note: Highlighted text indicates the threshold selected for use in the status assessment see Section 2.6.3.2 below.

The state freshwater temperature, dissolved oxygen, and pH standards are summarized in Table 8. The core summer salmonid habitat criteria for temperature, dissolved oxygen, and pH were selected for use as reference thresholds in the status assessment conducted for this study (see Section 2.6.3.2 below).

Water quality standards for temperature are meant to be applied to continuous (i.e., hourly or sub-hourly) measurements that would be used to calculate the highest 7-day average of the daily maximum temperature (7-DADMax). The SAM PLES temperature data are grab samples collected from near mid-day so do not likely represent the highest 7-DADMax that occurred at sampling sites in 2015. Dissolved oxygen and pH standards do not explicitly require continuous monitoring in order to compare data to the standards. However, continuous monitoring of small Puget Lowland streams has indicated that there can be a great deal of diurnal variation that might not be captured by monthly grab samples.

Table 8. Freshwater temperature, dissolved oxygen, and pH standards used in this study's ecological assessment.

Beneficial Use Category	Temperature (Highest 7-DADMax)	Dissolved Oxygen (lowest 1-Day Minimum)	pH
Char Spawning and Rearing *	12°C (53.6°F)	9.5 mg/L	pH shall be within the range of 6.5 to 8.5, with a human-caused variation within the above range of less than 0.2 units.
Core Summer Salmonid Habitat *	16°C (60.8°F)	9.5 mg/L	pH shall be within the range of 6.5 to 8.5, with a human-caused variation within the above range of less than 0.2 units..
Salmonid Spawning, Rearing, and Migration *	17.5°C (63.5°F)	8.0 mg/L	pH shall be within the range of 6.5 to 8.5 with a human-caused variation within the above range of less than 0.5 units.
Salmonid Rearing and Migration **Only**	17.5°C (63.5°F)	6.5 mg/L	Same as above.
Non-anadromous Interior Redband Trout	18°C (64.4°F)	8.0 mg/L	Same as above.
Indigenous Warm Water Species	20°C (68°F)	6.5 mg/L	Same as above.

7-DADMax = 7-day average of the daily maximum temperature.

Note: Highlighted text indicates the threshold selected for use in the status assessment see Section 2.6.3.2 below.

**Some streams have a more stringent temperature criterion that is applied seasonally to further protect salmonid spawning and egg incubation. See WAC 173-201A.*

2.6.2.2 Metal Standards

Comparisons to acute and chronic metals standards do not depend on reach-specific use designations. Instead, metals standards are dependent on the hardness measured in the same sample. Comparisons to acute and chronic metals standards used simultaneously measured hardness values to calculate the sample-specific standard to compare each result to. The standards assessed as part of this study are illustrated in Table 9.

Table 9. State metal standards used in this study's ecological assessment.

Parameter (units)	Acute	Chronic
Arsenic, Dissolved (ug/L)	360	190
Cadmium, Disssolved (ug/L) [a]	1.75	0.62
Chromium, Total (ug/L)	311	110
Copper, Dissolved (ug/L) [a]	8.86	6.28
Lead, Dissolved (ug/L) [a]	30.14	1.17
Silver, Dissolved (ug/L) [a]	1.05	na
Zinc, Dissolved (ug/L) [a]	63.61	58.09

[a] – Criteria are hardness dependent.
[b] – Trivalent or total recoverable chromium.
Note: All hardness dependent criteria based on a hardness of 50 mg/L.
Source: WAC 173-201A.

2.6.2.3 Sediment Chemistry Standards

Comparisons to relevant state sediment quality standards are considered conservative. The sediment quality standards were developed using whole sediment chemical analyses, whereas the sediment chemical analyses for the SAM PLES study were conducted on a sieved sediment fraction of fine sediment. Sieved sediment samples typically have higher concentrations of metals and organic contaminants relative to whole sediment analyses. We evaluated two freshwater sediment-benthic community thresholds: Sediment Cleanup Objectives and Cleanup Screening Levels (Table 10). Exceedance of the Sediment Screening Levels indicates a high potential for adverse effects to benthic invertebrates and can trigger a remediation effort to minimize toxic effects. Exceedances of the Sediment Cleanup levels are of moderate concern as these levels are "no-effects concentrations" that provide long-term sediment quality cleanup goals.

Table 10. State freshwater sediment standards used in this study's ecological assessment.

	Ecology SMS	
Parameter (units; Dry weight)	**Sediment Cleanup Objective**	**Cleanup Screening Level**
Di(2-ethylhexyl) phthalate (ug/Kg)	500	2200
Carbazole (ug/Kg)	900	1100
Dibenzo(a,h)anthracene (ug/Kg)	200	680
Dibutyl phthalate (ug/Kg)	380	1000
Di-n-octyl phthalate (ug/Kg)	39	1100
Arsenic (mg/Kg)	14	120
Cadmium (mg/Kg)	2.1	5.4
Chromium (mg/Kg)	72	88
Copper (mg/Kg)	400	1200
Lead (mg/Kg)	360	1300
Silver (mg/Kg)	0.57	1.7
Zinc (mg/Kg)	3200	4200
Total PAHs (ug/Kg)	17000	30000
Total PCBs (ug/Kg)	110	2500

2.6.3 Thresholds for the Status Assessment

The assessment of status followed the approach recommended by the U.S. EPA which required the development of thresholds based on regulatory standards, literature values, or data from "least-disturbed" Puget Sound Lowland reference sites (DeGasperi, 2016). The thresholds used in the status assessment presented in this report are summarized below.

2.6.3.1 Biological thresholds

Biological thresholds were developed from available Puget Sound lowlands reference site data, which is consistent with U.S. EPA and Ecology ecological assessment frameworks referenced above (Table 11). There are currently no state standards established for B-IBI, HBTI, FSSI, MTI, or TDI scores. The continuous gradient of the five biological response indicators was partitioned into three condition classes; Good, Fair, and Poor. The classes were defined to represent indicator ranges that are, with respect to the three condition classes, not different from (Good), somewhat different from (Fair), and markedly different from (Poor) the range of values sampled at the reference sites (Van Sickle et al., 2006; Van Sickle and Paulsen, 2008).

Table 11. Biological indicator thresholds developed for use in this study.

Metric	Poor	Fair	Good	Reference
Benthic Invertebrate Metrics				
B-IBI 0-100 (Distribution threshold by reference values)	<60.8		>77.4	Puget Sound reference (n=12)
Hilsenhoff Biotic Tolerance Index (HBTI)	>4.98		<4.55	Puget Sound reference (n=12)
Fine Sediment Sensitivity Index (FSSI)	<77		>110	Puget Sound reference (n=12)
Metals Tolerance Index (MTI)	>3.57		<2.267	Puget Sound reference (n=12)
Periphyton Metrics				
Trophic Diatom Index (TDI)	>61.1		<58.3	Puget Sound reference (n=12)

Because higher B-IBI and FSSI scores indicate improved condition, the lower 5th percentile of the reference site distribution was used as the threshold for Poor/Fair condition (i.e., <5th percentile classified as Poor). The 25th percentile was used as the threshold for Fair/Good condition (i.e., >25th percentile classified as Good). Higher HBTI, MTI, and TDI scores indicate worse condition. Therefore, the upper 95th percentile of the reference site distribution was used as the threshold for Poor/Fair condition (i.e., >95th percentile classified as Poor). The 75th percentile was used as the threshold for Fair/Good condition (i.e., <75th percentile classified as Good). The thresholds developed for this study's ecological assessment are provided in Table 11.

The reference levels of >4.98 and <4.55 for the Hilsenhoff Biotic Tolerance Index, indicative of poor and good conditions, respectively, determined in this assessment are consistent with the thresholds found in Hilsenhoff (1988). Hilsenhoff (1988) classified values of 5 or less as "Good" to "Excellent", with values from 5.01 to 5.75 indicating "fairly substantial pollution likely" (Fair) to 7.26-10.0 "Severe organic pollution likely" (Very Poor).

As currently used within the PSSB website, B-IBI scores less than 40 would be considered indicative of poor conditions. Based on data from least-disturbed Puget Lowland reference sites scores less than 60.8 are considered in this study to be in poor condition.[7] While the three categories we've presented (good, fair, poor) are different than those from the five categories in PSSB (very poor, poor, fair, good, excellent), they are consistent with similar status assessments conducted by King County (2014b), U.S. EPA (e.g., U.S. EPA, 2006), and Ecology (report in preparation) that rely on least-disturbed reference conditions to evaluate regional ecological conditions.

[7] Puget Sound Stream Benthos: http://pugetsoundstreambenthos.org/About-BIBI.aspx

2.6.3.2 Water Quality

No criteria have been established for the WQI and a complete year of monthly grab samples for the WQI component parameters has not been undertaken at Puget Sound reference sites. Therefore, the conventional thresholds developed by Ecology were used to classify the WQI into Good (lowest concern) and Poor (highest concern). The water quality thresholds used in this study are provided in Table 12.

Table 12. Water quality thresholds used in this study's ecological assessment.

Parameter	Poor	Fair	Good	Reference
WQI	<40		>=80	Ecology (see note)
Total Phosphorus (mg/L)	>0.050		<0.041	Hausmann et al. (2016); Puget Sound reference (n=14)
Total Nitrogen (mg/L)	>0.862		<0.459	Puget Sound reference (n=9)
	Poor		**Not Poor**	
Fecal Coliform (cfu/100 mL)	<100		>100	Primary Contact Recreation Standard (WAC 173-201A)
Temperature (^oC)	<16		>16	Core Summer Salmonid Habitat (WAC 173-201A)
Dissolved Oxygen (mg/L)	<9.5		>9.5	
Minimum pH	<8.5		>8.5	
Maximum pH	<6.5		>6.5	

Note: *Ecology WQI:* https://ecology.wa.gov/Research-Data/Monitoring-assessment/River-stream-monitoring/Water-quality-monitoring/River-stream-water-quality-index

Total phosphorus and total nitrogen were evaluated because of their relevance to the Trophic Diatom Index. Total phosphorus data were available from 14 of the 16 reference sites, which resulted in a threshold of 0.094 mg/L above which would be considered Poor. The threshold of 0.094 mg/L was considered to be too high based on recent research using a Biological Condition Gradient approach that indicated a threshold of 0.050 mg/L may be more appropriate (Hausmann et al., 2016). Therefore, a threshold of 0.050 mg/L was used in this study's ecological condition assessment. Concentrations above 0.050 mg/L were considered Poor and concentrations equal to or less than 0.041 mg/L were considered Good. Total nitrogen data were available from 9 of the 16 reference sites, which resulted in a threshold of 0.862 mg/L above which would be considered Poor and 0.459 mg/L below which would be considered Good.

The geometric mean fecal coliform concentrations of the monthly data from the 52 WQI sites was compared to the threshold. Sites with geometric mean concentration greater than 100 cfu/100 mL were categorized as Poor and sites with a concentration less than or equal to 100 cfu/100 mL were categorized as Good.

The maximum temperature measured at the 52 WQI sites during the study was compared to the 16 ^oC threshold to classify streams in Poor condition (temperature greater than 16 ^oC) or in Good condition (less than or equal to 16 ^oC). The minimum dissolved oxygen concentration measured was compared to a threshold of 9.5 mg/L with minimum concentrations above 9.5 mg/L categorized as Good (<9.5 mg/L = Poor). The evaluation of pH required two evaluations, one comparison based on the minimum of pH observations compared to a pH of 6.5 and a second comparison based on the maximum observed pH compared to a pH threshold of 8.5. Values below the minimum or above the maximum were classified as Poor.

2.6.3.3 Sediment chemistry

We focused the sediment chemistry status assessment on six metals (arsenic, cadmium, chromium, copper, lead and zinc), three organic chemical compound groups (total PAH, total PCB, and total PBDE), and dichlobenil. These were the more frequently detected parameters in both strata (within and outside of UGAs). Ecology's WH reference site sediment chemistry data were available for a few of the same parameters. However, data were available from only four reference sites and many of the parameters were infrequently detected. The low frequency of detection may be in part due to the difference in protocols (reference site sediments were not sieved before analysis) and highlights the need for sieved sediment reference data using the same protocols.

MacDonald et al. (2002) was selected as the source of threshold values for sediment chemistry, which is consistent with a recent study that evaluated sediment quality measured in sieved stream sediments from seven U.S. metropolitan areas (Moran et al., 2012). [8] MacDonald et al. (2002) reported threshold effects concentrations (TECs) and probable effects concentrations (PECs) for metals and organic compounds based on published sediment quality guidelines that provided a reliable basis for assessing freshwater sediment quality conditions. The reliability of their TECs and PECs was based on an evaluation of 347 paired sediment chemistry and toxicity samples. The TEC concentration represents the concentration below which no adverse effect is expected to occur. Concentrations below the TEC threshold were classed as Good. The PEC concentration represents the concentration above which adverse effects are expected to occur so concentrations above this threshold were classed as Poor. Concentrations between these two thresholds were classed as Fair. The sediment quality thresholds used in the ecological assessment are presented in Table 13. Note that a threshold for dichlobenil, a common roadside use pesticide was detected frequently in this study, was not identified.

There is a potential for the PECs for some metals to be near to or lower than background concentrations. As a result of a change in the Sediment Management Standard rules in 2013, Ecology is in the process of establishing regional background concentrations for specific organic contaminants and metals.[9]

[8] Sieved samples are appropriate for comparison to the threshold screening values of MacDonald et al. (2002) used in this section, which are based on sieved sediment data. Sieved samples are not completely appropriate for comparison to the State Sediment Management Standards which are based on whole-sediment data. Regardless, comparisons to the State Sediment Management Standards are conservative (i.e., concentrations in sieved sediment samples are expected to be similar to or higher than those measured in the whole sediment).

[9] Sediment Regional Background: https://fortress.wa.gov/ecy/publications/SummaryPages/1309051.html

Table 13. Freshwater sediment quality thresholds used in this study's ecological assessment.

Parameter (units)	TEC (<TEC = Good)	PEC (>PEC = Poor)
Arsenic (mg/Kg)	9.79	33
Cadmium (mg/Kg)	0.99	4.98
Chromium (mg/Kg)	43.4	111
Copper (mg/Kg)	31.6	149
Lead (mg/Kg)	35.8	128
Zinc (mg/Kg)	121	459
Total PAH (mg/Kg)	1610	22800
Total PCB (µg/Kg)	59.8	676
Total PBDE (µg/Kg)	31	3,100
Dichlobenil	NA	NA

TEC = Threshold effects concentration; PEC = Probable effects concentration; NA = Threshold not available
Source: MacDonald et al. (2002)

2.6.3.4 Stream habitat

Stream habitat status was assessed based on Puget Lowland reference site conditions. Five of the approximately 146 stream habitat health metrics (not including individual wood size metrics) were selected for evaluation in this report. The selected metrics represent traditional reporting categories for this type of habitat data:

- Riparian Canopy Closure - Stream center densitometer measurement
- Wood - Wood volume normalized to a 100-m reach length
- Pools - Residual pool area
- Substrate - Median particle diameter
- Bed stability - Logarithm of relative bed stability

The selected reference thresholds for these metrics are summarized in Table 14.

Table 14. Stream habitat thresholds developed for use in this study.

Metric	Poor	Fair	Good	Reference
Riparian cover (Densiometer)	<66.1		>81.6	Puget Sound reference (n=16)
Wood Volume per 100 m	<8.3		>16.5	Puget Sound reference (n=16)
Residual pool area	<3.2		>6.4	Puget Sound reference (n=16)
Substrate median particle diameter (D50)	<0.65		>10.7	Puget Sound reference (n=16)
Logarithm of Relative Bed Stability (LRBS)	< -3.15		> -2.747	Puget Sound reference (n=16)

2.6.3.5 Landscape and Land Cover data

The status based on a selected number of landscape/land cover metrics was assessed based on Puget Lowland reference site conditions. Of the over 100 watershed and riparian metrics, four were selected for evaluation in this report. The selected metrics represent human landscape and land cover variables that were closely related to biological status (i.e., status of B-IBI and TDI). These landscape and land cover metrics were:

- Watershed percent urban development
- Watershed canopy cover
- Riparian canopy cover
- Areal nitrogen loading rate

The selected reference thresholds for these metrics are summarized in Table 15.

Table 15. Landscape data thresholds developed for use in this study.

Metric	Poor	Fair	Good	Reference
Watershed %urban development	>4.47		<0.03	Puget Sound reference (n=16)
Watershed canopy cover	<62.83		>66.50	Puget Sound reference (n=16)
Riparian canopy cover	<55.32		>67.17	Puget Sound reference (n=16)
Areal nitrogen loading (kg/km^2)	>376.27		<136.94	Puget Sound reference (n=16)

2.7 Data analysis

The following subsections describe the data analysis methods used in addressing questions 1 through 3 of this study. The first subsection describes the data summary methods and the appendix containing the summary statistics. The second subsection describes the methods used in the status assessment. The third subsection describes the methods used to compare the data to water and sediment quality standards. The fourth subsection describes the methods used to evaluate the correlation of biological response variables (B-IBI and Trophic Diatom Index) to natural and human variables and the methods used in the relative risk/attributable risk analysis.

2.7.1 Statistical Summaries

Laboratory water and sediment quality data that included non-detects (i.e., concentrations that could not be quantified because they were below the laboratory's detection limit) presented complications for calculating summary statistics. The approach to handling non-detect data followed the approach outlined by Hobbs et al. (2015). Essentially, each water or sediment quality parameter was categorized as Case A, Case B, or Case C, depending on the frequency of detection. Each case was then handled as follows using R (R Core Development Team, 2012) and the NADA package (Helsel, 2012; Lee, 2013):

Case A (data detected in at least 50 percent of the samples analyzed): Kaplan-Meier non-parametric method using the "cenfit" function in the NADA package for R.

Case B (frequency of detection between 20 and 50 percent): Regression on Order Statistics (ROS) method using the "cenros" function in the NADA package for R.

Case C (frequency of detection less than 20 percent): Data summarized by ranges as calculating any summary statistics would have been unreliable (Helsel, 2012).

Only Case A water and sediment quality data were tested for statistically significant differences between within and outside UGA results. Summary statistics and tests for statistically significant differences were also compiled for biological, WQI, habitat, and landscape/land cover data.

Permutation tests were used to test for differences in mean and median because permutation tests do not require assumptions regarding the underlying distribution of the data. Tests were conducted for differences in the mean and median, because the two tests answer different questions. Differences in the mean evaluate whether average conditions are different within and outside UGAs. Differences in the median evaluate whether the distribution of the data are different (e.g., does one group exhibit higher concentrations than another). Permutation tests for the difference in the mean and median were conducted using the coin package in R (Hothorn et al., 2008; Hothorn et al., 2015). Statistically significant differences were identified when the p-value was less than 0.05.

For the technical details regarding the statistical approaches outlined above, the reader is referred to Hothorn et al. (2008), Hothorn et al. (2015), Helsel (2012), Lee (2003), and Hobbs et al. (2015).

Statistical summaries included the quantification of the number of samples collected and analyzed, summary statistics (mean, standard deviation, median, and 25th, 75th, and 90th percentiles), and the result of the tests of differences in the mean and median values of parameters measured within and outside UGAs.

2.7.2 Status Assessment

Cumulative Distribution Frequency (CDF) analysis is part of probabilistic survey design analysis that extrapolates site specific results to the length of Puget Lowland ecoregion streams. CDFs are the most concise summary for any indicator across the entire range of the data set. They accurately represent the distribution of conditions for Puget Lowland ecoregion streams within and outside UGAs. The CDF plot uses the adjusted spatial weights to plot the percent of the strata length represented by the study. Confidence intervals for each CDF provide a statistical basis for assessing differences between two CDFs. Here we evaluate differences within and outside UGAs, but the same approach could be used to evaluate differences between surveys conducted during two different points in time. The R package spsurvey (Kincaid and Olsen, 2015) was used to generate CDF plots for selected metrics measured within and outside UGAs. Nahorniak (2012) provided useful guidance on the development of R scripts using spsurvey to generate CDF plots and perform categorical analyses. The Wald F test was used to identify statistically significant differences between CDFs (within vs. outside UGAs) based on the recommendation in Kincaid and Olsen (2012).

Figure 3 provides an illustrative example of how to read the CDF plot. The y-axis describes the percentage of the target population (i.e., length of target streams) that is less than or equal to any particular value of the metric shown on the x-axis (Kincaid and Olsen, 2012).

In this example (Figure 3) we see that both the median and 75th percentile for all the streams sampled were below the higher threshold for "Metric A." From where a particular threshold passes through the CDF, a line perpendicular to this point can be drawn to the left or right axis to find the information needed to estimate the length of stream with values above, below or between thresholds (see Figure 3).

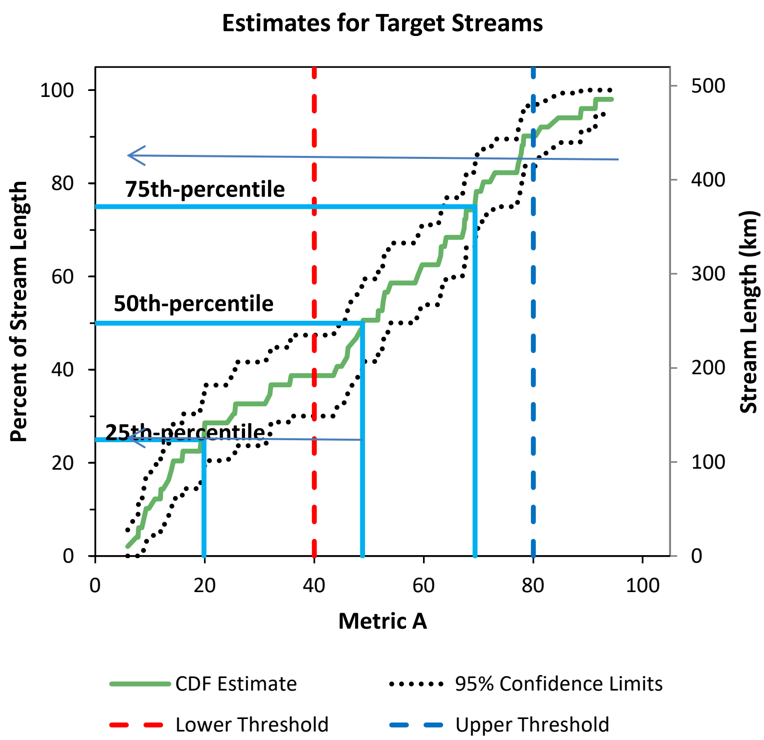


Figure 3. Cumulative distribution function (CDF) plot for a hypothetical metric, including 95% confidence limits of CDF.

In addition to providing complete information about the distribution of a particular metric, CDF plots can be readily transformed into a categorical analysis (i.e., condition or status assessment) using thresholds established by using data from reference sites, literature values, or by using regulatory standards as noted above. Example upper and lower thresholds are also shown to illustrate how the CDF can be categorized into the length of streams in poor condition (i.e., below the lower threshold), fair condition (between the lower and upper threshold), and good condition (above the upper threshold). Figure 4 provides an example from this report to show how the categories are developed.

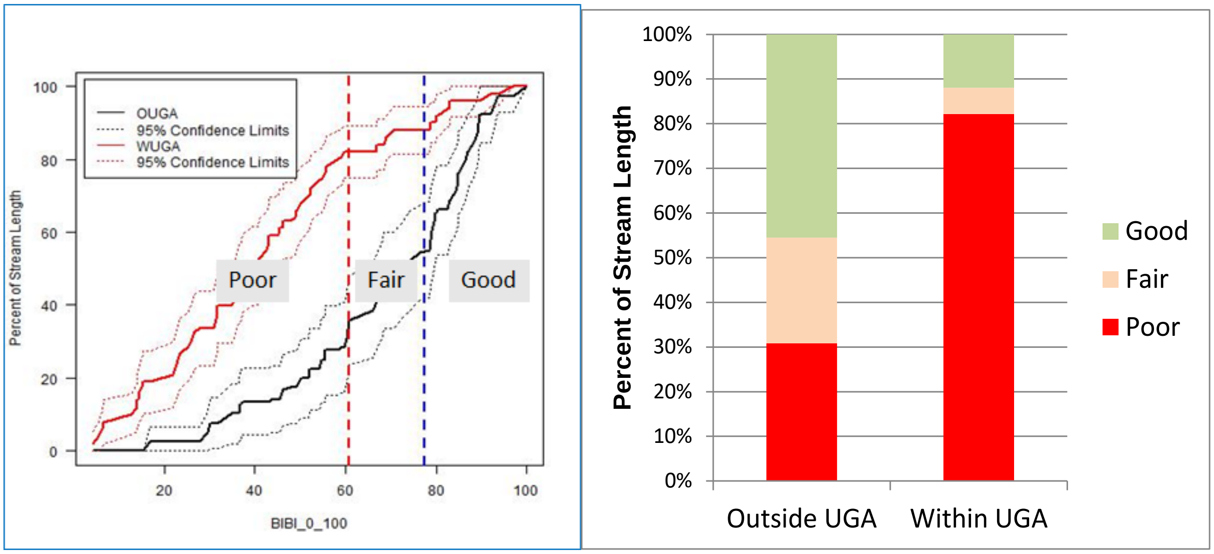


Figure 4. Cumulative distribution function (CDF) plot and categorical analysis bar chart for a an example metric, including 95% confidence limits and thresholds for categories of good, fair, poor on the CDF plot.

In Figure 4, the y-axis reports the percent of stream length of the represented streams in the strata within and outside UGAs and the x-axis is the range of B-IBI scores. The B-IBI value 60.8 is the lower threshold based on reference conditions. Therefore, the stream length below 60.8 is considered to be in poor condition. Based on this threshold 82 percent of stream length within UGAs and 30 percent of stream length outside of UGAs are in poor condition. Given two categorical strata in this study, outside and inside of UGAs, CDF plots provide the visual differences of status of stream health between them. Each CDF plot and stream length evaluation by thresholds was then converted to a stacked bar chart (chart on right in Figure 4), which describes the status of streams for each stratum. Note that both graphs display percent of stream length and total stream length differs within and outside UGAs (see Section 3.1 below).

Box plots are also used in this report to compare data collected from within and outside UGAs. Figure 5 provides an example of a box plot. The blue vertical line within each box represents the median value and the lower and upper ends of each box represent the 25th (Q1) and 75th (Q3) percentiles. The black diamond within each box is the mean value. The height of the box represents the interquartile range (IQR = Q3-Q1). The vertical lines on each box are known as whiskers. The upper whisker is defined by the minimum of the maximum value or the result of Q3 + 1.5 * IQR. The lower whisker is defined by the maximum of the minimum value or the result of Q1 – 1.5 * IQR. Solid blue points beyond the whiskers are “outliers” determined using the 1.5 rule – values less than Q1 – 1.5*IQR or greater than Q3 + 1.5*IQR.

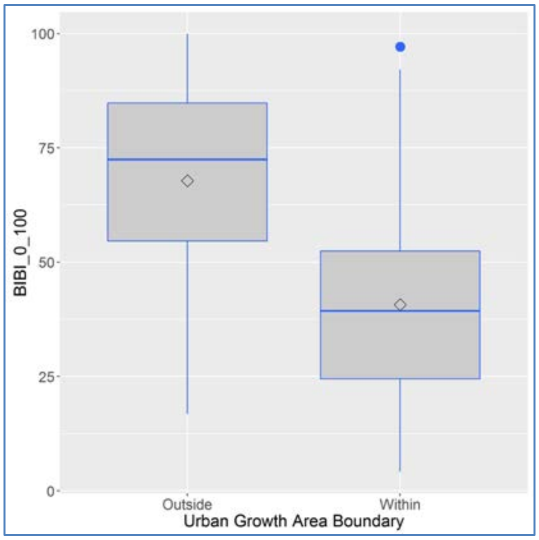


Figure 5. Box plot for an example metric for stream sites sampled outside and within Urban Growth Areas.

2.7.3 Correlation with Natural and Human Variables

Two distinct approaches, boosted regression trees and relative risk/attributable risk, were used to evaluate the study questions regarding what natural and human variables correlate with stream status within and outside UGAs. We note here that separate analyses were not conducted for sites within and outside UGAs for reasons provided below for each model. The approaches focused on identifying relationships between natural and human stressors and two key biological response measures, B-IBI and the TDI. Before describing the two approaches used, it may be helpful to define what is meant by natural and human variables.

To be considered a reliable biological response indicator, the indicator should respond to natural and human stressors, but be relatively insensitive to natural landscape variables that are largely beyond human control (Dorfmeier, 2014). Examples of such natural landscape variables include watershed area, average or seasonal precipitation variability, mean watershed elevation, surficial geology, and latitude/longitude. Biological metrics that are sensitive to these background variables might require some modification to minimize their influence. Biological response metrics that are not sensitive to natural background variables (or have been modified to remove their influence) can then be used to understand, if, and to what degree, natural and anthropogenic stressors influence these biological response metrics.

Generally, a stressor is a variable that as a result of human activity can exceed its range of normal variation with the potential to cause a negative biological response (Wagenhoff et al., 2011). Natural stressors, which are different than the natural landscape variables described above, can be naturally-occurring constituents like nutrients, metals, sediment, or streamflow which have a natural normal range in watersheds with little human disturbance (Munn et al, 2018). However, humans can alter the normal range of these stressors with the potential to cause a negative response in biological indicators. Other stressors are more clearly human, such as the occurrence of synthetic organic contaminants such as pesticides, PCBs, and PBDEs. Stressor-response relationships are not necessarily linear and can follow unimodal or threshold response shapes and can be complicated by interactions among multiple stressors (Wagenhoff et al., 2011). Regardless of the complexity, stormwater managers would potentially benefit from improved understanding of cause-effect relationships between stressors and biological responses.

Boosted regression trees were used to evaluate the potential influence of natural background variables as well as identify potentially important natural and human stressors. Relative risk/attributable risk analysis was used to evaluate the potential importance of natural and human stressors.

2.7.3.1 Boosted Regression Trees

Non-parametric boosted regression tree (BRT) models are generally well suited to problems where the number of predictor variables exceeds the number of samples, interactions exist among variables, non-linear relationships occur, data are missing, and variables do not satisfy the requirements of parametric statistical approaches (De'ath, 2007). BRT models do not "prove" or "disprove" cause-effect relationships. However, they do provide insights into the relative importance (i.e., correlation) of the predictor variables tested to explain variation in the target response variables; in this case B-IBI and Trophic Diatom Index scores. The BRT models were used to evaluate the potential influence of natural landscape variables and natural and human stressors on B-IBI and TDI scores.

BRT models were developed using the "dismo" (Hijmans et al., 2014) and "gbm" (Ridgeway, 2013) packages in R following guidance provided by Elith et al. (2008) and Elith and Leathwick (2014). Models were developed using a bag fraction of 0.75, a learning rate of 0.001 and a tree complexity of 5. A tree complexity of 5 allows the assessment of up to 5-way interactions among input variables. A bag fraction of 0.75 means that a random selection of 75 percent of the data is used each time a tree is developed. The learning rate sets the shrinkage applied to each tree in the BRT model and affects the total number of trees needed to fit the model. The learning rate of 0.001 resulted in a relatively "slow" learning rate and resulted in BRT models that were typically made of more than 1000 trees. As the goal was identifying natural and human stressor variables most strongly related to the biological response variables, the number of variables in the model were not reduced to find the most parsimonious set of predictor variables as one might do if prediction was the goal.

The output from the BRT models is somewhat different than that of parametric linear regression models. Model results reported include the cross-validation (CV) correlation, which gives an indication of the amount of variance in the response variable explained by the model. Cross-validation refers to the technique used to test the predictive capability of the model, which entails randomly partitioning the data into training (model fitting) and validation (testing) data sets. Variable relative importance (VRI) values can also be reported. Calculations of VRI are based on the number of times a variable is selected for splitting, weighted by the squared improvement to the models as a result of each split, and averaged over all trees. The relative importance of each variable is scaled so that together they add to 100 percent, with higher numbers indicating stronger influence on the modeled response.

BRT models were developed for separate groups of monitoring variables (landscape, habitat, water quality, sediment quality and subgroups of landscape and habitat variables), and the most important variables were tested in an overall model to identify the most important variables across all categories that explained the variance in the biological response variable. Initially, models were tested using within and outside UGAs as a categorical variable, but this classification was not a relatively important variable in any model. Therefore, the data for within and outside UGAs was combined for the development of the BRT models.

2.7.3.2 Relative Risk/Attributable Risk Analysis

Another approach to identifying the relative importance of stressor variables relative to a specific response variable is derived from the epidemiological literature (Van Sickle, 2013). The method is generally known as a relative risk/attributable risk (RR/AR) analysis and is an extension of the status assessment presented above. RR/AR analysis provides quantifiable associations between stressors of concern and biological response, making this a useful tool to identify potential causes of poor biological condition and providing information to inform management decisions. This approach did not include an evaluation of the potential influence of natural landscape variables.

Relative risk is the conditional probability of finding poor biological response when a specific stressor is also poor relative to the conditional probability of poor response when the stressor is not poor. A relative risk of 1 or less indicates no association between the response and stressor variables. Attributable risk is the relative improvement (going from poor to not-poor) in a response variable if all sites in poor condition for a stressor are converted to not-poor condition. Attributable risk ranges from 0 to 100 percent with 0 indicating no association between stressor and response. Although it is recognized that relative and attributable risk estimates can be confounded by multiple comparisons, methods to control these effects explored by Van Sickle (2013) were beyond the scope of this analysis.

The methods developed to conduct RR/AR analyses of ecological survey data are described by Van Sickle (2013) and the approach used in our study is similar to the approach used by King County (2014b) to evaluate the RR/AR for B-IBI stressors across western Washington. It was not possible to conduct separate analyses for sites within and outside UGAs as more than 50 sites is considered necessary to meet the statistical requirements of this analysis. A previous study combined data from the Puget Sound basin with data from the lower Columbia River and Washington coastal basins to create a data set of 146 sites (King County, 2014b).

Because of the large number of potential stressors that could be evaluated, landscape, water, sediment, and habitat metric groups were first screened by calculating Spearman rank correlations with the biological response variable. Metrics within each group with the highest rank correlations were then selected for inclusion in the RR/AR analysis. The RR/AR for the TDI focused on landscape and nutrient stressors since TDI is a nutrient enrichment (i.e., eutrophication) indicator.[10] The R package "spsurvey" (Kincaid and Olsen, 2015) was used to perform the RR/AR analysis.

The three steps of the RR/AR analysis were:

1. Establish the relative extent of poor condition across the study area for the selected stressor and response variables. For example, the extent of streams in poor condition as defined by B-IBI scores and the extent of environmental stressors like riparian canopy cover classified in poor condition.
2. Identify the relative risk (ratio) of poor biological condition when poor stressor condition is observed. When the relative risk of poor B-IBI associated with a particular stressor is greater than 1 then an elevated risk of poor B-IBI condition is associated with the stressor. For example, if the relative risk of poor B-IBI condition associated with canopy cover was found to be 3, then the risk of poor B-BIBI scores is 3 times greater in streams with poor canopy condition.
3. Lastly, attributable risk combines the two prior steps into one number that can be used to rank environmental stressors across the whole study area. Because attributable risk is reflective of both extent and risk to biota, attributable risk can estimate the regional-level impact of each variable. If a particular stressor condition were improved, the potential percent improvement in B-IBI scores (i.e., shift from poor to not poor condition) can be estimated. For example, an RR/AR analysis might suggest that improving stream canopy cover has the potential to improve B-IBI condition across 40 percent of the Puget Lowland stream length.[11]

The final set of stressors that were evaluated with respect to the RR/AR of B-IBI scores, and the thresholds used in the analysis are identified in Table 16. Note that two watershed landscape metrics (Watershed %Urban Development and Watershed Canopy Cover) were included in the RR/AR analysis. Both of these metrics were highly correlated with B-IBI scores and similar metrics have been used in previous B-IBI studies. Those studies illustrated the close relationship of development and forest cover loss on B-IBI scores (Alberti et al., 2007; DeGasperi et al., 2009) so both were included in the analysis.

[10] Also note, that the sediment thresholds identified were for effects on benthic invertebrates so they would not be appropriate for a RR/AR analysis of TDI,

[11] We recognize that relative and attributable risk estimates can be confounded by multiple comparisons, methods to control these effects explored by Van Sickle (2013) were beyond the scope of this analysis.

Table 16. Thresholds used in the B-IBI 0-100 scale Relative Risk/Attributable Risk analysis.

Parameter (units)	Poor	Not-poor
B-IBI 0-100 scale	<60.8	>77.4
Watershed %Urban Development	>6.76	<3.76
Watershed Canopy Cover (%)	<62.8	>66.5
Riparian Canopy Cover (%)	<55.3	>67.2
Median Substrate Diameter (mm)	<0.653	>10.7
Stream Embeddedness	>73.9	<51.4
Total Phosphorus (mg/L)	>0.094	<0.041
Total Nitrogen (mg/L)	>0.862	<0.459
Chloride (mg/L)	>12.6	<9.9
Sediment Zinc (mg/Kg)	>459	<121

The final set of stressors that were evaluated with respect to the RR/AR of TDI scores, and the thresholds used in the analysis are identified in Table 17.

Table 17. Thresholds used in the Trophic Diatom Index Relative Risk/Attributable Risk analysis.

Parameter (units)	Poor	Not-poor
Trophic Diatom Index (TDI)	>61.1	<58.3
Watershed %Urban Development	>6.76	<3.76
Total Phosphorus (mg/L)	>0.050	<0.041
Total Nitrogen (mg/L)	>0.862	<0.459

3.0 RESULTS

In this section we present the results for the 2015 SAM PLES survey design implementation (Section 3.1); an overview of the landscape GIS analysis results (Section 3.2); the biological, water and sediment quality, and habitat status assessment (Section 3.3); and identify the natural and human variables that best correlate with the observed status of stream conditions (Section 3.4).

3.1 Survey Design Implementation

There are two key advantages to probabilistic sampling design frameworks. First, sampling effort allows the extrapolation of any measured indicator (biological, chemical, and physical) from the sites sampled to estimates of the status of the extent of the represented region, in this case, small Puget Lowland ecoregion streams within and outside urban growth area boundaries. Second, datasets can be combined or compared as long as programs have a common set of indicators and protocols and overlapping geographic regions (Larsen et al., 2008).

One of the first steps of implementing a probabilistic sampling design is to calculate the adjusted spatial weight of each site within the sampling frame for the study just completed. The resultant spatial weights for SAM PLES regional (Option 1) sites and Option 2 sites are shown in Table 18. The total length of streams represented by the SAM PLES probabilistic framework is 2,685 km (1,668 mi). As SAM PLES Option 1 and Option 2 sampling sites were selected using the same probabilistic framework and sampling efforts measured the same parameters following the same protocols, the results and spatial weights can be combined for status assessment.

For water quality parameters, adjusted spatial weights were calculated only for the Option 1 sites (24 outside UGAs and 28 within UGAs) because of the difference in timing of water quality sampling between Option 1 and Option 2 (calendar year 2015 for Option 1 and water year 2015 for Option 2). Given the relatively small contribution of Option 2 sites to the regional scale analysis, this was considered to be a reasonable compromise.[12]

[12] The SAM PLES regional sites (Option 1) represented 2,527.3 km of streams within the Puget Lowland. In comparison, Option 2 sites (City of Redmond and unincorporated Pierce County) represented 157.2 km of stream length. Therefore, 94 percent of Puget Sound lowland regions stream length is represented by the SAM PLES regional sampling effort whereas Option 2 sites represent 6 percent of the target streams.

Table 18. Summary of adjusted spatial weights used in the status assessment.

		WH			WQ		
Study frame	**Strata**	**km (# of sites)**	**Total km**	**%**	**km (# of sites)**	**Total km**	**%**
SAM PLES	WUGA	11.44 (48)	549.24	94.1	21.81 (28)	610.68	26.2
	OUGA	53.46 (37)	1,978.10		71.75 (24)	1,722.0	73.8
Redmond	WUGA	1.46 (7)	10.25	0.4			
	OUGA	-					
Pierce County	WUGA	4.44 (4)	17.77	5.5			
	OUGA	14.35 (9)	129.20				

WH = Watershed Health Monitoring; WQ = Monthly Water Quality Monitoring

Study ID in Ecology EIM database: SAM_PLES for SAM, RSMP_PC_PLES2015 for Pierce County, and RSMP_RD_PLES2015 for City of Redmond

3.2 Physical Landscape and Land Cover Data

The GIS analysis confirmed the expectation that sites within UGAs would be more developed than sites outside UGAs and that reference sites would have the least amount of development. The average upstream watershed distribution of four major land use categories (percent urban development, percent agriculture, percent forest cover, and percent wetland cover) is shown in Figure 6. Average percent urban land cover of sites within UGAs was 45 percent compared to 6 percent for sites outside UGAs and less than 3 percent for reference sites. The pattern was reversed for percent forest cover. Average forest cover was greatest for reference sites (74 percent) with less average watershed forest cover for sites outside UGAs (60 percent) and sites within UGAs (27 percent). The average amount of watershed agriculture and wetland cover was generally low; less than 5 percent for reference, outside UGA, and inside UGA sites.

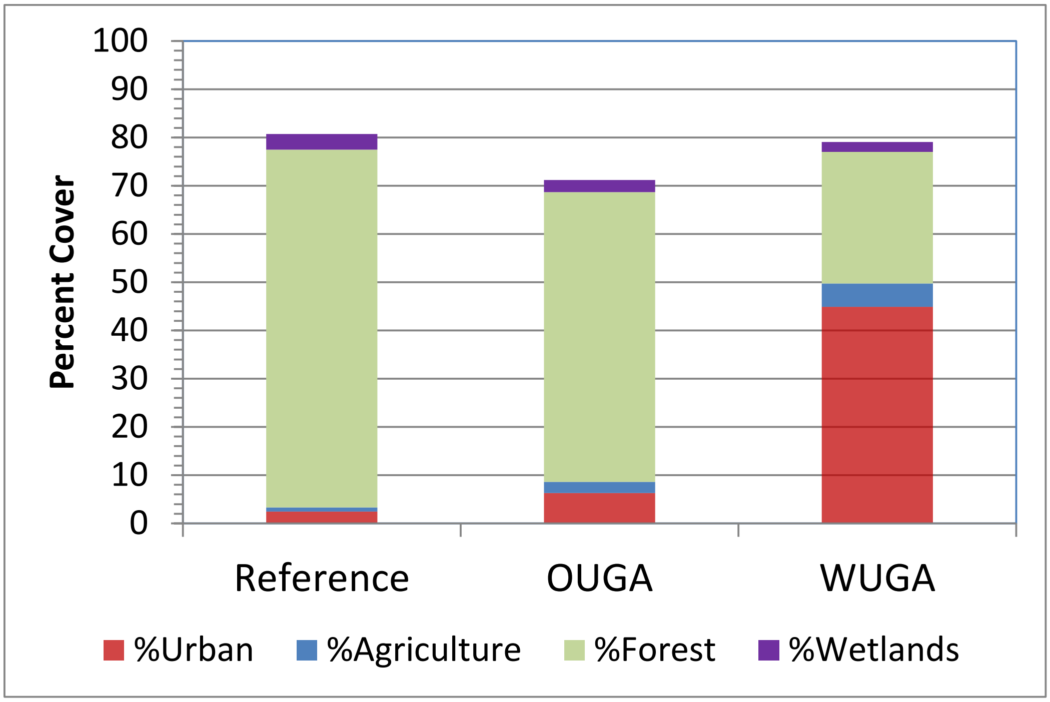


Figure 6. Bar chart illustrating the average distribution of four watershed land cover categories (%Urban, %Agriculture, % Forest, %Wetlands) for sites within and outside UGAs and in 16 reference watersheds.

Note: The four cover categories do not sum to 100 percent because there are several additional categories that account for the remainder of the watershed land cover. The four categories shown are considered the most relevant for comparison.

The average upstream riparian distribution of the same four major land use categories is shown in Figure 7. Average percent urban land cover of sites within UGAs was 25 percent compared to 3 percent for sites outside UGAs and less than 1 percent for reference sites. The pattern was reversed for percent forest cover. Average forest cover was greatest for reference sites (74 percent) with less average watershed forest cover for sites outside UGAs (56 percent) and even less for sites within UGAs (35 percent). The average amount of watershed agriculture use was generally low; 5 percent or less for reference, outside UGA, and inside UGA sites. However, riparian percent wetland cover was higher, with similar average cover in reference and outside UGA basins (16 and 18 percent, respectively) and somewhat less in within UGA basins (9 percent).

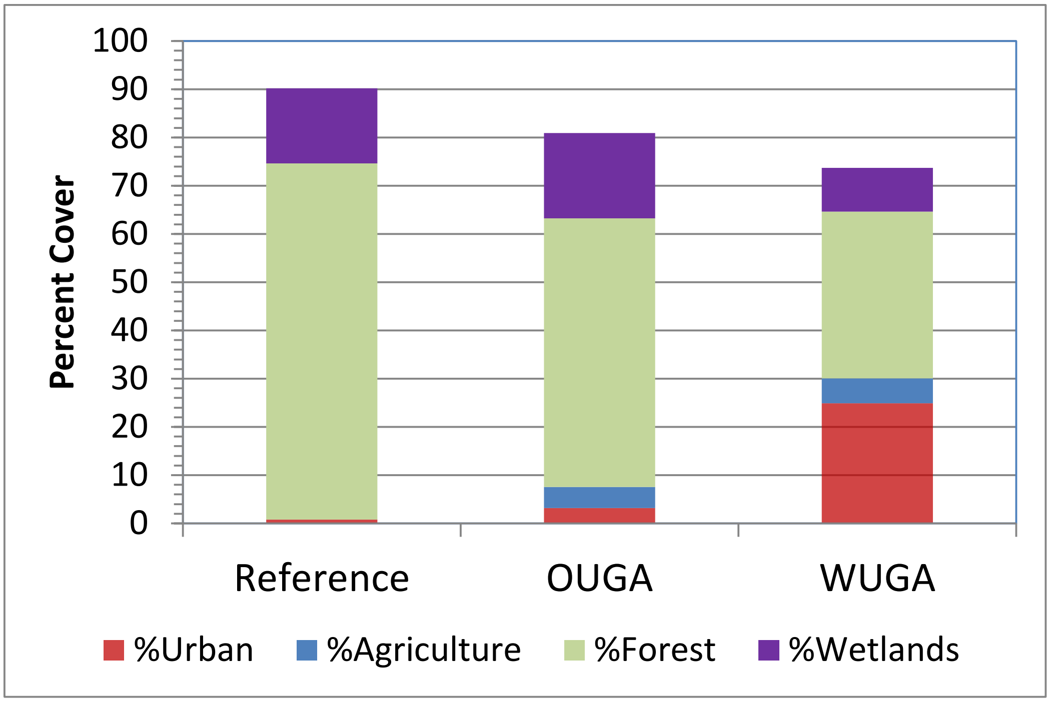


Figure 7. Bar chart illustrating the average distribution of four riparian land cover categories (%Urban, %Agriculture, % Forest, %Wetlands) for sites within and outside UGAs and in 16 reference watersheds.

Note: The four cover categories do not sum to 100 percent because there are several additional categories that account for the remainder of the watershed land cover. The four categories shown are considered the most relevant for comparison.

Although the average amount of urban and agriculture land cover was consistent with expectations, there were high levels of urban and agriculture land cover that were not expected within and outside UGAs. For example, there were outside UGA sites with levels of watershed percent urban development of as much as 44 percent (Figure 8). The watershed percent agriculture upstream of each site was also quite variable. The highest levels of agricultural land cover were found at within UGA sites (two sites had percent agriculture cover greater than 80 percent). The highest watershed agriculture cover for outside UGA sites was over 40 percent (Figure 7).

The target streams were intended to be small – 1st to 3rd Strahler order.[13] Strahler stream order is a method for estimating stream size and it is based on the number and hierarchical relationship of mapped tributaries. However, the determination of stream order depends on the quality and scale (resolution) of the map used. The drainage area above the sampling point is another indicator of stream size (and the area of upstream influence).

[13] Strahler stream order is determined starting from the uppermost perennial or intermittent headwater stream (a stream with no tributaries). First order streams are the outermost tributaries. If two streams of the same order merge, the resulting stream is given a number that is one higher. If two streams with different stream orders merge, the resulting stream is given the higher of the two numbers.

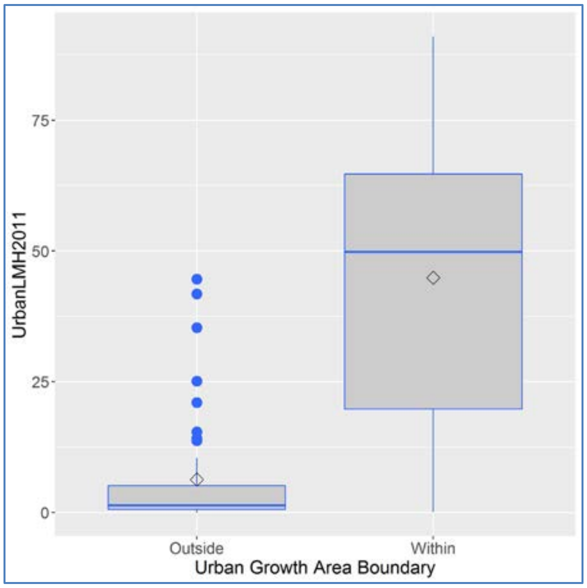


Figure 8. Box plot of watershed percent urban development for sites sampled outside and within Urban Growth Areas.

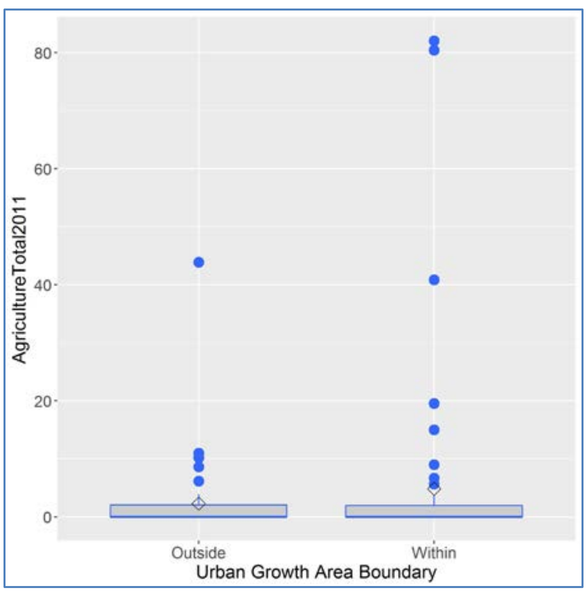


Figure 9. Box plot of watershed percent agriculture for sites sampled outside and within Urban Growth Areas.

The upstream watershed drainage areas were typically less than 50 km^2 (12,355 acres), but 18 of the sites had drainage areas greater than 50 km^2 and one site had a drainage area of almost 400 km^2 (98,842 acres) (Figure 10).

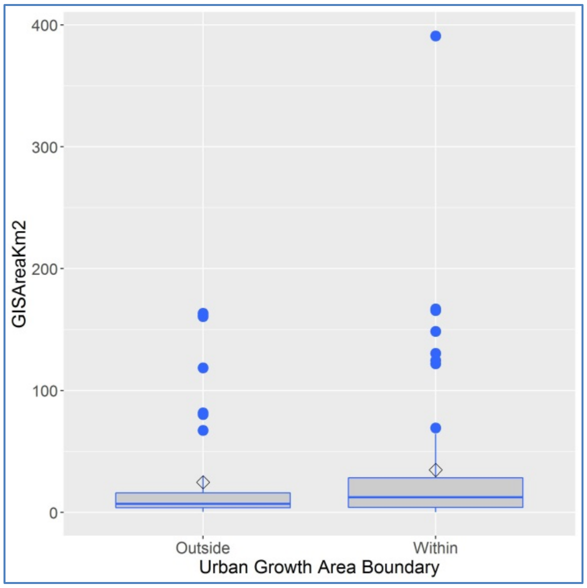


Figure 10. Box plot of watershed drainage area for sites sampled outside and within Urban Growth Areas.

3.3 Status Assessment

In this section we will answer Q1, Q2 and Q3 based on the one year (2015) of data from SAM Streams. The 2015 SAM PLES study generated data for hundreds of parameters and indicator metrics. Because of the large number of parameters measured, we don't present the results for all parameters. Only status assessments for key parameters are presented here. For readability, we present and discuss representative metrics based on their familiarity, availability of state standards, judgment regarding their potential utility (e.g., importance in BRT and/or RR/AR analysis), or to illustrate important points. For each parameter, we also indicate whether or not there was statistical evidence (based on the Wald F test) for differences within and outside UGAs. This is meant to provide an indication that the difference in the particular parameter was due to the rural-urban gradient, which the outside and within UGA survey strata generally captures.

The status results for the selected parameters are summarized in Table 19. The detailed biological, water and sediment quality, and habitat status assessments are presented in topic sections below. A summary of laboratory detection frequency is provided in Appendix A of this report. Statistical summaries of the 2015 study data within and outside UGAs are provided as separate tables in Appendix B for each data type as indicated below:

- Appendix B Table B1: Benthic invertebrate metrics
- Appendix B Table B2: Periphyton metrics
- Appendix B Table B3: Water Quality Index (annual and monthly)

Table 19. Summary of the status assessment results presented in this report.

Numbers of sites assessed within and outside Urban Growth Areas (UGAs) for particular parameters; total percentages of Puget Lowland Ecosystem Stream length found to be in poor, fair, and good condition within and outside UGAs; and whether there was a statistically significant difference between conditions within and outside UGAs Percentages in poor, fair, good do not always add up to 100% due to rounding.

Parameter	Number of sites with data assessed		Percent of stream length in "poor" condition		Percent of stream length in "fair" condition		Percent of stream length in "good" condition		Difference between OUGA and WUGA?
	OUGA	WUGA	OUGA	WUGA	OUGA	WUGA	OUGA	WUGA	
Biological:									
B-IBI	45	59	31	82	24	6	46	12	Yes
HBTI	45	59	33	63	11	5	56	33	Yes
FSSI	45	59	62	90	15	8	23	2	Yes
MTI	45	59	3	4	3	19	94	77	Yes
TDI	45	59	29	66	0	8	71	26	Yes
Water quality:									
WQI	24	28	0	0	33	57	67	43	Yes
Fecal bacteria	24	28	0	32	-	-	100	68	Yes
Minimum DO	24	28	63	64	-	-	38	36	No
Minimum pH	24	28	29	11	-	-	71	89	Yes
Maximum pH	24	28	13	7	-	-	88	93	No
Max.Temperature	24	28	54	54			46	46	No
Total phosphorus (Aug-Oct mean)	24	28	8	46	12	18	80	36	Yes
Total nitrogen (Aug-Oct mean)	24	28	12	43	20	18	68	39	Yes
Sediment quality:									
Arsenic	46	59	1	10	28	39	72	51	Yes
Cadmium	46	59	3	0	1	8	97	92	Yes
Chromium	46	59	3	2	50	55	48	43	No
Copper	46	59	3	6	63	54	35	40	No
Lead	46	59	3	4	6	25	92	70	Yes
Zinc	46	59	0	2	15	59	85	39	Yes
Total PAHs	46	59	0	2	0	9	100	89	NA
Total PCBs	46	59	0	0	0	5	100	95	Yes
Total PBDEs	46	59	0	0	0	7	100	93	Yes
Dichlobenil	46	59	NA	NA	NA	NA	NA	NA	No

Parameter	Number of sites with data assessed		Percent of stream length in "poor" condition		Percent of stream length in "fair" condition		Percent of stream length in "good" condition		Difference between OUGA and WUGA?
	OUGA	WUGA	OUGA	WUGA	OUGA	WUGA	OUGA	WUGA	
Habitat:									
Canopy closure	46	59	20	20	18	6	61	74	No
Wood volume	46	59	44	61	17	9	39	31	No
Residual pool area	46	59	16	29	40	18	44	53	No
Stream Substrate	46	59	20	25	36	62	45	13	Yes
Bed Stability	46	59	25	36	12	38	64	26	Yes
Landscape:									
Watershed urban development	46	59	17	86	12	4	72	10	Yes
Watershed canopy cover	46	59	41	94	13	2	46	4	Yes
Riparian canopy cover	46	59	29	56	14	24	57	21	Yes
Areal nitrogen loading	46	59	16	76	27	12	56	12	Yes

***OUGA** is outside Urban Growth Areas; **WUGA** is within Urban Growth Areas; **B-IBI** is benthic index of biotic integrity; **HBTI** is Hilsenhoff Biotic Tolerance Index; **FSSI** is Fine Sediment Sensitivity Index; **MTI** is Metals Tolerance Index; **TDI** is trophic diatom index; **WQI** is Water Quality Index; **PAHs** are polycyclic aromatic hydrocarbons; **PCBs** are polychlorinated biphenyls; **PBDEs** are polybrominated diphenyl ethers. **NA** is Not Assessed due to limited frequency of detection outside UGAs (Total PAH) or due to lack of screening level (dichlobenil).*

- Appendix B Table B4: In situ water quality data
- Appendix B Table B5: Laboratory water quality data
- Appendix B Table B6: Laboratory sediment quality data
- Appendix B Table B7: Stream habitat metrics
- Appendix B Table B8: Landscape GIS data

The results of all cumulative distribution frequency analyses, including the results of statistical comparisons of streams within versus outside UGAs (i.e., Wald F test) are provided in Appendix C for the following data types:

- Benthic invertebrate metrics
- Appendix C1: Benthic macroinvertebrate metrics
- Appendix C2: Periphyton metrics
- Appendix C3: Water Quality Index (annual and monthly)
- Appendix C4: In situ water quality data
- Appendix C5: Laboratory water quality data
- Appendix C6: Laboratory sediment quality data
- Appendix C7: Stream habitat metrics
- Appendix C8: Landscape GIS data

For each data type in Appendix C there are three tables (using Appendix C1 as an example):

- Appendix C1 Table C1-1: Summary statistics from the CDF analyses produced by spsurvey (within and outside UGAs), including the 5th through the 95th percentile (including the median), and the mean, variance, and standard deviation for each measured parameter or metric.
- Appendix C1 Table C1-2: The tabular data for each CDF, including the estimated proportion of data below each value and lower and upper 95 percent confidence limits.
- Appendix C1 Table C1-3: The results of the Wald tests for differences in the CDFs for data within and outside UGAs. For the monthly water quality and in situ data, these tests are based on the median of the values reported for each of the 52 sites for which adjusted spatial weights were calculated.

3.3.1 Biological Indicators: comparison with reference conditions

The biological data (benthic macroinvertebrates and periphyton) were used to calculate metrics that are commonly used in western Washington to understand stream biological health and potential stressors. Here we present five metrics with a relatively strong differentiation observed between the two strata; within and outside UGAs. The biological indicators included:

- Benthic Index of Biotic Integrity (B-IBI; 0-100 scale) – Decreases with impairment
- Hilsenhoff Biotic Tolerance Index (HBTI – Increases with inputs of organic pollution
- Fine Sediment Sensitivity Index (FSSI) – Decreases with increasing fine sediment
- Metals Tolerance Index (MTI) – Increases with metal concentrations
- Trophic Diatom Index (TDI) – Increases with inputs of nutrients

Benthic invertebrate diversity is commonly used to assess wadeable stream health. The benthic invertebrate taxa data are used to calculate the B-IBI index or score, where a higher number indicates a better health score for Puget Lowland streams. Benthic invertebrate taxa counts were also used to calculate three other metrics that help identify stressors or causes of poor benthic invertebrate condition. The stressors evaluated by these metrics, respectively, include the input of excessive decomposable organic matter, fine sediment, and metal pollution. The Trophic Diatom Index is a periphyton metric which is a sensitive indicator of stream nutrient enrichment. There are no established Washington state biological standards, so no evaluation of the biological data with respect to state standards is presented.

3.3.1.1 B-IBI

B-IBI scores were calculated within and outside UGAs using the 0-100 scale; higher B-IBI scores indicate better biological condition. B-IBI scores were typically higher outside of UGAs (Figure 11 and Figure 12). The differences between the CDFs within and outside UGAs were statistically significant (Table 19 and the left panel in Figure 12). By comparing to least-disturbed reference conditions (thresholds shown in left panel of Figure 12), 82 percent of the stream length within UGAs was in poor condition whereas 30 percent of the stream length outside UGAs was in poor condition (Figure 12). A greater proportion of stream length outside UGAs was in good condition (45 percent) relative to streams within UGAs (12 percent). Low B-IBI scores in streams are indicative of a lack of sensitive or long-lived benthic taxa, a high prevalence of stress tolerant taxa, and a lack of diversity in taxa.

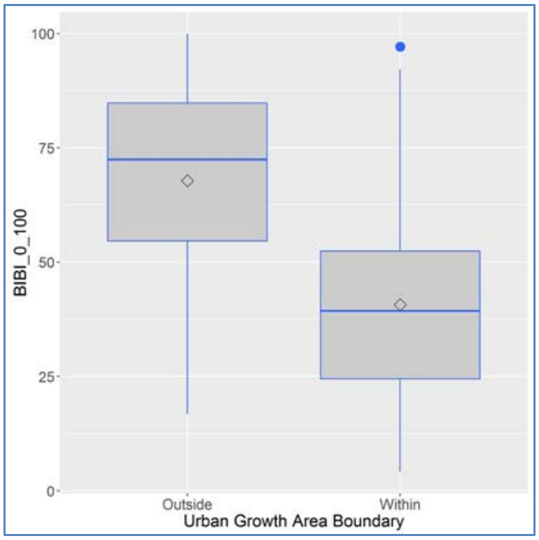
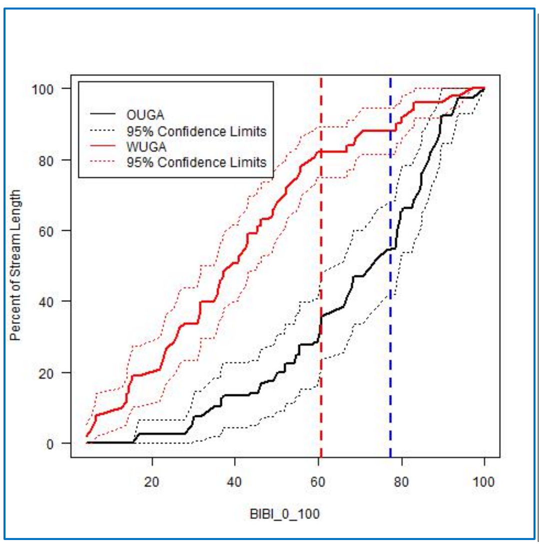


Figure 11. B-IBI (0-100 scale) box plot for stream sites sampled outside and within Urban Growth Areas.



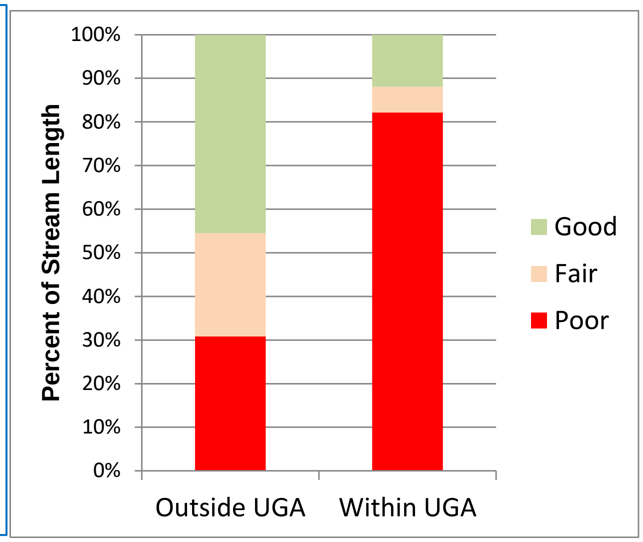


Figure 12. B-IBI (0-100 scale) cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.1.2 Hilsenhoff Biotic Tolerance Index

Higher HBTI scores indicate poorer biological condition due to easily decomposable organic matter pollution (Hilsenhoff, 1988). HBTI scores were higher within UGAs (Figure 13 and Figure 14). The differences between the CDFs within and outside UGAs were statistically significant (Table 19 and left panel of Figure 14Figure 13). Based on least-disturbed reference condition, 63 percent of the stream length within UGAs was in poor condition with respect to HBTI scores, while 33 percent of the stream length outside UGAs was in poor condition (Figure 14). A greater proportion of stream length outside UGAs was in good condition (56 percent) relative to streams within UGAs (32 percent).

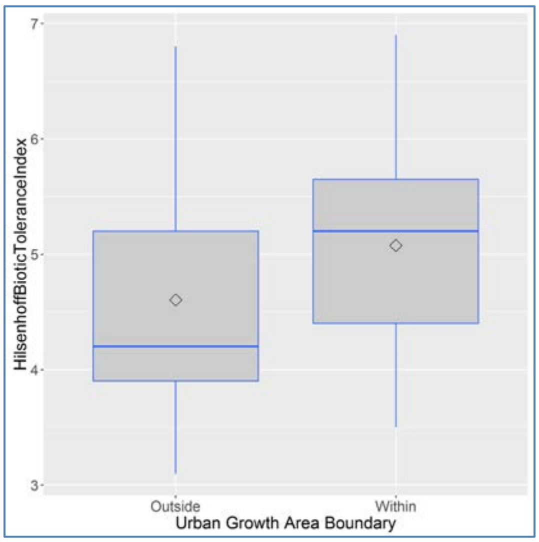


Figure 13. Hilsenhoff Biotic Tolerance Index box plot for stream sites sampled outside and within Urban Growth Areas.

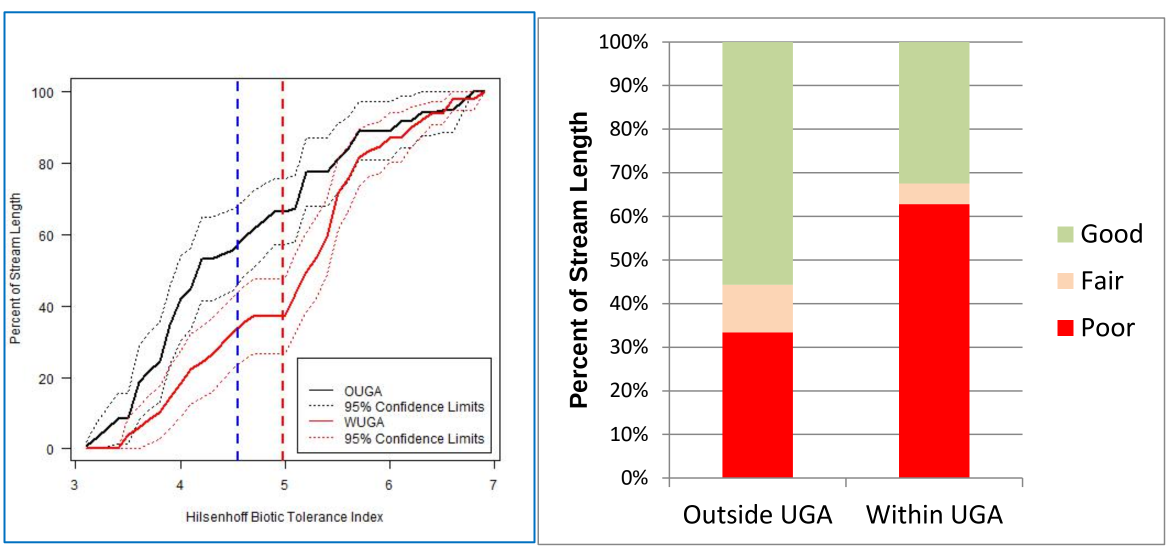


Figure 14. Hilsenhoff Biotic Tolerance Index cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.1.3 Fine Sediment Sensitivity Index

Lower FSSI scores indicate a lack of invertebrate taxa sensitive to fine sediment (Reylea et al., 2012). In PLES, FSSI scores were typically lower within of UGAs (Figure 15 and Figure 16). The differences between the CDFs within and outside UGAs were statistically significant (Table 19 and the left panel in Figure 16). 90 percent of the stream length within UGAs was in poor condition with respect to FSSI scores, while 62 percent of the stream length outside UGAs was in poor condition (Figure 16). A greater proportion of stream length outside UGAs was in good condition (23 percent) relative to streams within UGAs (2 percent).

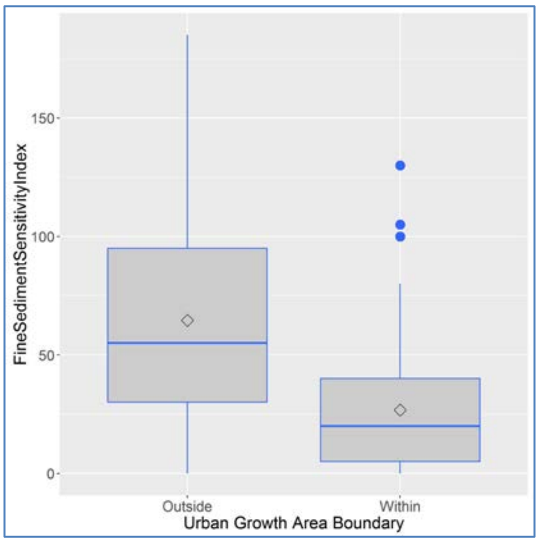


Figure 15. Fine Sediment Sensitivity Index box plot for stream sites sampled outside and within Urban Growth Areas.

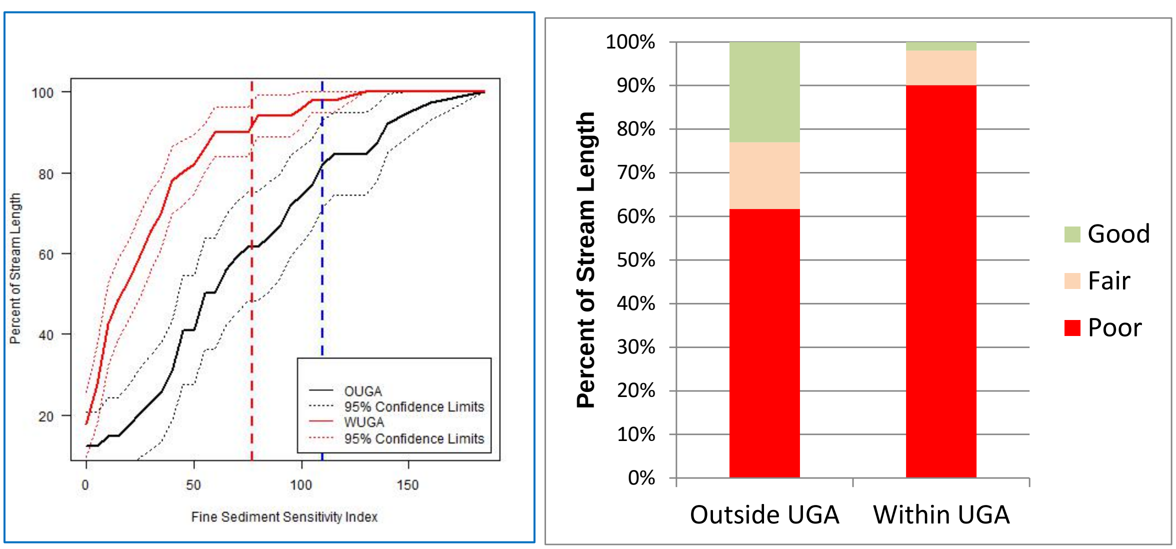


Figure 16. Fine Sediment Sensitivity Index cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.1.4 Metals Tolerance Index

Higher MTI scores (range 0-10) indicate poorer biological condition resulting from metal pollution (McGuire, 2009). MTI values less than 4 are typical of benthic macroinvertebrate communities dominated by species intolerant of metals (McGuire, 2009). Only one site (within a UGA) had an MTI score greater than 4 (Figure 15 and Figure 18). We found a statistically significant difference between CDFs of MTI scores within and outside UGAs (Table 19 and the left panel in Figure 18). Based on reference condition, 4 percent of the stream length within UGAs was in poor condition with respect to MTI scores, while 3 percent of the stream length outside UGAs was in poor condition (Figure 18). 94 percent of stream length outside UGAs was in good condition whereas it was 77 percent for within UGAs.

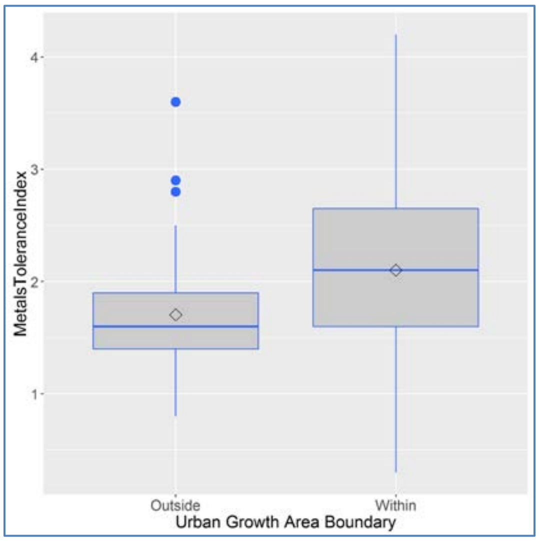


Figure 17. Metals Tolerance Index box plot for stream sites sampled outside and within Urban Growth Areas.

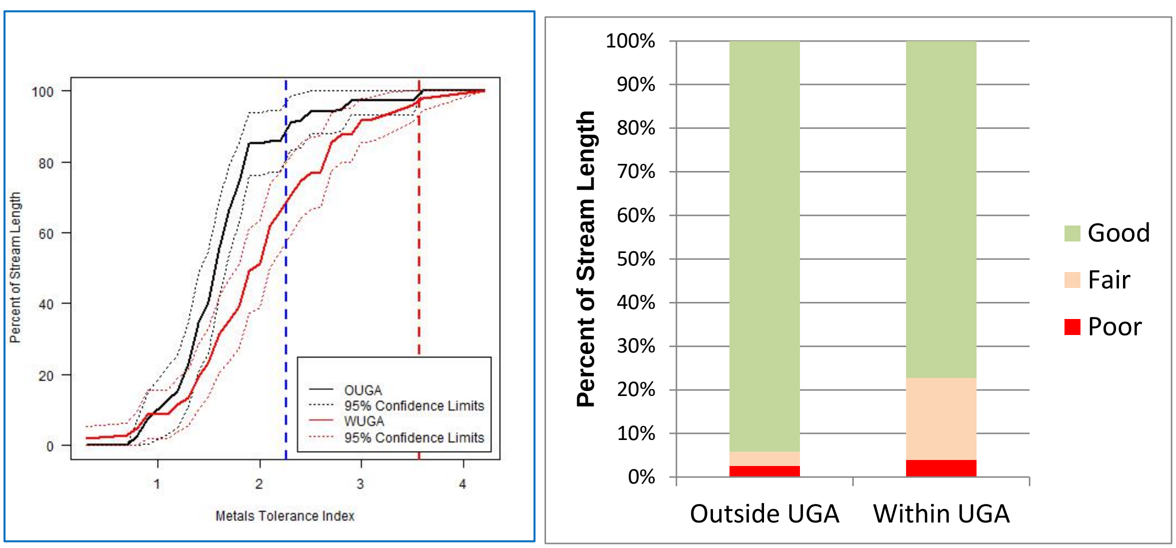


Figure 18. Metals Tolerance Index cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.1.5 Trophic Diatom Index

Lower TDI values indicate generally less eutrophic (i.e., lower nutrient concentrations) stream conditions (Kelly and Whitton, 1995; Kelly, 1998). We found that TDI scores were typically lower (better condition) outside of UGAs (Figure 19 and Figure 20). The differences between the CDFs within and outside UGAs were statistically significant (Table 19 and the left panel in Figure 20). Based on least-disturbed reference condition, 66 percent of the stream length within UGAs was in poor condition, while 30 percent of the stream length outside UGAs was in poor condition (Figure 20). A greater proportion of stream length outside UGAs was in good condition (71 percent) relative to streams within UGAs (26 percent).

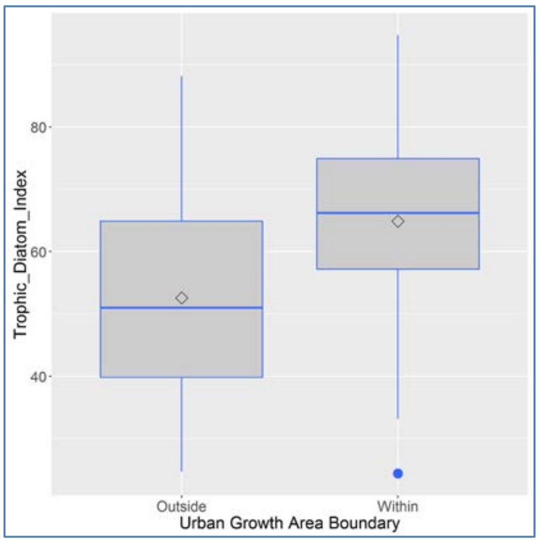


Figure 19. Trophic Diatom Index box plot for stream sites sampled outside and within Urban Growth Areas.

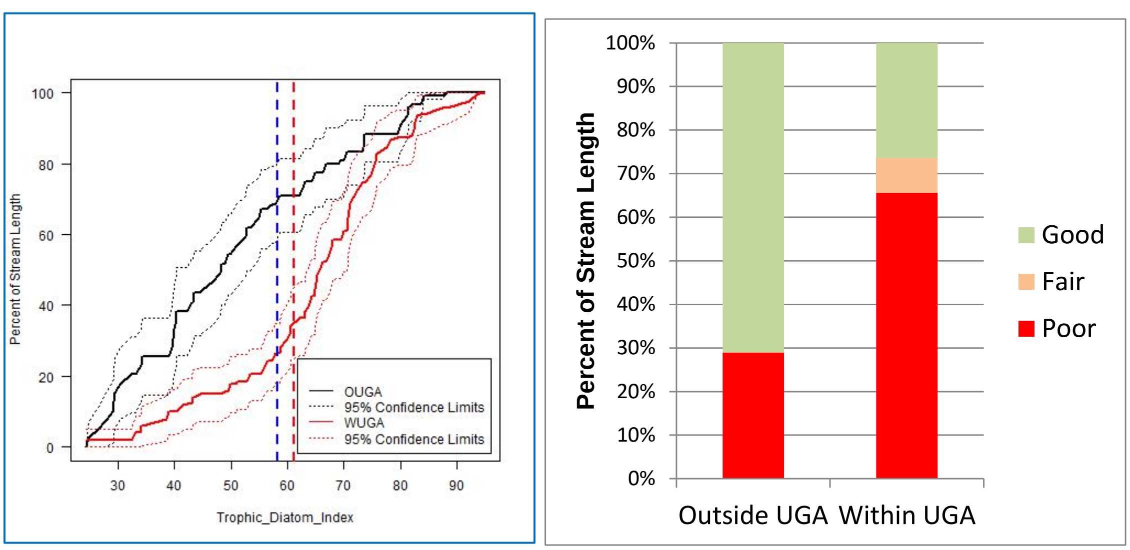


Figure 20. Trophic Diatom Index cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.2 All Other Biological Metrics

Twenty other measures of the biological health of stream invertebrates were calculated for this study (including the component B-IBI metrics). Eighteen of these metrics indicated better biological conditions outside of UGAs (Appendix C1 Table C1). There were 44 other measures of the biological health of the periphyton community, 29 of this indicated better biological condition outside of UGAs (Appendix C2 Table C2). Areal periphyton biomass (mg Chlorophyll a per m2) was also measured as part of this study. Areal periphyton biomass was significantly higher outside vs. inside UGAs (Appendix C2 Table C2).

3.3.3 Water Quality Index: Spatially Adjusted Results

The WQI was a specific focus for the 2015 SAM PLES sampling. There are no regulatory standards for the WQI scores. We use the Ecology condition categories for comparison.

The WQI can range from 0-100, where higher values indicate better water quality. Scores above 80 meet expectations for Good water quality and below 40 is indicative of a stream of high concern. We found that the annual WQI scores were slightly lower within UGAs (Figure 21 and Figure 22) and the differences between the CDFs within and outside UGAs was statistically significant (Table 19 and left panel in Figure 22). Based on Ecology's established WQI condition categories, none of the stream length within or outside UGAs was determined to be in poor condition (Figure 22). A greater proportion of stream length outside UGAs was in good condition (67 percent) relative to streams within UGAs (43 percent).

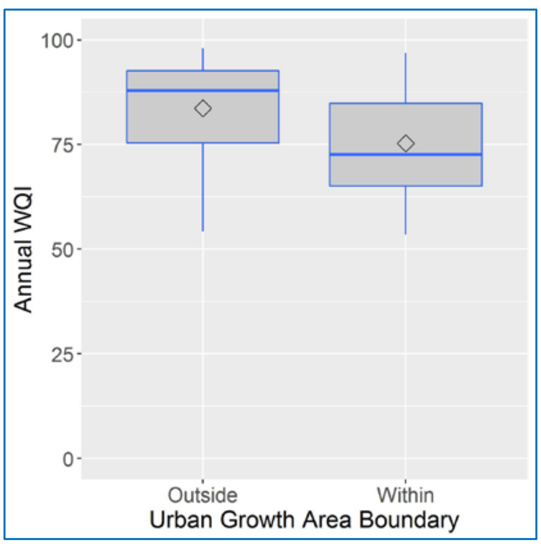


Figure 21. Water Quality Index box plot for stream sites sampled outside and within Urban Growth Areas.

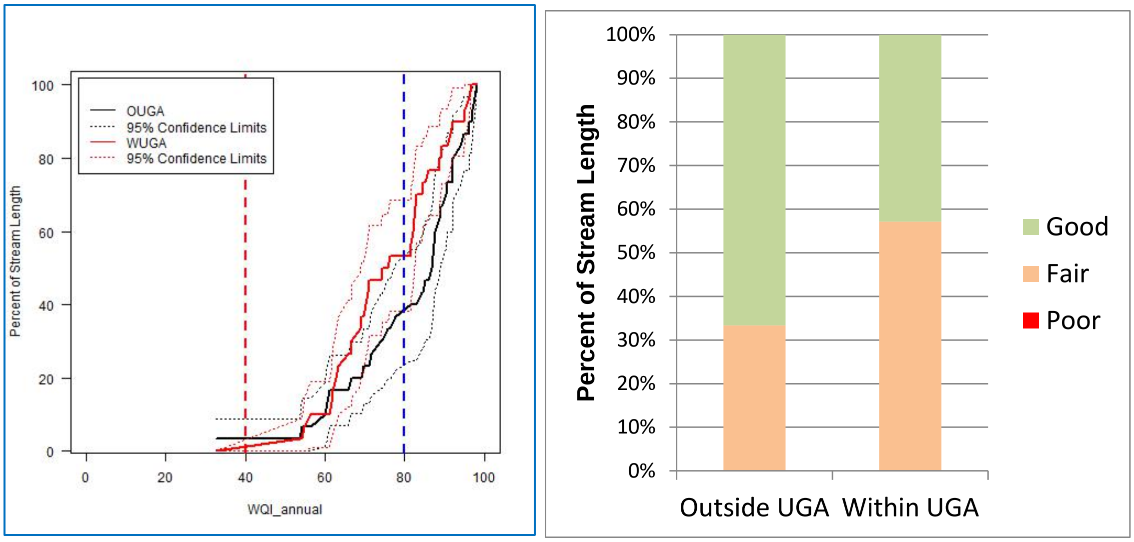


Figure 22. Water Quality Index cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

In addition to the annual WQI, WQI values for each month and WQI values for each parameter in each month were calculated (summary statistics provided in Appendix B Table B3). Statistically significant differences in the monthly WQI values within and outside UGAs were found in 7 of 12 months (Table 20). The months in which statistically significant differences were found were March, April, May, June, August, September, and October (Appendix C3 Table C3-3). The lowest monthly WQI values generally occurred in August, September, and October (Figure 23). The monthly parameter-specific WQI values that were most often statistically different within and outside UGAs were WQIs for fecal coliform (8 of 12 months), total phosphorus (10 of 12 months), and total nitrogen (9 of 12 months) (Table 20). The differences found occurred in several months, but all of the monthly fecal coliform, total phosphorus, and total nitrogen WQIs were statistically different in August, September, and October (see Appendix C3 Table C3-3). The other parameter monthly WQIs (dissolved oxygen, pH, temperature, total suspended solids, and turbidity) were less often statistically different within and outside UGAs – in 4 of 12 months or less (Table 20).

Table 20. Frequency of occurrence of statistically significant differences (Wald F test) in monthly component Water Quality Index (WQI) scores.

Monthly WQI Component	Frequency of Statistically Significant Differences
Monthly WQI	7/12
Dissolved Oxygen	1/12
Fecal Coliform	8/12
pH	4/12
Temperature	2/12
Total Nitrogen	9/12
Total Phosphorus	10/12
Total Suspended Solids	0/12
Turbidity	1/12

Note: Detailed results can be found in Appendix C3 Table C3-3.

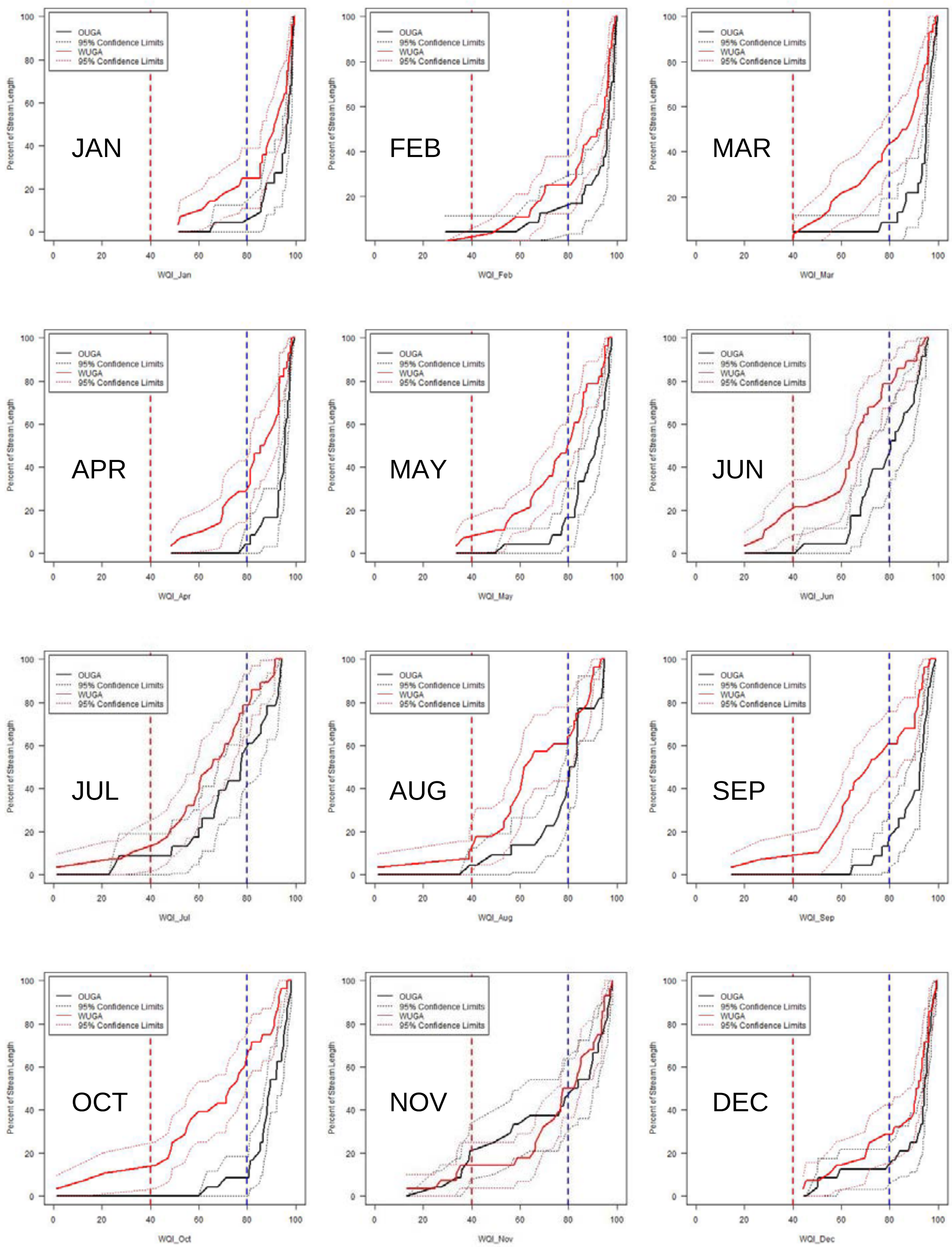


Figure 23. Monthly Water Quality Index (WQI) cumulative distribution function (CDF) plots for streams sampled outside and within Urban Growth Areas.

3.3.4 Water Quality: Spatially Adjusted Results

We assessed status for individual water quality parameters based on regulatory standards for water temperature, dissolved oxygen, pH, and fecal coliform bacteria. Total phosphorus and total nitrogen were also assessed using a combination of literature and reference values due to the relevance to the periphyton results (TDI).

Metals and organic contaminant concentrations were not evaluated beyond summary statistics or comparison to standards, because many of these constituents were detected infrequently (Appendix A and Appendix B Table B5). Of the metals that were frequently detected, they very rarely exceeded water quality standards (see Section 3.3.5.5 below). Comparisons of water quality data to reach-specific beneficial uses and state standards are provided in Section 3.3.5 below.

3.3.4.1 Fecal Coliform

Geometric mean fecal coliform concentrations were lower outside of UGAs (Figure 24 and Figure 25). The differences between the CDFs within and outside UGAs were statistically significant (Table 19 and left panel of Figure 25). Based on the primary recreational contact standard (< 100 colonies per 100 mL), 32 percent of the stream length within UGAs have geometric fecal coliform concentrations greater than the primary recreational standard, while none of the stream length outside UGAs has geometric mean concentrations greater than the standard (Figure 25).

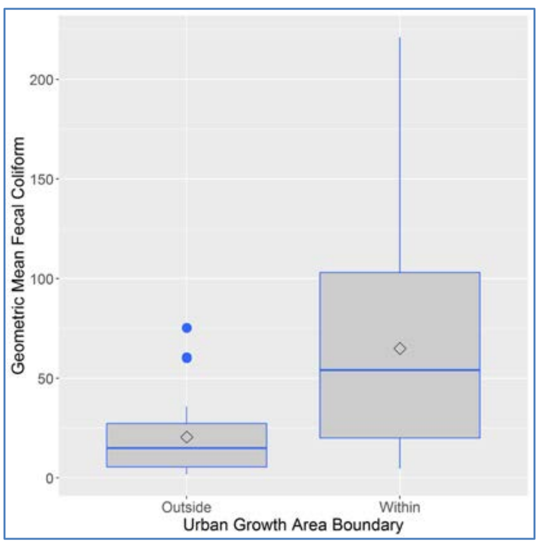


Figure 24. Geometric mean fecal coliform box plot for stream sites sampled outside and within Urban Growth Areas.

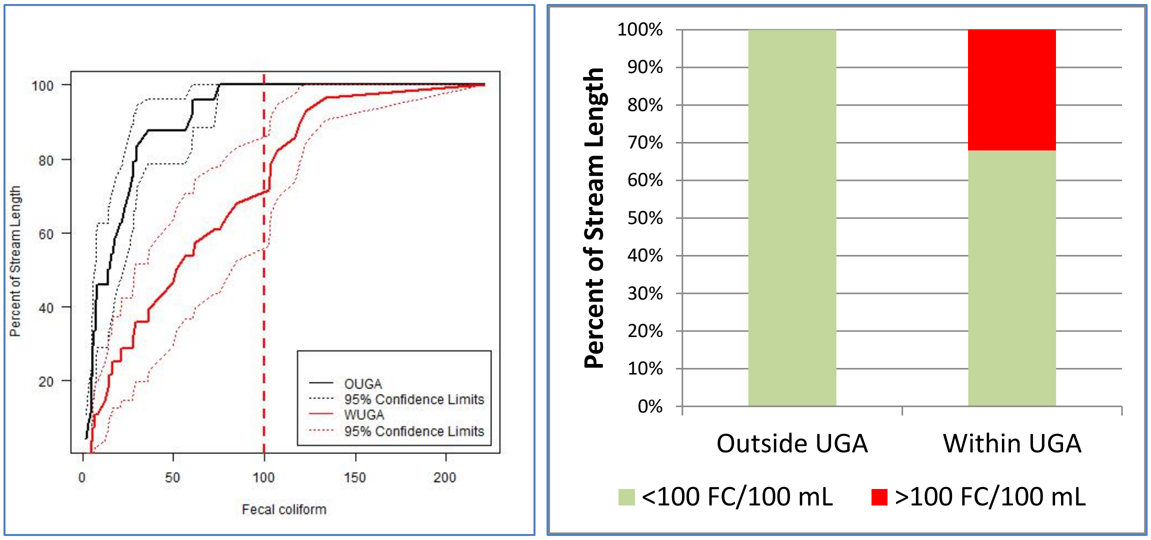


Figure 25. Geometric mean fecal coliform cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.4.2 Dissolved Oxygen

Minimum dissolved oxygen concentrations were similar within and outside of UGAs (Figure 26 and Figure 27) with no statistical difference between CDFs for within and outside UGAs (Table 19 and left panel of Figure 27Figure 26). Based on the core summer salmonid dissolved oxygen standard (dissolved oxygen not less than 9.5 mg/L), 64 percent of the stream length within UGAs had dissolved oxygen concentrations less than the 9.5 mg/L standard, and 63 percent of the stream length outside UGAs had concentrations less than the standard (Figure 27).

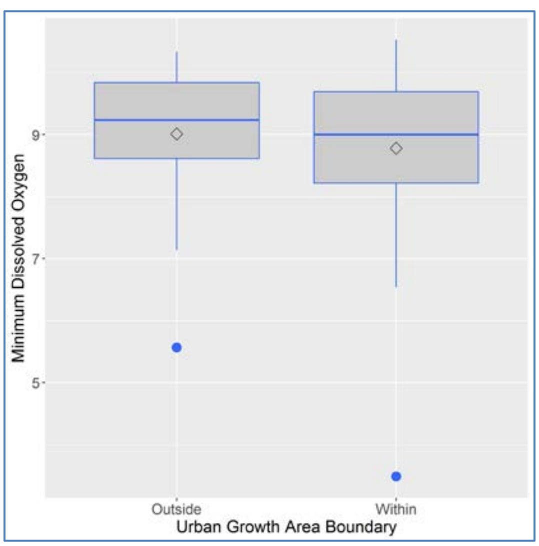
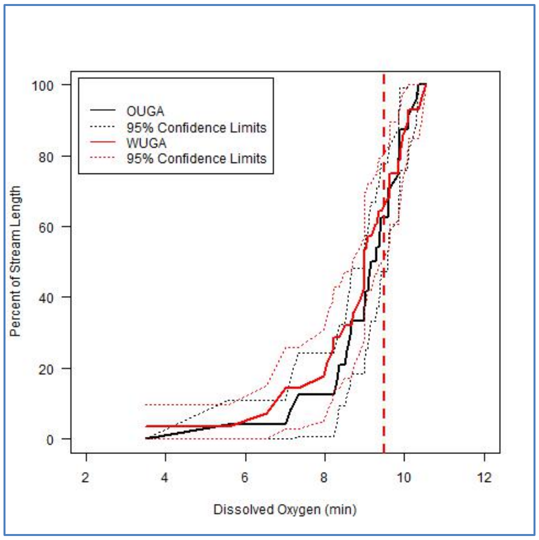


Figure 26. Minimum dissolved oxygen concentration box plot for stream sites sampled outside and within Urban Growth Areas.



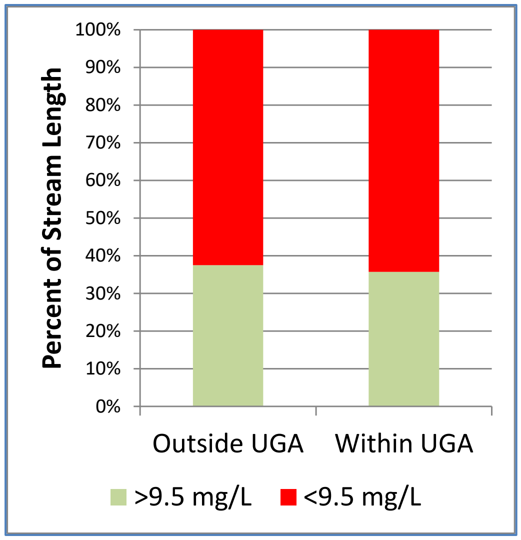


Figure 27. Minimum dissolved oxygen concentration cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.4.3 pH

Box plots of minimum and maximum pH observed at sites within and outside UGAs indicate that pH extremes were similar within and outside of UGAs, although minimum pH outside UGAs were typically somewhat lower than those measured within UGAs (Figure 28). CDF plots of minimum and maximum pH in streams within and outside UGAs illustrate a similar pattern as the box plots (Figure 29). The differences between the CDFs for minimum pH within and outside UGAs were statistically significant while the CDFs for maximum pH were not statistically significant (Table 19 and Figure 29).

Based on the core summer salmonid pH standard (pH shall be in the range of 6.5 to 8.5), 11 percent of the stream length within UGAs has minimum pH less than 6.5, and 29 percent of the stream length outside UGAs has minimum pH less than 6.5 (Figure 30). With respect to the upper pH threshold, 7 and 13 percent of the stream length within and outside UGAs, respectively, have maximum pH greater than 8.5.

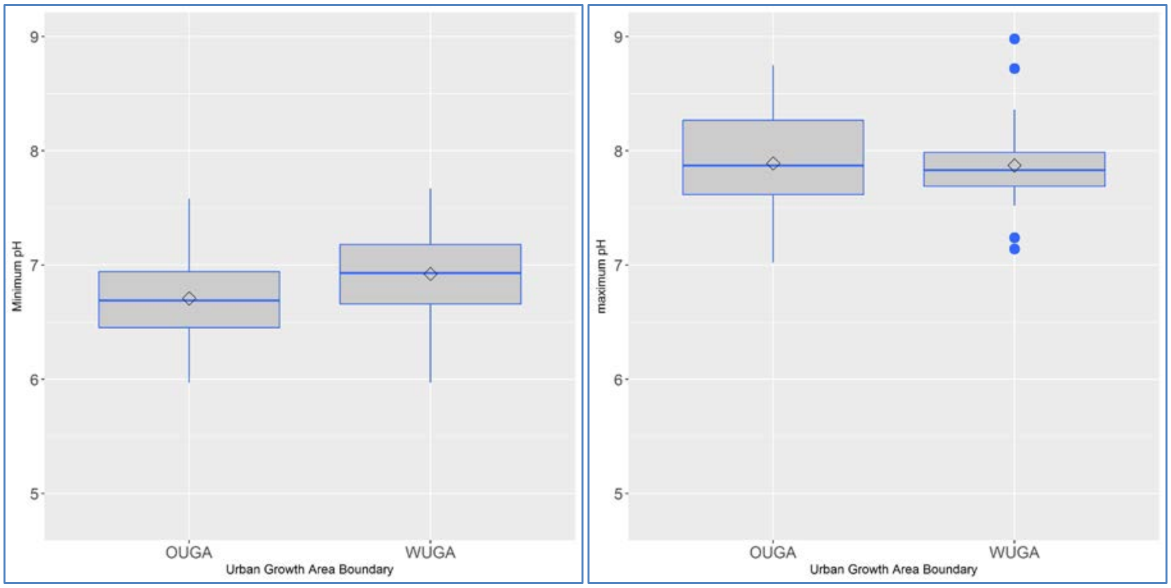


Figure 28. Box plots of the minimum (left) and maximum (right) pH measured at stream sites outside and within Urban Growth Areas.

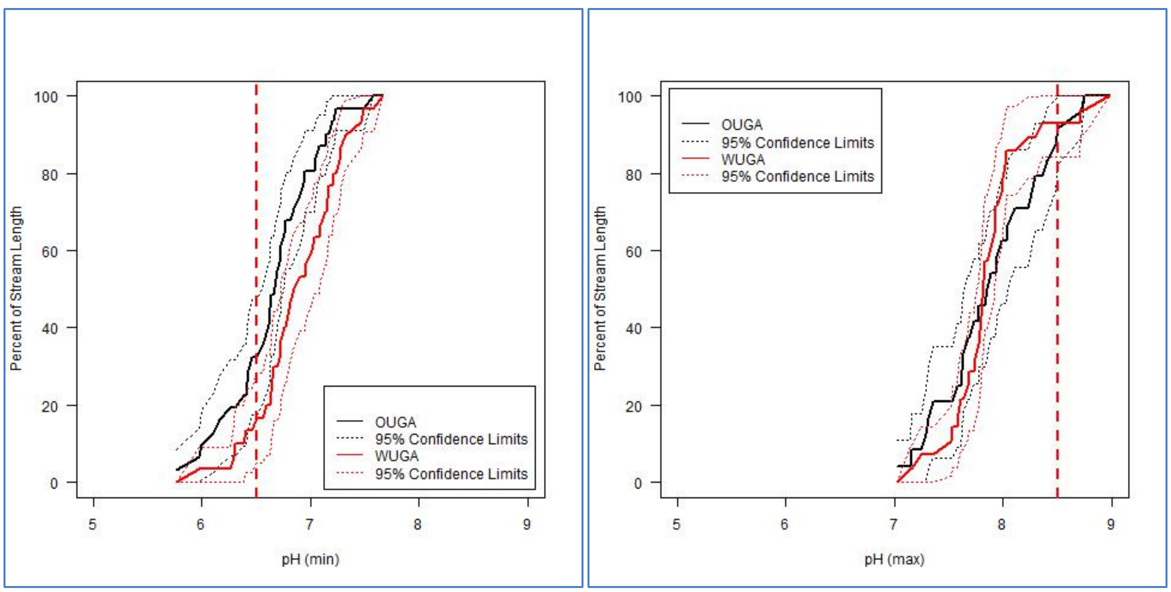


Figure 29. Cumulative distribution function (CDF) plot for the minimum (right) and maximum (left) pH concentration measured in streams outside and within Urban Growth Areas.

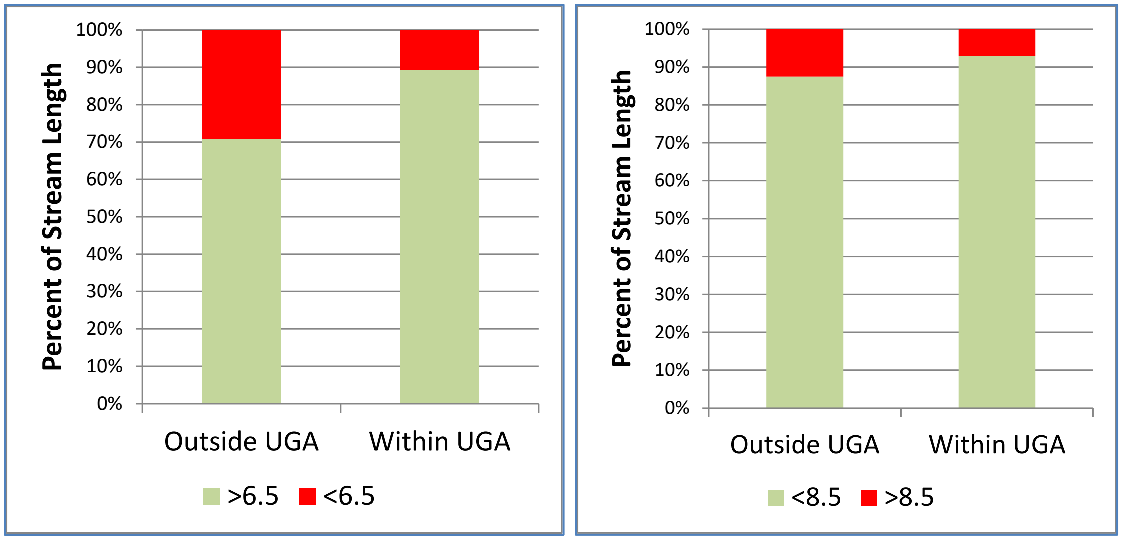


Figure 30. Categorical analysis bar plots for measured pH below the minimum (left) or above the maximum (right) pH thresholds in streams outside and within Urban Growth Areas.

3.3.4.4 Temperature

Annual maximum temperatures were similar within and outside of UGAs (Figure 31 and Figure 32) with no significant differences between the CDFs within and outside UGAs (Table 19 and left panel of Figure 32).

Temperature standards are site specific based on the designated uses of the stream. For this comparison to standards we compare our observed instantaneous values during summer months to 16 °C, the core summer salmonid temperature standard (7-day average of the daily maximum temperature). 54 percent of the stream length within and outside of UGAs had maximum temperatures greater than 16 °C (Figure 32).

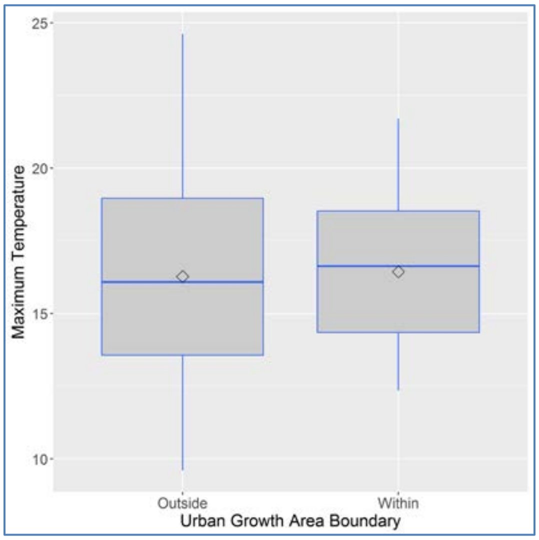
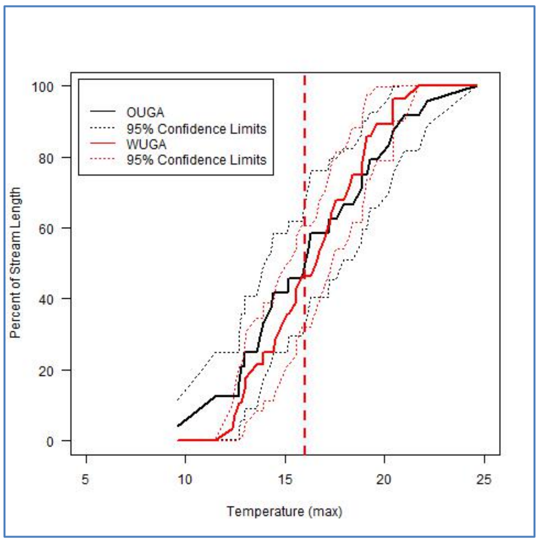


Figure 31. Maximum temperature box plot for stream sites sampled outside and within Urban Growth Areas.



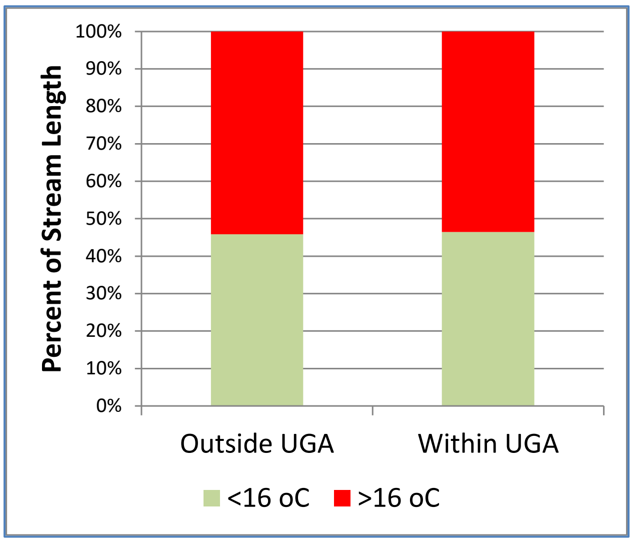


Figure 32. Maximum temperature cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.4.5 Total Phosphorus (Aug-Oct)

Higher total phosphorus concentrations potentially indicate human-caused nutrient enrichment. Total phosphorus concentrations during stream low flow conditions (Aug-Oct) were lower outside of UGAs (Figure 33 and Figure 34).[14] The differences between the CDFs within and outside UGAs were statistically significant (Table 19 and left panel of Figure 34). Based on the selected thresholds, 46 percent of the stream length within UGAs was in poor condition with respect to TP, while 8 percent of the stream length outside UGAs was in poor condition (Figure 34). A much greater proportion of stream length outside UGAs was in good condition (80 percent) relative to streams within UGAs (36 percent).

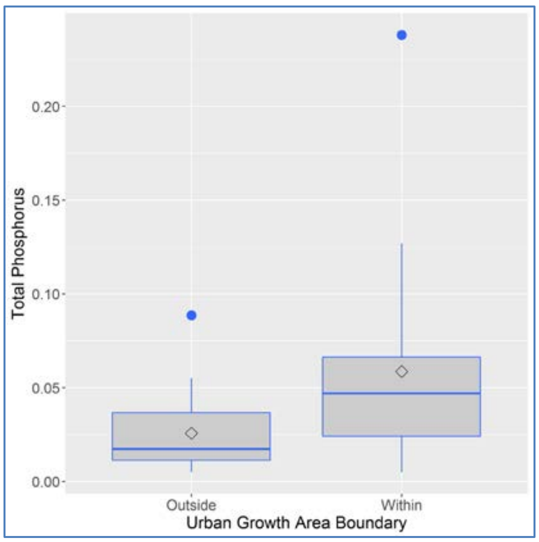


Figure 33. Mean total phosphorus concentration (Aug-Oct) box plot for sites sampled outside and within Urban Growth Areas.

[14] The averaging period Aug-Oct was selected based on various lines of evidence that suggest that late summer nutrient concentrations would most closely relate to summer periphyton biomass and/or species composition in small Puget Lowland streams.

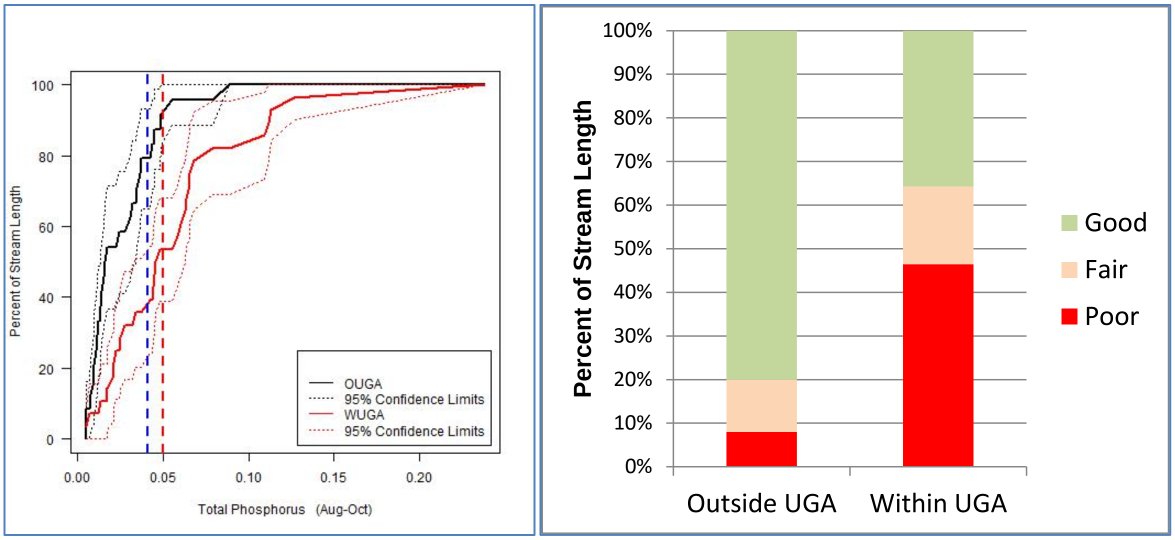


Figure 34. Mean total phosphorus (Aug-Oct) cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.4.6 Total Nitrogen (Aug-Oct)

Higher total nitrogen concentrations also potentially indicate human-caused nutrient enrichment. Total nitrogen concentrations during stream low flow conditions (Aug–Oct) were lower outside of UGAs (Figure 35 and Figure 36). The differences between the CDFs within and outside UGAs were statistically significant (Table 19 and left panel of Figure 36). Based on the selected thresholds, 43 percent of the stream length within UGAs was in poor condition with respect to TN, while 12 percent of the stream length outside UGAs was in poor condition (Figure 36). A much greater proportion of stream length outside UGAs was in good condition (68 percent) relative to streams within UGAs (39 percent).

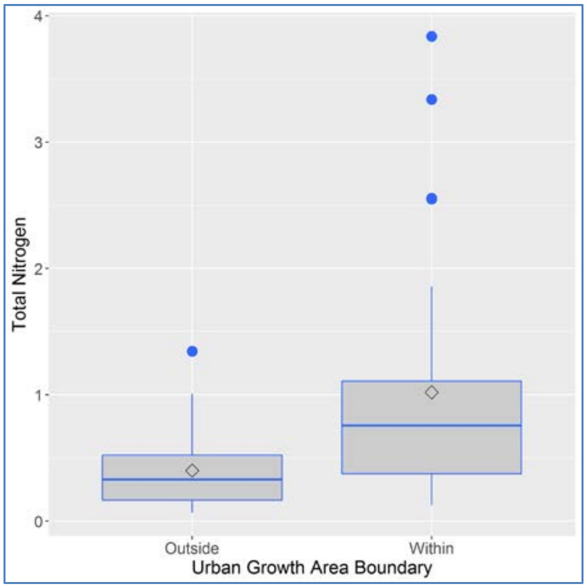
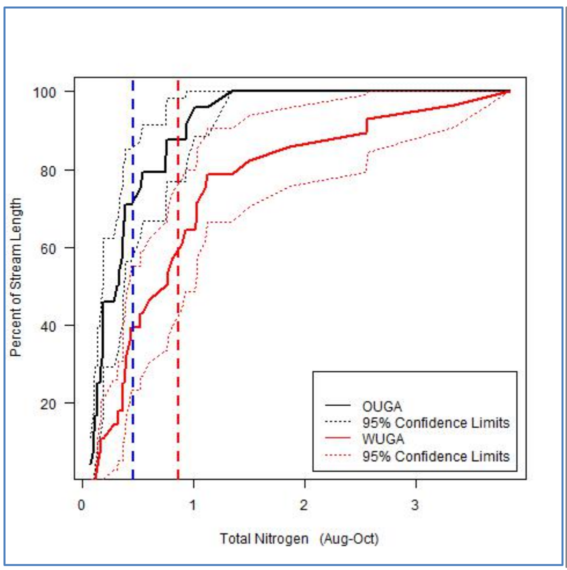


Figure 35. Mean total nitrogen (Aug-Oct) box plot for stream sites sampled outside and within Urban Growth Areas.



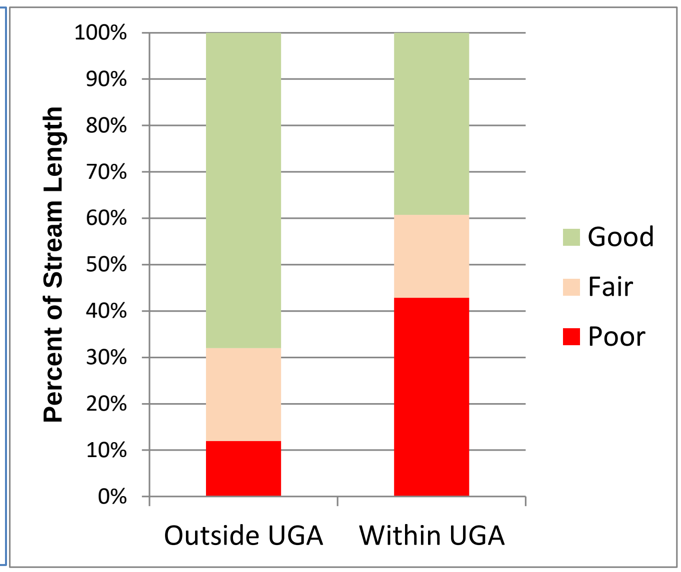


Figure 36. Mean total nitrogen (Aug-Oct) cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.5 Water Quality: Site-specific Comparisons to Water Quality Standards

In order to specifically evaluate the status of the SAM PLES stream population with respect to these parameters, a different set of spatial weights would need to be created (one for fecal coliform and another for temperature, dissolved oxygen, and pH) to account for the length of small streams assigned to each beneficial use category within and outside UGAs. This was beyond the scope of this study. Therefore, in this section, we simply compared the data to the specific criteria, and presented results based on number of sites above or below the standards, rather than extrapolating to stream lengths.

We identified the relevant beneficial use of each of the sample site location, which allowed for an assessment of the number of sampled stream sites that met (or did not meet) state water quality standards for temperature, dissolved oxygen, pH, and fecal coliform bacteria based on site-specific beneficial use designations. The focus of the comparison to water quality standards was primarily on standards for the protection of aquatic life, with the exception of the fecal coliform standards for the protection of human health.

3.3.5.1 Fecal Coliform

A similar pattern to the one based on comparison to a single threshold (see Section 3.3.4) was apparent with a greater number of exceedances of the relevant criteria within UGAs. Almost half of the sites within UGAs with complete monthly data exceeded the relevant contact recreation criteria, while only 12 percent of the sites outside of UGAs exceeded relevant criteria (Table 21).

Table 21. Number of sites within and outside urban growth areas with a complete year of monthly fecal coliform data that exceeded the appropriate geometric mean criterion.

Human Recreational Use (criterion)	OUGA	WUGA
Extraordinary Primary Contact (50 cfu/100 mL)	3/12 (25%)	10/18 (56%)
Primary Contact (100 cfu/100 mL)	0/14 (0%)	7/20 (35%)
Overall	3/26 (12%)	17/38 (45%)

3.3.5.2 Dissolved Oxygen

Based on comparison to relevant state dissolved oxygen criteria, a similar pattern to the one based on comparison to a single threshold was apparent. Generally, minimum dissolved oxygen concentrations observed at about half of the stream sites within and outside UGAs were below relevant state criteria (Table 22).

Table 22. Number of sites within and outside urban growth areas with a complete year of monthly dissolved oxygen data where minimum dissolved oxygen was below the appropriate criterion.

Aquatic Species Use (criterion)	OUGA	WUGA
Char Spawning/Rearing (9.5 mg/L)	0/1 (0%)	NA
Core Summer Habitat (9.5 mg/L)	14/23 (61%)	11/22 (50%)
Spawning/Rearing (8.0 mg/L)	1/2 (50%)	3/9 (33%)
Overall	15/26 (60%)	14/31 (56%)

3.3.5.3 pH

The annual minimum pH was below the minimum pH criterion at a larger proportion of sites with more minimum pH values observed below the criterion at outside relative to within UGA sites (Table 23). Less than 10 percent of sites within or outside UGAs exceeded the maximum pH criterion (Table 24).

Table 23. Number of sites within and outside urban growth areas with a complete year of monthly pH data where minimum pH was below the appropriate criterion.

Aquatic Species Use (criterion)	OUGA	WUGA
Char Spawning/Rearing (6.5)	0/1 (0%)	NA
Core Summer Habitat (6.5)	8/23 (35%)	5/22 (23%)
Spawning/Rearing (6.5)	2/2 (100%)	2/10 (20%)
Overall	10/26 (38%)	7/32 (29%)

Table 24. Number of sites within and outside urban growth areas with a complete year of monthly pH data where the maximum pH exceeded the appropriate criterion.

Aquatic Species Use	OUGA	WUGA
Char Spawning/Rearing (8.5)	0/1 (0%)	NA
Core Summer Habitat (8.5)	1/23 (4%)	1/22 (5%)
Spawning/Rearing (8.5)	0/2 (0%)	1/10 (10%)
Overall	1/26 (4%)	2/32 (9%)

3.3.5.4 Temperature

Based on comparison to relevant state temperature criteria, a similar pattern to the one based on the probabilistic comparison using a single threshold was observed. The maximum temperature exceeded the relevant criterion at about 50 percent of the sites both outside and within UGAs (Table 25).

Table 25. Number of sites within and outside urban growth areas with a complete year of monthly temperature data that exceeded the appropriate criterion.

Aquatic Species Use (criterion)	OUGA	WUGA
Char Spawning/Rearing (12 ^{o}C)	1/1 (100%)	NA
Core Summer Habitat (16 ^{o}C)	13/23 (57%)	13/22 (59%)
Spawning/Rearing (17.5 ^{o}C)	1/2 (50%)	1/10 (10%)
Overall	15/26 (58%)	14/32 (44%)

3.3.5.5 Metals

Stream water column metals concentrations were below state standards for the protection of aquatic life (acute and chronic) for all but 2 of over 740 grab samples collected from the SAM regional and Option 2 sites. The February 2015 sample collected from a site on the Upper Deschutes River (RSM06600-001702) in Thurston County exceeded both acute and chronic standards for cadmium, copper, and zinc; and the chronic standard for lead. The chronic lead criterion was exceeded in another single sample collected in November 2015 from Jim Creek (RSM06600-002596), a tributary to the South Fork Stillaguamish River in Snohomish County. Both of these sites were outside UGAs.

3.3.6 All Other Water Quality Parameters

Other water quality parameters that were measured, but not discussed above, include conductivity, chloride, orthophosphate phosphorus, nitrate+nitrite, ammonia nitrogen, dissolved organic carbon, and PAHs.

The frequency of detection of these parameters is summarized in Appendix A Table A1 and summary statistics for results from sites within and outside UGAs is provided in Appendix B Table B4 (in situ measurement, including conductivity) and Appendix B Table B5 (water chemistry parameters). The summary of CDF results for annual median concentrations are provided in Appendix C4 (in situ measurement, including conductivity) and Appendix C5 (water chemistry parameters).

Consistent with research that has identified a link between urbanization and increasing conductivity and chloride concentrations (Roy et al., 2003; Kaushal et al., 2005; Kaushal et al., 2018), conductivity and chloride concentrations were higher within compared to outside UGAs.[15] Although the median chloride concentration was higher within UGAs, the highest chloride concentrations were found at sites outside UGAs. Conductivity was more consistently higher at sites within UGAs. The human sources of chloride and dissolved solutes that contribute to increases in conductivity include septic systems, leaky sewers, concrete weathering, road salts and other deicing agents, and releases of domestic animal wastes to streams (Kaushal et al., 2018). Another potential source is the removal of the solute depleted upper soil horizon as a result of forest removal and development that exposes a less weathered soil horizon that more readily releases solutes to infiltrating water that eventually reaches the stream (Edmondson, 1994).

Ammonia was detected more frequently at sites within UGAs relative to sites outside UGAs (Appendix B Table B5). Ammonia was not detected frequently enough to allow for statistical comparisons of concentrations within and outside UGAs. Unionized ammonia concentrations did not exceed the state water quality standards. Orthophosphate and nitrate+nitrite concentrations were detected frequently enough to statistically compare concentrations measured within and outside UGAs. Consistent with the total phosphorus and total nitrogen results, the distribution of annual median concentrations was

[15] Hardness (a measure of the calcium and magnesium concentration) used to calculate relevant state metal standards for freshwater was also higher within versus outside UGAs (Appendix C5 Table C5-3).

significantly higher within UGAs. The annual median concentration of dissolved organic carbon was not statistically different within versus outside UGAs.

With the exception of naphthalene, PAHs were very infrequently detected (Appendix A). Naphthalene was detected in 24 percent of the samples, while the detection frequency of the remaining PAH compounds was 3 percent or lower. Although the frequency of detection was generally low, the frequency of detection was typically higher at sites within UGAs (Appendix B Table B5). There are no state water quality standards for the protection of aquatic life for PAHs, but the concentrations of individual PAHs detected at three sites did exceed chronic screening levels provided by the National Oceanic and Atmospheric Administration (NOAA, 2008) and U.S. EPA Region 3 (Pluta, 2006) (Table 26). Even though naphthalene was frequently detected, the highest detected naphthalene concentration measured in Carey Creek (RSM06600-002259), a tributary to Issaquah Creek in King County, on May 27, 2015 (0.214 µg/L) did not exceed the chronic screening levels provided in NOAA (2008) and Pluta (2006) of 1.1 µg/L.

Table 26. Sites where water detected polycyclic aromatic hydrocarbons (PAHs) exceeded screening levels.

Parameter / Location ID	Name	Date	Concentration (µg/L)	Chronic Screening Level (µg/L)	Screening Level Reference
Benzo(a)pyrene					
RSM06600-001550	Skookum Creek [a]	12/1/2015	0.04	0.015	Pluta (2006)
RSM06600-005456	Whatcom Creek [b]	4/22/2015	0.027		
Anthracene					
RSM06600-001550	Skookum Creek [a]	8/12/2015	0.015	0.012	Pluta (2006)
Fluoranthene					
RSM06600-001550	Skookum Creek [a]	12/1/2015	0.047	0.04	NOAA (2008), Pluta (2006)
Pyrene					
RSM06600-001550	Skookum Creek [a]	12/1/2015	0.042	0.025	Pluta (2006)
RSM06600-013054	Dumas Creek [b]	11/17/2015	0.025		

[a] *Outside Urban Growth Area (OUGA) site*
[b] *Inside Urban Growth Area (WUGA) site*

Monthly instantaneous flow measurements were made at each site during the study. The distribution of median annual flow was not significantly different within versus outside UGAs (Appendix C4 Table C4-3). However, the highest flow recorded during the study was 1,620 cfs at an outside UGA site on the Raging River (RSM06600-004615) in King County on November 17, 2015. This flow was four times higher than the next highest flow observed. Generally, the magnitude and variability of flow at a particular site will be determined by drainage area, elevation, and surficial geology. Forest removal and development of impervious cover increasingly connected to the stream then modifies the natural flow regime. Historically, evaluation of development and stormwater management on flow has relied on continuous flow measurements that more readily capture the effects of impervious cover and management activities on the frequency, magnitude, and duration of flow (e.g., Konrad et al., 2005). Continuous flow measurements have also typically been used to evaluate potential relationships of flow with biological responses (e.g., Konrad et al., 2008).

The use of stage data to characterize hydrologic conditions has been explored in some studies (e.g., McMahon et al., 2003; Booth and Konrad, 2017) and stage was recorded at sites without established flow gages as part of the USGS Pacific Northwest Stream Quality Assessment (Sheibley et al., 2017b). Measurement of stage without the additional field and desktop effort to develop and update stage-discharge relationships to estimate flow has the potential to provide more cost-effective hydrologic response information.

Note that monthly water quality sampling (and flow measurements) were essentially random (i.e., there was no attempt to time sampling with storm events). Therefore, it was not possible to relate observed concentrations to storm events. It may be possible to identify sites and dates that characterize storm event conditions as an extension to this study, but this may require the determination of site-specific antecedent rainfall conditions for each sampling event – data that is not readily available for each site.

3.3.7 Sediment Quality: Spatially Adjusted Results

Sediment samples were collected from depositional areas of the streambed within the wetted channel at the same sites where biological samples were collected. All sediment was sieved to exclude large material from the sediment chemistry samples. Approximately 50 parameters were measured, including:

- Metals: (<0.063 mm sieve size fraction) arsenic, cadmium, chromium, copper, lead, silver, and zinc.[16]
- Organic contaminants: (<2.0 mm sieve size fraction) PAHs, common roadside use pesticides, phthalates, polychlorinated biphenyl compounds (PCBs), and polybrominated diphenyl ethers (PBDEs).

This section focuses on metals (arsenic, cadmium, chromium, copper, lead, and zinc) and organic contaminants (total PAH, total PCB, total PBDE, and dichlobenil). These metals and organic compounds were frequently detected (in greater than 50 percent of the samples within and outside UGAs), with the exception of total PAHs. PAH compounds were detected in 68 percent of sites within UGAs, but PAHs were detected in only 10 percent of the samples collected at sites outside UGAs (Appendix B Table B6). With the exception of the common roadside use pesticide dichlobenil, these metals and organic contaminants also have established ecologically-relevant thresholds.

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[16] At six stations the volume of fine (<0.063 mm sieve size) sediment was insufficient for analysis. The 2.0 mm sieve fraction was analyzed for metals at these sites. Concentrations measured in the coarser fraction will generally be lower. Because there were no paired sieve fraction samples analyzed (analysis of 2.0 and 0.063 mm sieve samples from the same site) it is not possible to make any adjustment to the data.

The overall frequency of detection of sediment metals and organic contaminants is summarized in Appendix A Table A2 (metals, PAHs, phthalates, and pesticides) and Appendix A Table A3 (PCB and PBDE congeners). Summary statistics for within and outside UGA sites is provided in Appendix B Table B6. The summary of CDF analyses are provided in Appendix C6. Comparisons of sediment quality data to state standards are provided in Section 3.3.8 below.

3.3.7.1 Arsenic

Box plots and CDF plots of arsenic concentrations observed in sediments within and outside UGAs indicate that sediment arsenic concentrations were somewhat higher within relative to outside of UGAs (Figure 37 and Figure 38). The differences between the CDFs within and outside UGAs were statistically significant (Table 19 and the left panel in Figure 38). Based on threshold effect and probable effect concentrations, less than 1 percent of the stream length outside UGAs was in poor condition (Figure 38). A greater proportion of stream length outside UGAs was in good condition (72 percent) relative to streams within UGAs (51 percent). In the sediment CDF figures below, the vertical blue dashed line (lower threshold) represents the TEC and the red dashed line (higher threshold) is the PEC.

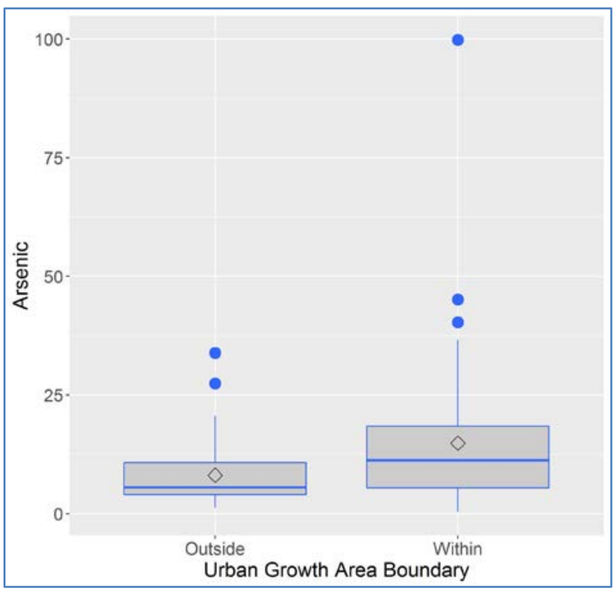


Figure 37. Sediment arsenic concentration box plot for stream sites sampled outside and within Urban Growth Areas.

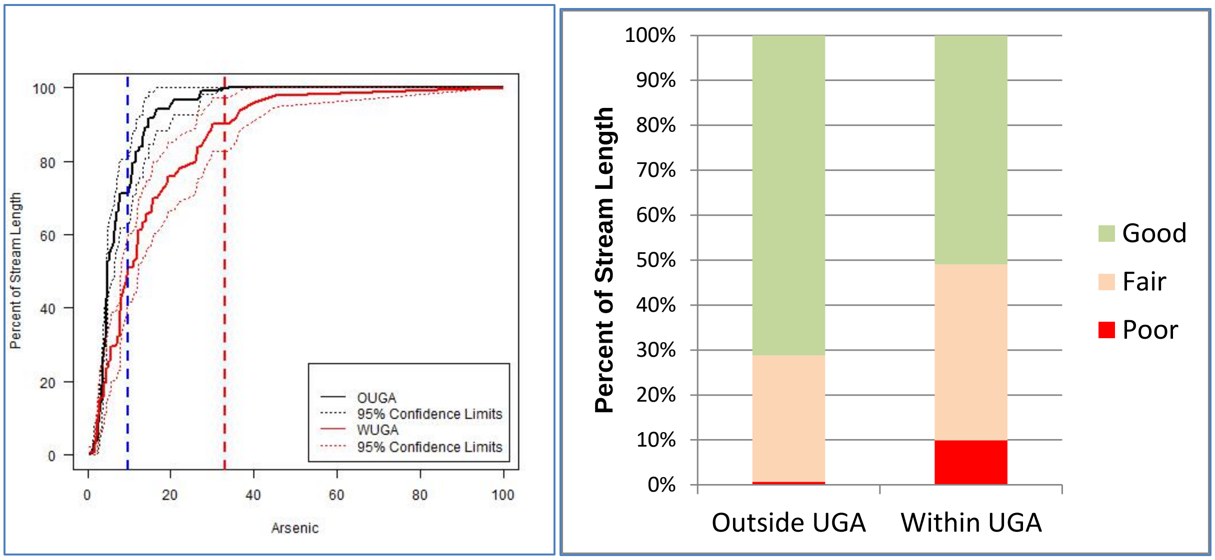


Figure 38. Sediment arsenic concentration cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.7.2 Cadmium

Box plots and CDF plots of cadmium concentrations observed in sediments within and outside UGAs indicate that sediment cadmium concentrations were somewhat higher within relative to outside of UGAs, although the highest sediment concentration was observed outside UGAs (Figure 39 and Figure 40). The differences between the CDFs within and outside UGAs were statistically significant (Table 19 and left panel of Figure 40). Based on threshold effect and probable effect concentrations, 3 percent of the stream length outside UGAs was in poor condition (Figure 40). A greater proportion of stream length outside UGAs was in good condition (97 percent) relative to streams within UGAs (92 percent).

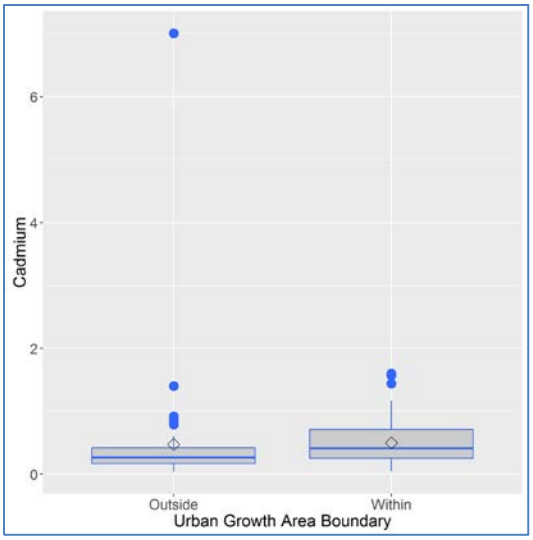


Figure 39. Sediment cadmium concentration box plot for stream sites sampled outside and within Urban Growth Areas.

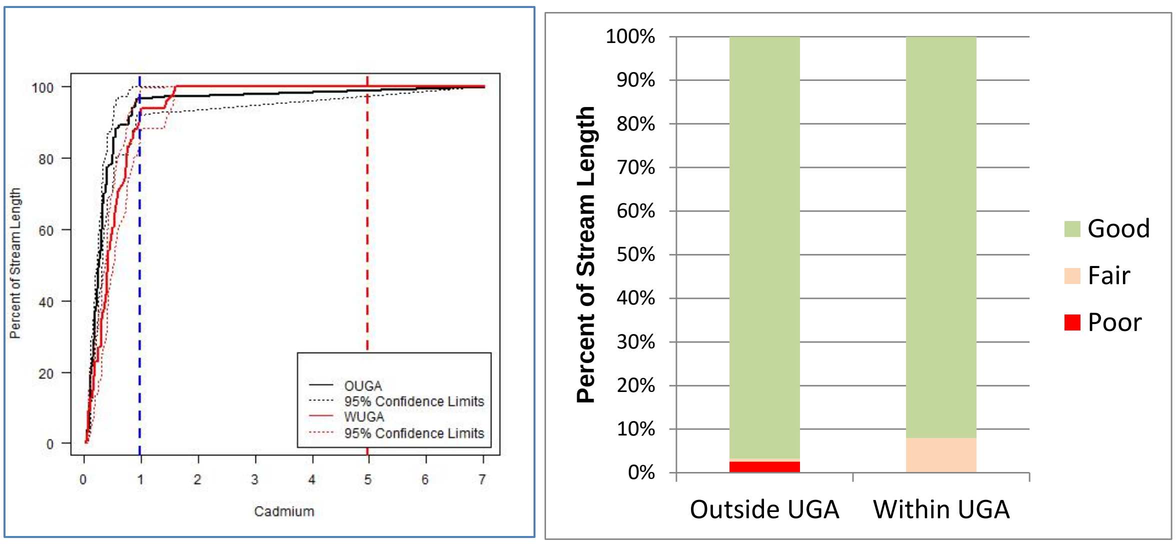


Figure 40. Sediment cadmium concentration cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.7.3 Chromium

Box plots and CDF plots of chromium concentrations observed in sediments within and outside UGAs indicate that sediment chromium concentrations were fairly similar within and outside of UGAs, although the highest sediment concentration was observed outside UGAs (Figure 41 and Figure 42). The differences between the CDFs within and outside UGAs were not statistically significant (Table 19 and left panel of Figure 42). Based on threshold effect and probable effect concentrations, 3 percent of the stream length outside and 2 percent of stream length within UGAs was in poor condition with respect to sediment chromium concentrations (Figure 42). Similar proportions of stream length outside and within UGAs were in good condition, 48 and 43 percent, respectively.

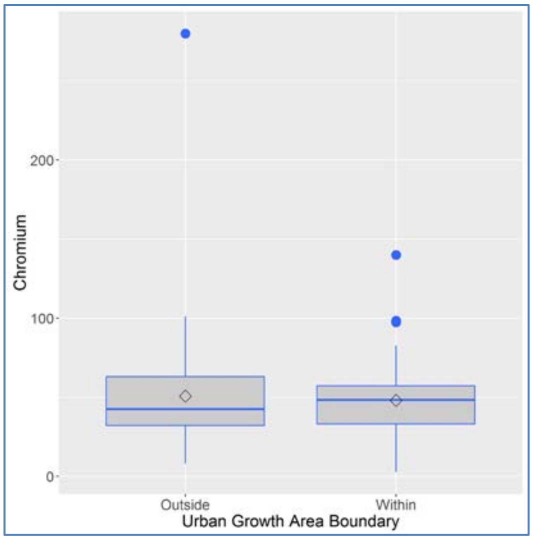
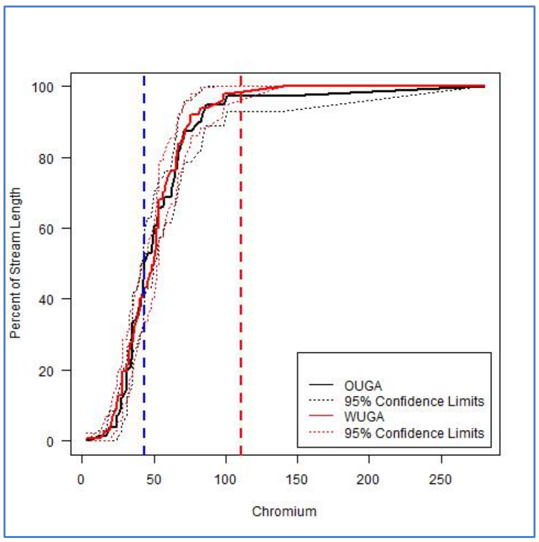


Figure 41. Sediment chromium concentration box plot for stream sites sampled outside and within Urban Growth Areas.



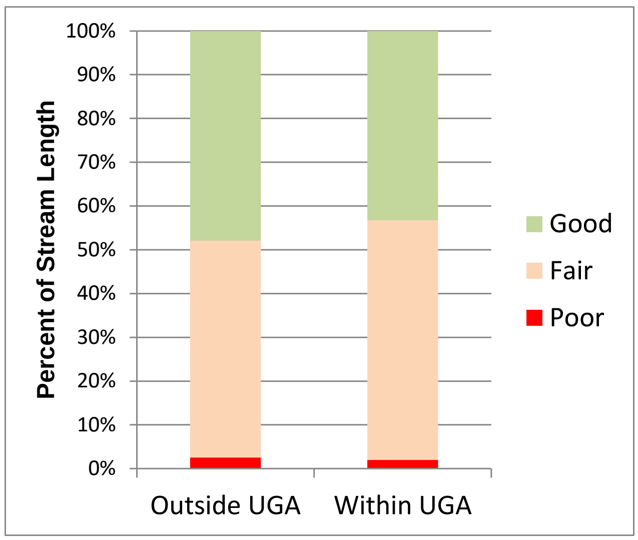


Figure 42. Sediment chromium concentration cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.7.1 Copper

Box plots and CDF plots of copper concentrations observed in sediments within and outside UGAs indicate that sediment copper concentrations were fairly similar within and outside of UGAs, although the highest sediment concentrations were observed within UGAs (Figure 43 and Figure 44). The differences between the CDFs within and outside UGAs were not statistically significant (Table 19 and left panel of Figure 44). Based on threshold effect and probable effect concentrations, 3 percent of the stream length outside and 6 percent of stream length within UGAs was in poor condition with respect to sediment chromium concentrations (Figure 44). Similar proportions of stream length outside and within UGAs were in good condition, 35 and 40 percent, respectively.

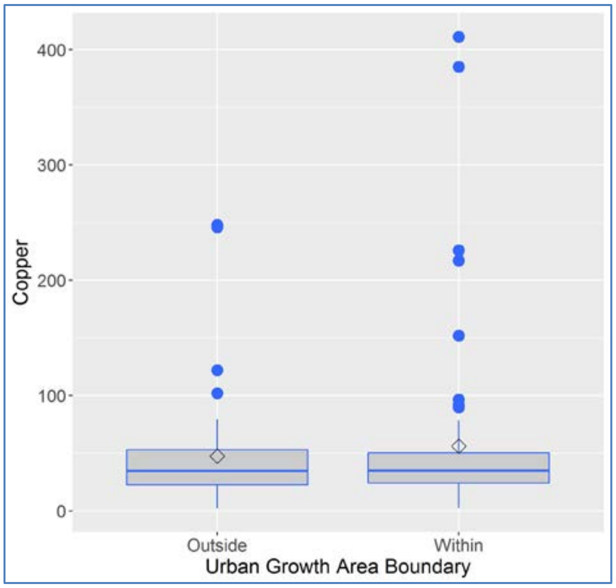


Figure 43. Sediment copper concentration box plot for stream sites sampled outside and within Urban Growth Areas.

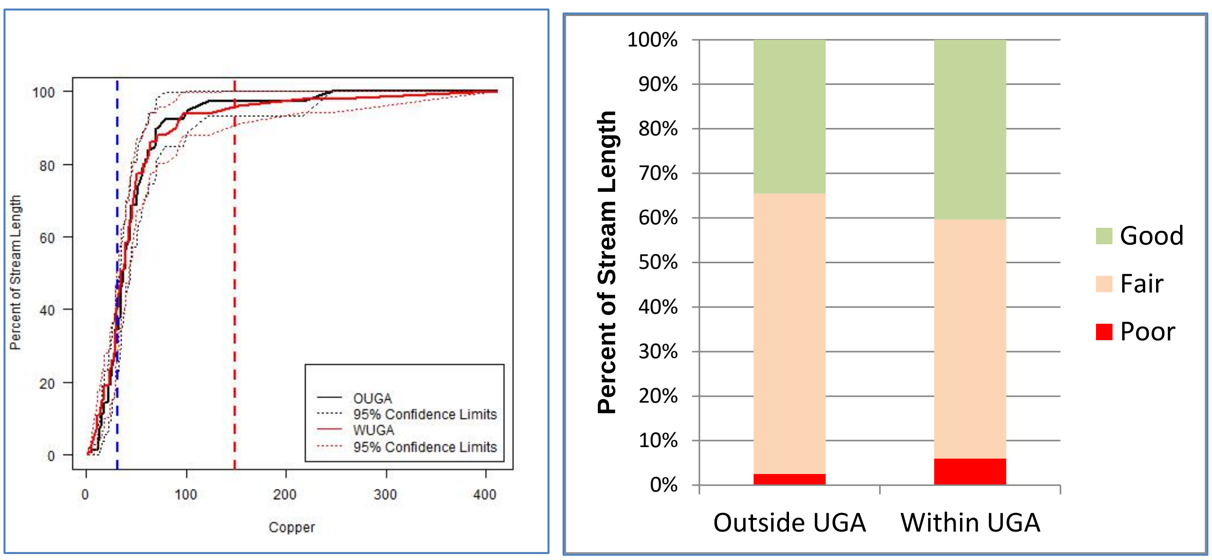


Figure 44. Sediment copper concentration cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.7.2 Lead

Box plots and CDF plots of lead concentrations observed in sediments within and outside UGAs indicate that sediment lead concentrations were typically higher within relative to outside of UGAs (Figure 45 and Figure 46). The differences between the CDFs within and outside UGAs were statistically significant (Table 19 and left panel of Figure 46). Based on threshold effect and probable effect concentrations, 4 percent of the stream length within UGAs was in poor condition and 3 percent outside UGAs was in poor condition (Figure 46).

A greater proportion of stream length outside UGAs was in good condition (92 percent) relative to streams within UGAs (70 percent).

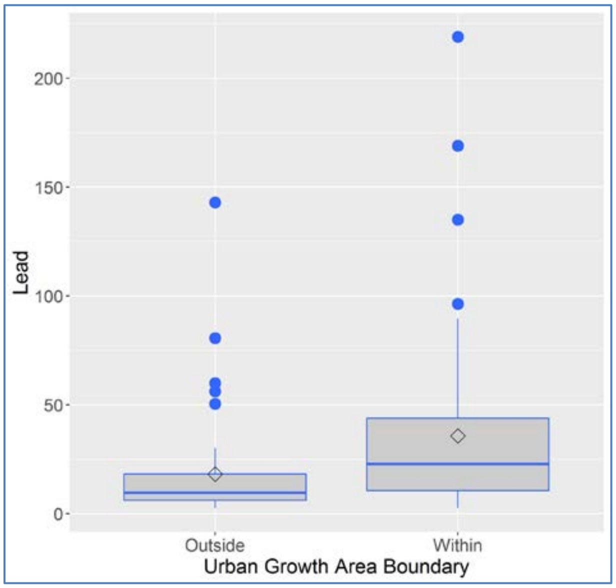


Figure 45. Sediment lead concentration box plot for stream sites sampled outside and within Urban Growth Areas.

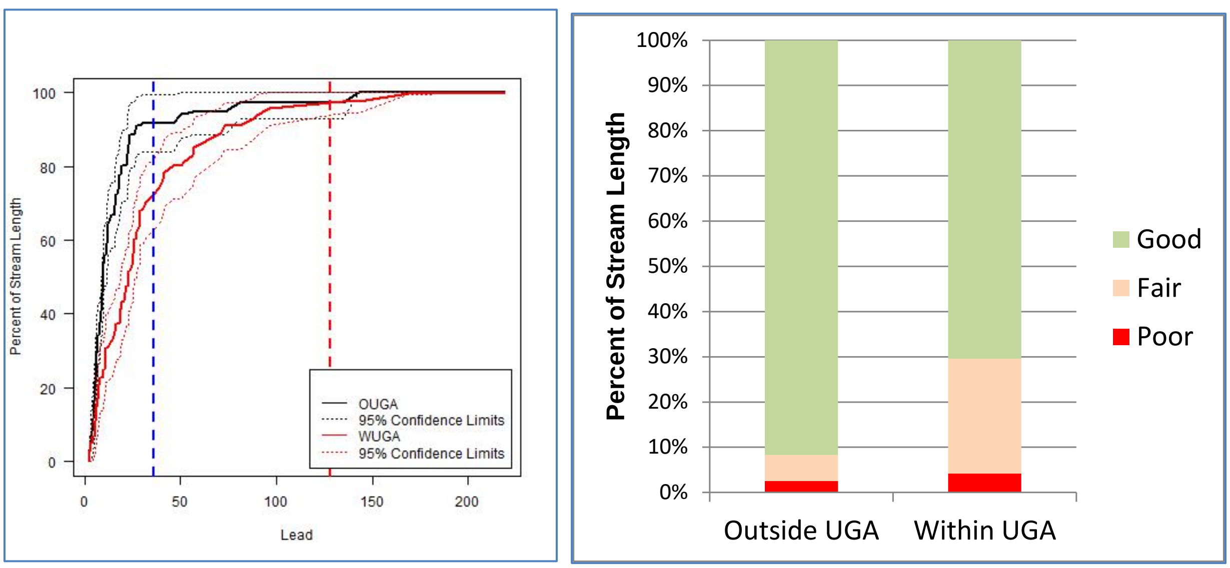


Figure 46. Sediment lead concentration cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.7.3 Zinc

Box plots and CDF plots of zinc concentrations observed in sediments within and outside UGAs indicate that sediment zinc concentrations were typically higher within relative to outside of UGAs (Figure 47 and Figure 48). The differences between the CDFs within and outside UGAs were statistically significant (Table 19 and left panel of Figure 48). Based on threshold effect and probable effect concentrations, 2 percent of the stream length within UGAs was in poor condition with respect to sediment zinc concentrations (Figure 48). A greater proportion of stream length outside UGAs was in good condition (85 percent) relative to streams within UGAs (39 percent).

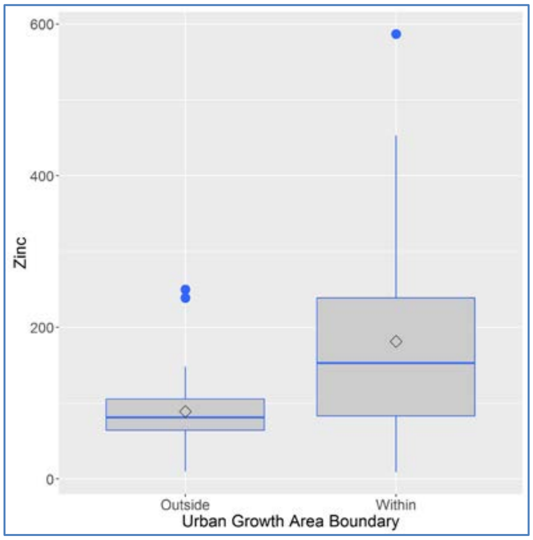


Figure 47. Sediment zinc concentration box plot for stream sites sampled outside and within Urban Growth Areas.

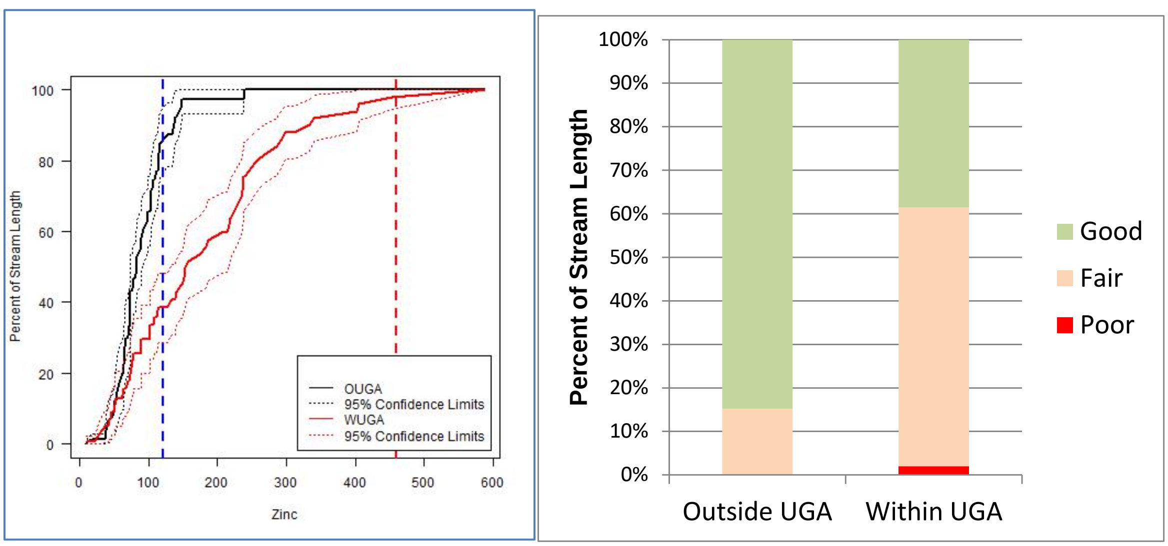


Figure 48. Sediment zinc concentration cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.7.4 Total PAH

Box plots and CDF plots of total PAH concentrations observed in sediments within and outside UGAs indicate that sediment PAH concentrations were typically higher within relative to outside of UGAs (Figure 49 and Figure 50). The differences between the CDFs within and outside UGAs could not be determined statistically because total PAH was reliably quantified in samples from only 10 percent of sites outside UGAs (Table 19). Based on threshold effect and probable effect concentrations, 2 percent of the stream length within UGAs was in poor condition with respect to sediment total PAH concentrations (Figure 50). A greater proportion of stream length outside UGAs was in good condition (100 percent) relative to streams within UGAs (89 percent).

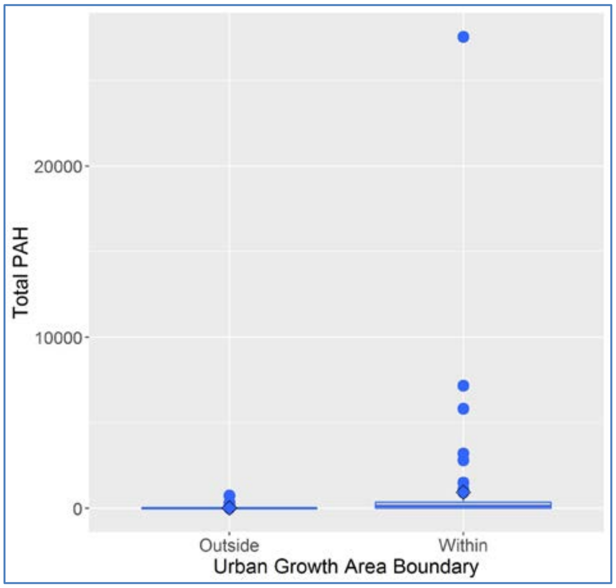


Figure 49. Sediment total PAH concentration box plot for stream sites sampled outside and within Urban Growth Areas.

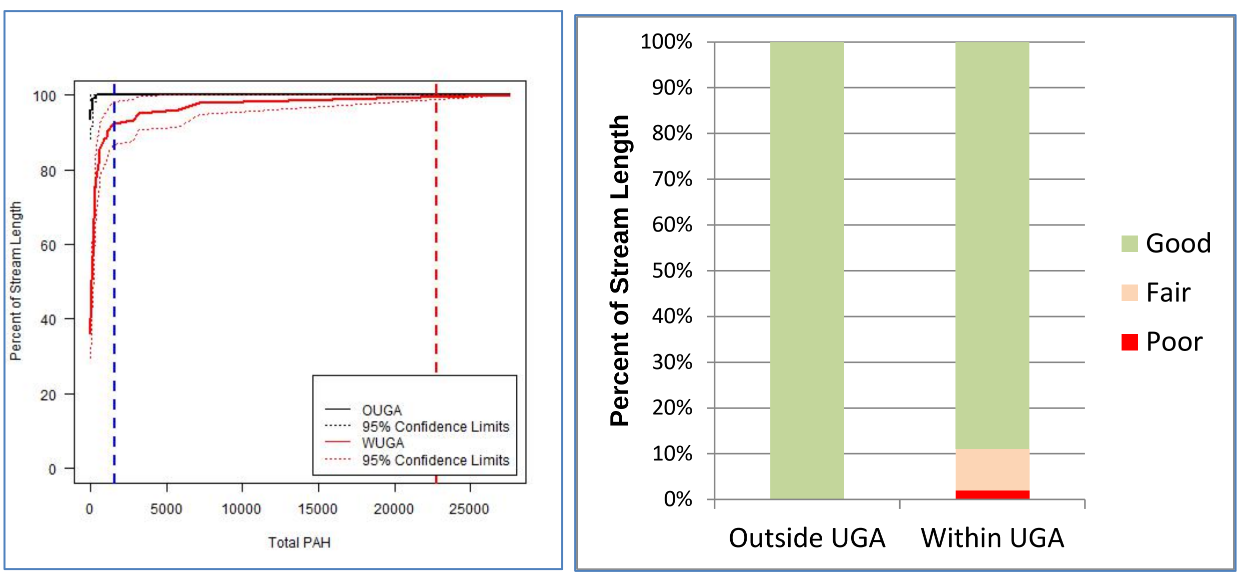


Figure 50. Sediment total PAH concentration cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.7.1 Total PCB

Box plots and CDF plots of total PCB concentrations observed in sediments within and outside UGAs indicate that sediment PCB concentrations were typically higher within relative to outside of UGAs (Figure 51 and Figure 52). The differences between the CDFs within and outside UGAs were statistically significant (Table 19 and left panel of Figure 52). Based on threshold effect and probable effect concentrations, none of the stream length within UGAs was in poor condition with respect to sediment total PCB concentrations (Figure 52). A slightly greater proportion of stream length outside UGAs was in good condition (100 percent) relative to streams within UGAs (95 percent).

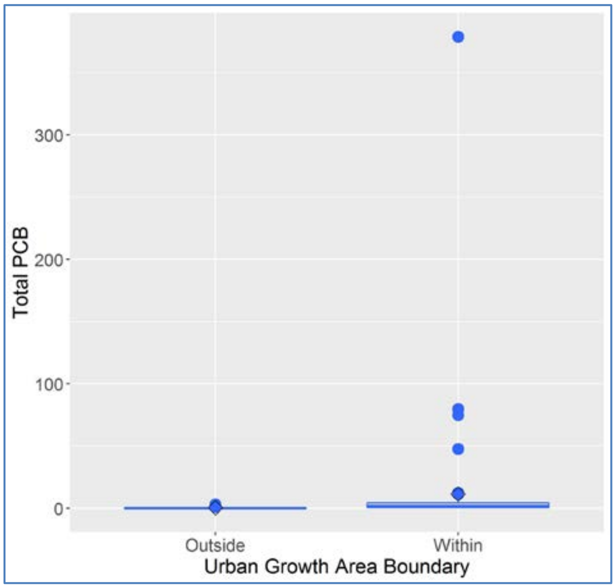
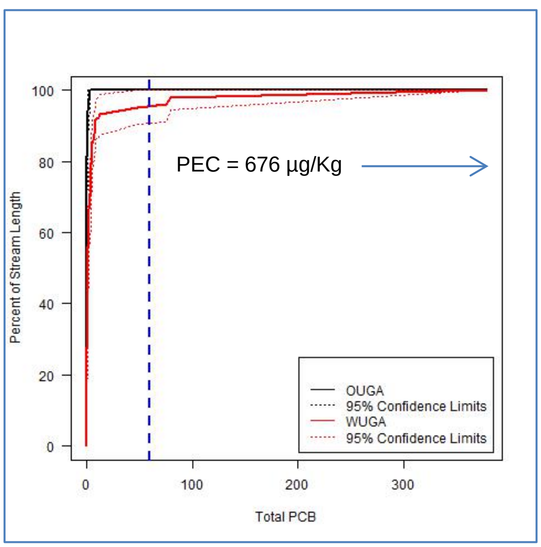


Figure 51. Sediment total PCB concentration box plot for stream sites sampled outside and within Urban Growth Areas.



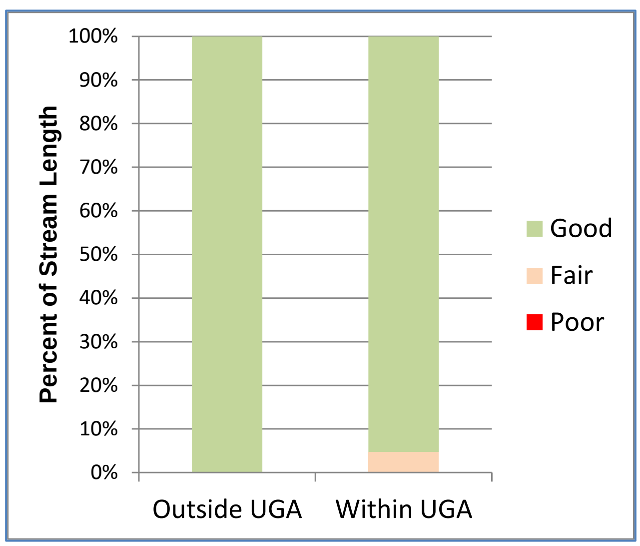


Figure 52. Sediment total PCB concentration cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.7.2 Total PBDE

Box plots and CDF plots of total PBDE concentrations observed in sediments within and outside UGAs indicate that sediment PBDE concentrations were typically higher within relative to outside of UGAs (Figure 53 and Figure 54). The differences between the CDFs within and outside UGAs were statistically significant (Table 19 and left panel of Figure 54). Based on threshold effect and probable effect concentrations, none of the stream length within UGAs was in poor condition with respect to sediment total PBDE concentrations (Figure 54). A slightly greater proportion of stream length outside UGAs was in good condition (100 percent) relative to streams within UGAs (93 percent).

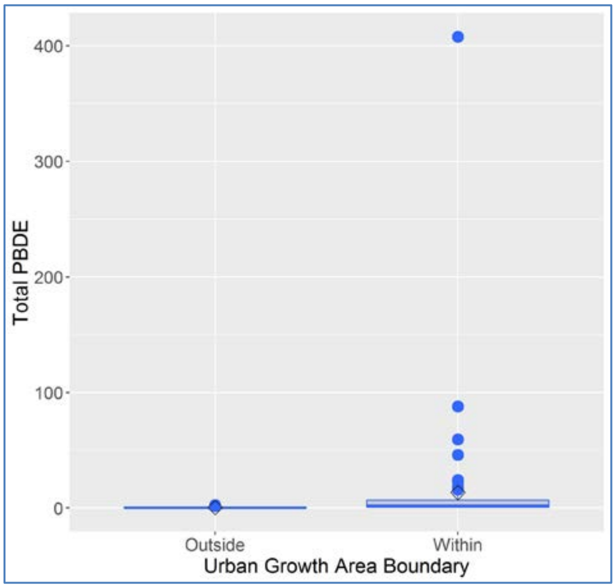


Figure 53. Box plot (left) and cumulative distribution function (CDF) plot (right) for sediment total PBDE concentrations for sites sampled outside and within Urban Growth Areas.

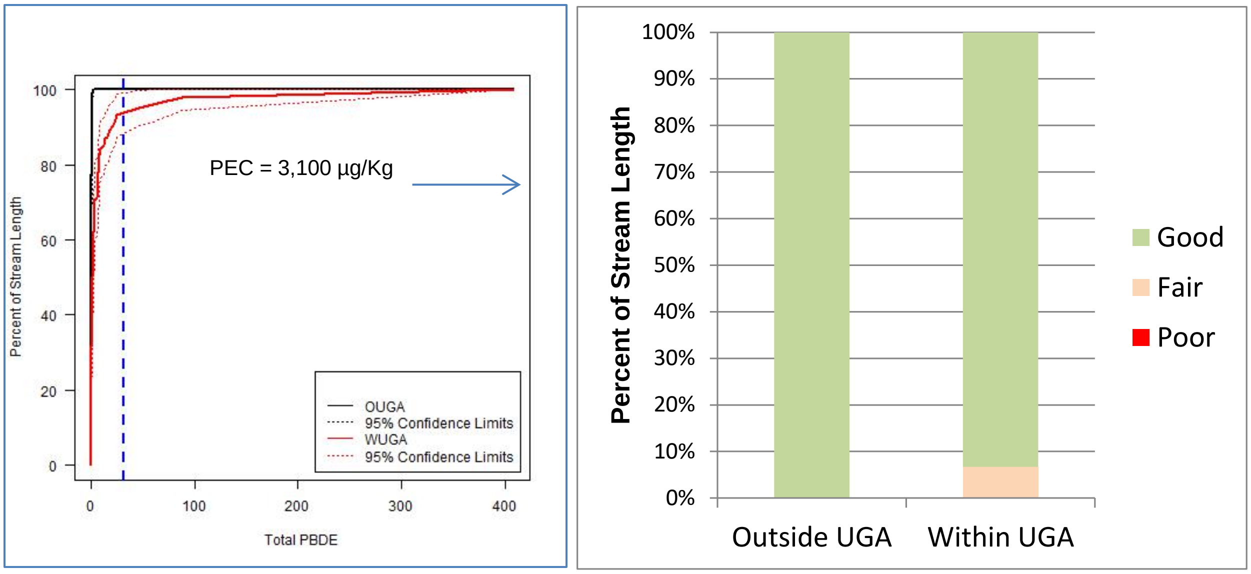


Figure 54. Categorical analysis bar plot for sediment total PBDE concentrations for sites sampled outside and within Urban Growth Areas.

3.3.7.3 Dichlobenil

Box plots and CDF plots of dichlobenil concentrations observed in sediments within and outside UGAs indicate that sediment dichlobenil concentrations were relatively similar within relative to outside of UGAs, although the highest concentration was found at an outside UGA site (Figure 55). The differences between the CDFs within and outside UGAs were not statistically significant (Table 19 and the right panel in Figure 55Figure 51). No ecologically relevant thresholds were identified for dichlobenil, so it was not possible to determine the portion of stream length within and outside UGAs in good, fair, or poor condition.

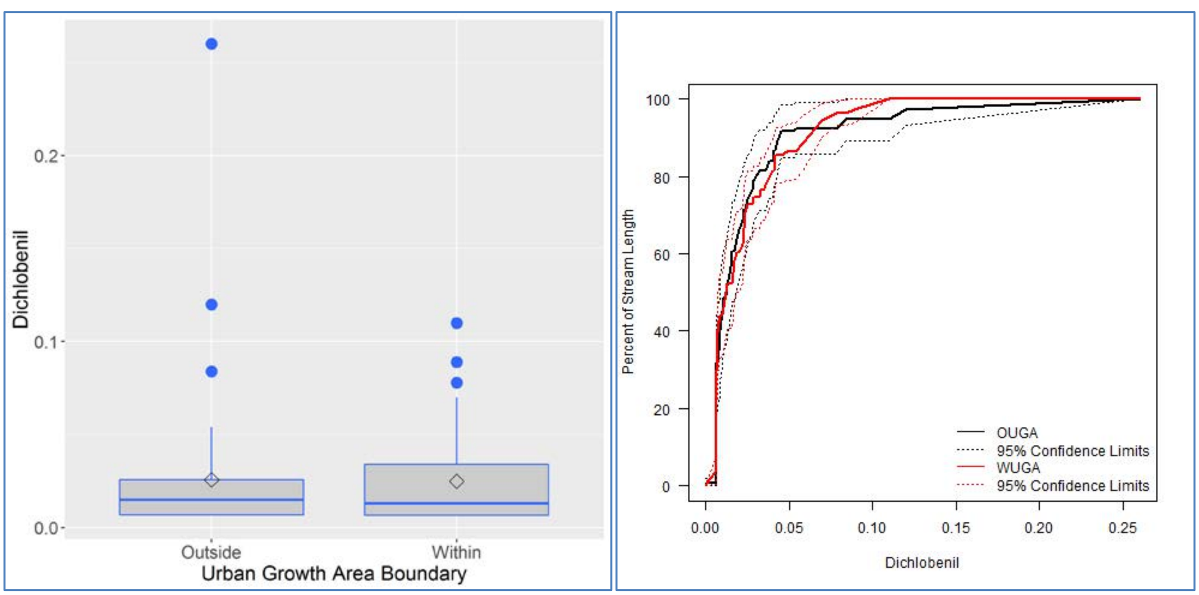


Figure 55. Sediment dichlobenil box plot (left) for stream sites and cumulative distribution function (CDF) plot (right) for streams outside and within Urban Growth Areas.

3.3.8 Sediment Quality: Site-specific Comparison to Sediment Management Standards

SAM PLES sediment samples were sieved prior to analysis, while the state Sediment Management Standards were developed for comparison to (and based on) whole sediment analyses. Therefore, the SAM PLES study results provide a conservative evaluation of stream sediment quality (i.e., concentrations will generally be higher in sieved vs whole sediment).

Most measured sediment contaminant concentrations did not typically exceed sediment quality standards, indicating that the risk of community level effects on stream benthic macroinvertebrates was generally low throughout the study area. This is consistent with the findings based on the use of ecologically-relevant thresholds in Section 3.3.7.

Sediment chromium and cadmium concentrations were the only contaminants that exceeded the state Sediment Screening Levels (Table 27 and Figure 56). Exceedance of the Sediment Screening Level indicates a high potential for adverse effects to benthic invertebrates and can trigger a remediation effort to minimize toxic effects. These exceedances occurred in both strata; within and outside UGAs (Table 27).

A number of other samples exceeded the lower Sediment Cleanup Objective for arsenic, copper, chromium, silver, and total PAH, total PCB, and bis(2-ethylhexyl)phthalate (Table 28 and Figure 57). Exceedances of the Sediment Cleanup Objective are of moderate concern as these levels are "no-effects concentrations" that provide long-term sediment quality cleanup goals.

Table 27. Sites within and outside urban growth areas where measured sediment contaminant concentrations exceeded Sediment Screening Levels.

UGA location	Concentration (mg/Kg)	Location Name
Cadmium (SCO= 2.1 mg/L; SSL = 5.4 mg/L)		
OUGA	7.01	COULTER CREEK TRIBUTARY AT COULTER CR RD
Chromium (SCO = 72 mg/kg; SSL = 88 mg/kg)		
WUGA	97.4	UNION RIVER TRIBUTARY
WUGA	98.4	GOLDSBOROUGH CREEK
OUGA	101	SKOOKUM CREEK TRIBUTARY
WUGA	140	PORTAGE CREEK TRIBUTARY
OUGA	280	COULTER CREEK TRIBUTARY AT COULTER CR RD

SSL = Sediment Screening Level; SCO = Sediment Cleanup Objective

Table 28. Sites within and outside urban growth areas where measured sediment contaminant concentrations exceeded the Sediment Cleanup Objectives but were less than the Sediment Screening Level.

UGA location	Concentration (mg/Kg)	Location Name
Arsenic (SCO = 14 mg/L; SSL = 120 mg/L)		
WUGA	99.8	QUILCEDA CREEK, MIDDLE FORK TRIBUTARY
WUGA	45.1	WEST HYLEBOS CREEK
WUGA	40.3	WEST HYLEBOS CREEK
WUGA	36.6	NORTH CREEK
WUGA	36.5	WEST HYLEBOS CREEK
WUGA	35.9	COAL CREEK
OUGA	33.9	SULLIVAN GULCH CREEK
WUGA	30.2	PORTAGE CREEK TRIBUTARY
WUGA	29.4	WILLOWS CREEK
WUGA	29.3	JOHNSON CREEK
WUGA	28.2	BOEING CREEK
OUGA	27.4	PILCHUCK RIVER TRIBUTARY
WUGA	26.5	WEST HYLEBOS CREEK
WUGA	26.2	DUMAS BAY TRIBUTARY
WUGA	26	JOHNSON CREEK
WUGA	22.2	JAPANESE GULCH
OUGA	20.6	JORDAN CREEK
WUGA	19.3	SCRIBER CREEK
WUGA	19.1	MAY CREEK
WUGA	17.8	WAPATO CREEK
OUGA	16.5	STOSSEL CREEK
WUGA	16	SWAMP CREEK
WUGA	15.9	SWAMP CREEK
WUGA	15.4	SCRIBER CREEK
WUGA	15.1	PETERS CREEK
OUGA	14.8	AUSTIN CREEK
WUGA	14.7	WILLOWS CREEK TRIBUTARY
WUGA	14.1	KIMBALL CREEK
Copper (SCO = 400 mg/kg; SSL = 1,200 mg/kg)		
WUGA	411	SWAMP CREEK
Chromium (SCO = 72 mg/kg; SSL = 88 mg/kg)		
OUGA	86.2	THOMAS CREEK
OUGA	84.3	JORDAN CREEK
WUGA	82.7	QUILCEDA CREEK, MIDDLE FORK TRIBUTARY
OUGA	80.6	MARCH CREEK
WUGA	75.5	CROUCH CREEK TRIBUTARY
WUGA	75.4	SWAMP CREEK
Silver (SCO = 0.57 mg/kg; SSL = 1.7 mg/kg)		

UGA location	Concentration (mg/Kg)	Location Name
WUGA	1.46	NORMA CREEK TRIBUTARY
WUGA	1.32	CLOVER CREEK
WUGA	0.884	LONGFELLOW CREEK
Bis(2-ethylhexyl)phthalate (SCO = 500 mg/kg; SSL = 2,200 mg/kg)		
WUGA	640	CLOVER CREEK
WUGA	582	POVERTY BAY TRIBUTARY
Total PAH (SCO = 17,000 mg/kg; SSL = 30,000 mg/kg)		
WUGA	27,544	JOHNSON CREEK
Total PCB (SCO = 110 mg/kg; SSL = 2,500 mg/kg)		
WUGA	378.8	CLOVER CREEK

SSL = Sediment Screening Level; SCO = Sediment Cleanup Objective

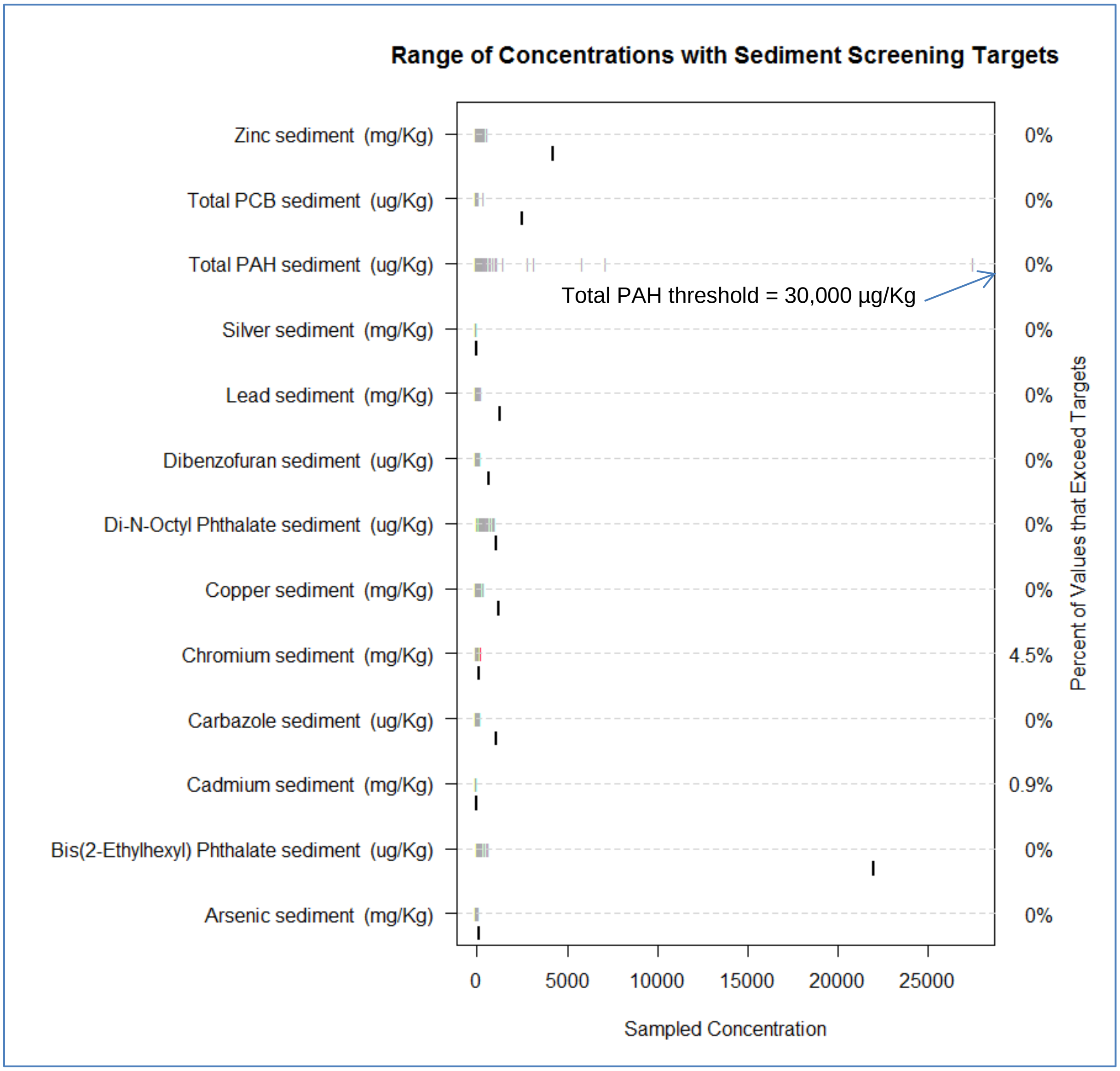


Figure 56. Range of concentrations compared with sediment quality screening levels for the protection of aquatic life.

Note: Thick dark vertical bars indicate relevant sediment quality screening level for the protection of aquatic life. Light gray vertical bars indicate observed sediment concentrations below the screening level and red vertical bars indicate observed concentrations that exceed the screening level.

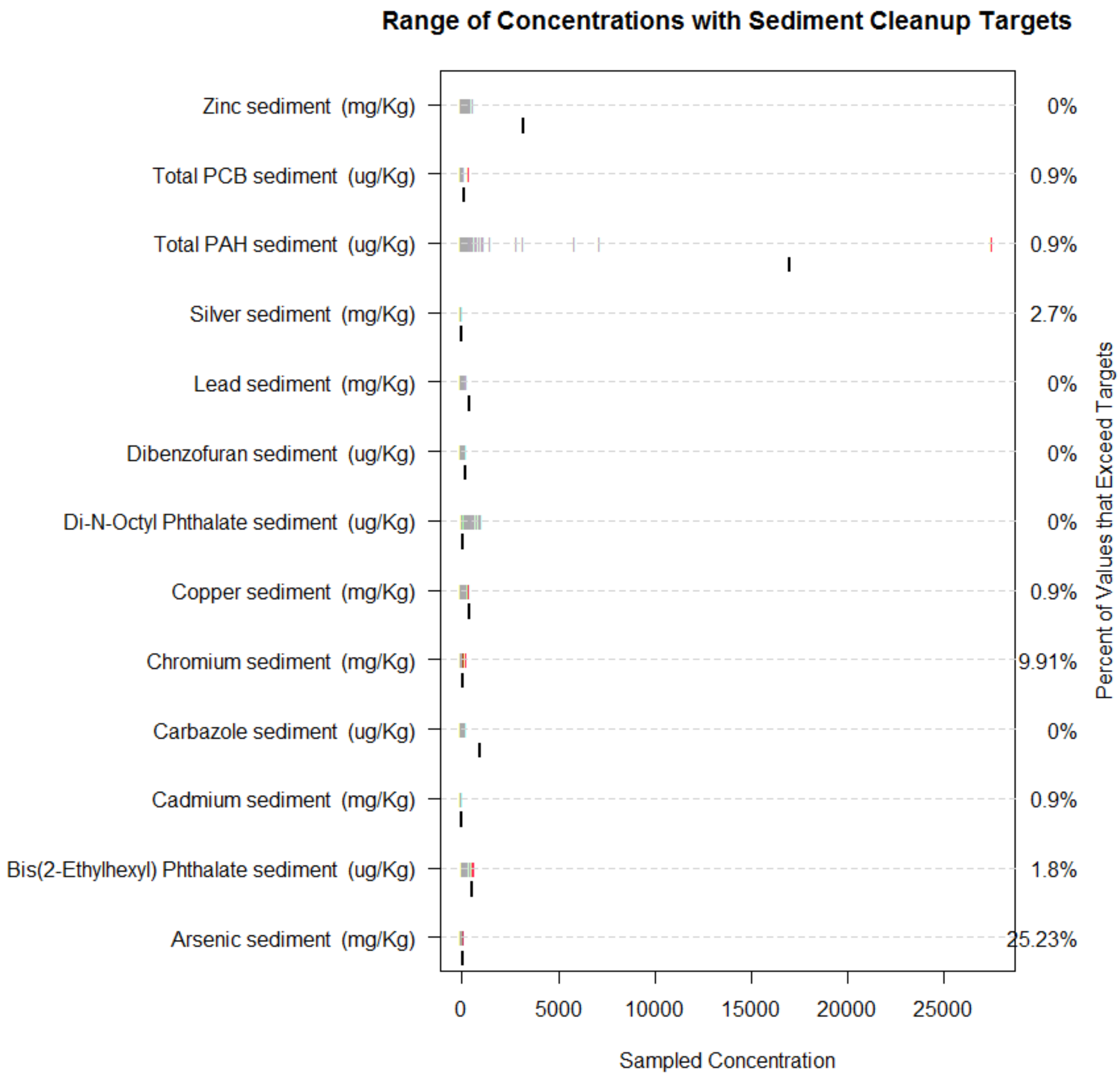


Figure 57. Range of concentrations compared with sediment quality cleanup levels for the protection of aquatic life.

Note: Thick dark vertical bars indicate relevant sediment quality cleanup level. Light gray vertical bars indicate observed sediment concentrations below the cleanup level and red vertical bars indicate observed concentrations that exceed the cleanup level.

3.3.9 All Other Sediment Quality Parameters

Other sediment quality parameters that were measured, but not discussed in detail above, include silver, individual PAH compounds, phthalates, common roadside use pesticides, sieved sediment total organic carbon content (TOC), and sieved sediment grain size. The frequency of detection of silver and the organic chemistry parameters is summarized in Appendix A Table A2 and summary statistics for chemistry results from sites within and outside UGAs is provided in Appendix B Table B6. The summary of CDF results are provided in Appendix C6.

Sediment silver concentrations were quantified in 57 percent of the samples collected (Appendix A Table A2). The detection frequency within UGAs was higher (67 percent) relative to sites outside UGAs (44 percent). The maximum silver concentration (1.46 mg/Kg) was measured at a site within a UGA. This concentration did not exceed the Sediment Screening Level of 1.7 mg/Kg.

With the exception of retene, individual PAHs were not detected in more than 50 percent of the sites sampled (Appendix A Table A2). Although individual PAHs (except retene) were infrequently detected, several PAHs were detected more frequently at sites within UGAs (Appendix B Table B6). The exception, retene, was detected frequently within and outside UGAs (Appendix B Table B6) and a statistically significant difference in retene concentrations was found within and outside UGAs (Appendix C6 Table C6-3). The highest retene concentrations were found outside UGAs, while the median concentration within UGAs was highest (Appendix B Table B6). Retene is typically used as a tracer for conifer combustion and is not included in the total PAH sum (Stogiannidis and Laane, 2015). Sources of retene include forest fires and wood burning in fireplaces and stoves.

Of the six phthalates analyzed, bis(2-ethylhexyl)phthalate was detected most frequency (46 percent), while the remaining compounds were detected at 7 percent or fewer of the sites (Appendix A Table A2). Phthalates were also generally detected more frequently within UGAs and the highest concentration of bis(2-ethylhexyl)phthalate was measured at a within UGA site (see Table 28).

As noted above, with the exception of dichlobenil, the common roadside use pesticides were infrequently detected (Appendix A Table A2). Diuron was the next most frequently detected pesticide (2 percent frequency of detection overall) and detected at only within UGA sites (3 percent detection frequency). The remaining pesticides (2,4-D, carbaryl, chlorpyrifos, and triclopyr) were never detected.

Sediment TOC was measured in both sieved sediment fractions (sediment passing 0.063 and 2.0 mm sieves). TOC was reliably quantified in all samples, although results were incomplete for the 0.063 mm fraction samples (Appendix A Table A2).[17] Although a statistically significant difference in CDFs within and outside UGAs was found for the 2.0 mm sediment fraction TOC concentration (Appendix C6 Table C6-3), the mean and maximum concentrations were similar (mean/max: 3.0/19.5 percent outside and 3.9/23.1 percent inside UGAs) (Appendix B Table B6). TOC data like that collected in this study is typically used to normalize (reduce the variance in) sediment contaminant data either through direct normalization or by including TOC as a model variable to help explain contaminant patterns in relationship to other variables (e.g., urbanization) (e.g., Nowell et al., 2013; Moran et al., 2017). A more in-depth study of the 2015 sediment contaminant

[17] Pierce County only reported TOC in the 2.0 mm sieve fraction. City of Redmond did not indicate which sediment fraction was analyzed. For this report, it was assumed that Redmond reported results for the 2.0 mm sediment fraction.

data might investigate how incorporation of sediment TOC data might affect the results of the sediment status assessment.

Grain size was measured on the 2.0 mm sediment fraction (essentially very coarse sand and finer) and reported in phi size classes from sizes less than -2.0 (gravel and larger) to sizes between 9 and 10 (very fine clay particles). Similar to sediment TOC data, sediment grain size data can be used to evaluate (and potentially normalize) the effect of grain size on sediment contaminant concentrations (Horowitz, 1991). There do appear to be some statistically significant differences in grain size distribution within and outside UGAs, particularly in the coarse sand (phi 0-1) to very fine sand fractions (phi 3-4) (Appendix B Table B6).[18]

It is recommended that the collection and analysis of sieved sediment samples continue as part of this program, although perhaps focusing on collection and analysis of the <2.0 mm size fraction to ensure a complete and consistent data set. Sediment contaminant concentrations (trace metals in particular) are strongly affected by the sediment particle size distribution (Rickert et al., 1977; Horowitz, 1991) so sieved sediment analysis are conducted to improve the likelihood of detection and to enhance the comparability of data among sites. Note that the USGS recommends sieving sediments through a 0.063 mm sieve for trace metal analysis (Shelton and Capel, 1994) and Rickert et al. (1977) argued that sieving to 0.020 mm was necessary to ensure comparability. An additional study of the 2015 sediment contaminant data might investigate how sediment grain size distribution is related to sediment metal or organic contaminant concentrations.

3.3.10 Stream Habitat: Comparisons to reference conditions

Over 260 habitat metrics were calculated from the stream reach field surveys conducted for this study. In this section, we report on only a few traditional habitat assessment categories:

- Riparian Canopy Closure: Stream center densiometer measurement
- Wood: Wood volume normalized to a 100 m reach length
- Pools: Residual pool area
- Substrate: Median particle diameter
- Bed stability: Logarithm of Relative Bed Stability

Summary statistics for all of the within and outside UGA site habitat data is provided in Appendix B Table B7. The summary of CDF analyses for the habitat data are provided in Appendix C7.

There are no state standards for these habitat measures, so the status assessment relied on comparisons to reference conditions for threshold comparisons.

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[18] Differences in grain size distribution within and outside UGAs is also apparent from the whole sediment grain size data evaluated in the SAM stream habitat data (see Section 3.3.10.4).

3.3.10.1 Riparian Canopy Closure

Lower values of riparian canopy closure indicate less shade producing riparian canopy cover. Shade is considered a positive habitat characteristic that minimizes stream heating and is an indication of the amount of leaf litter that provides a food base to aquatic organisms (Roberts and Bilby, 2009). Box plots and CDF plots of the mean values observed in the center of streams sampled within and outside UGAs indicate that values were very similar within and outside of UGAs (Figure 58 and Figure 59). The differences between the CDFs within and outside UGAs were not statistically significant (Table 19 and left panel of Figure 59). Based on least-disturbed reference condition, 20 percent of the stream length within and outside UGAs was in poor condition with respect to these values (Figure 59). A lesser proportion of stream length outside UGAs was in good condition (61 percent) relative to streams within UGAs (74 percent).

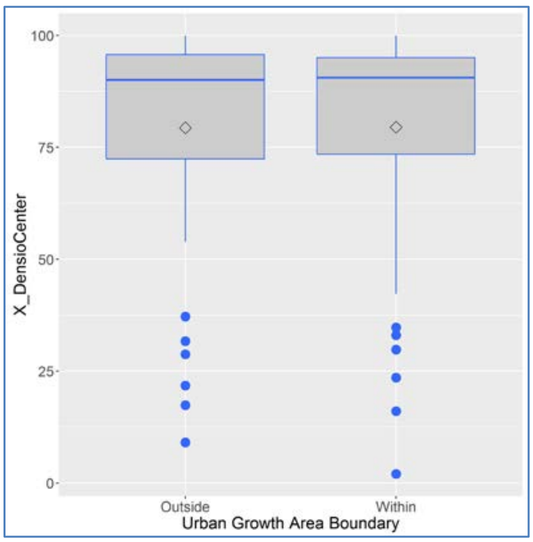


Figure 58. Stream center densiometer values (X_DensioCenter) box plot for stream sites outside and within Urban Growth Areas.

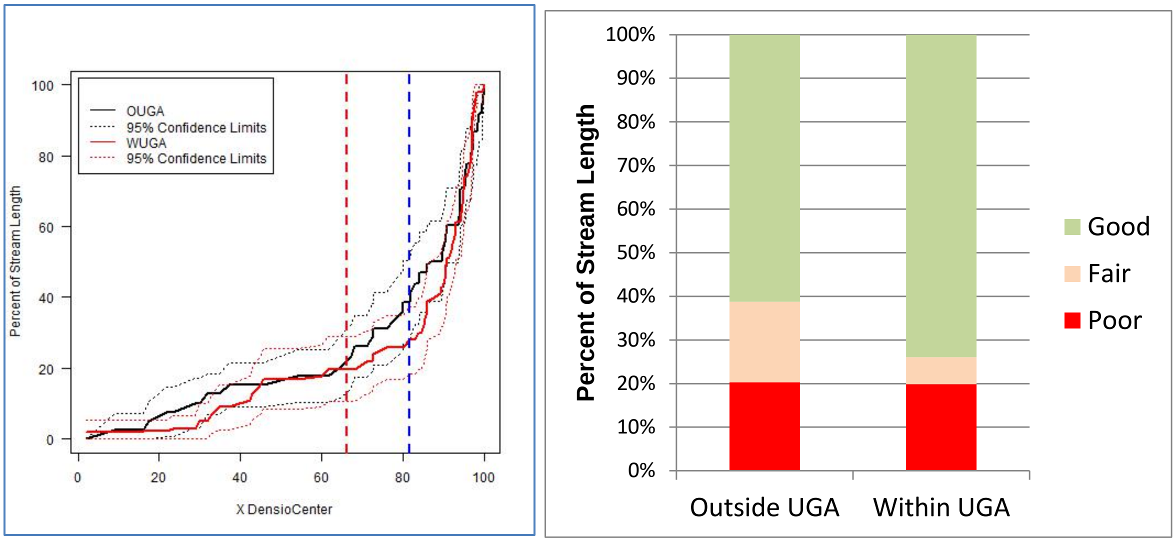


Figure 59. Stream center densiometer values (X_DensioCenter) cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.10.2 Wood Volume

Lower values indicate less stream wood debris and a poorer condition. Woody debris adds channel complexity, causes pool formation, and provides channel roughness that dissipates stream energy (Booth et al., 1997; Montgomery and Piégay, 2003). Box plots and CDF plots of the volume of large woody debris normalized to a 100 m reach length sampled within and outside UGAs indicate that large woody debris volume was slightly greater outside UGAs (Figure 60 and Figure 61). The differences between the CDFs within and outside UGAs were not statistically significant (Table 19 and left panel of Figure 61). Based on least-disturbed reference condition, 61 percent of the stream length within UGAs was in poor condition with respect to large woody debris volume, while 44 percent of the stream length outside UGAs was in poor condition (Figure 61). A greater proportion of stream length outside UGAs was in good condition (39 percent) relative to streams within UGAs (31 percent).

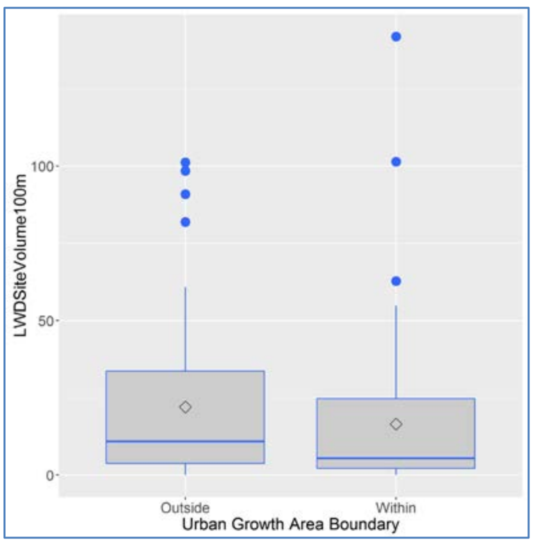


Figure 60. Volume of wood per 100 m reach length (LWDSiteVolume100 m) box plot for stream sites sampled outside and within Urban Growth Areas.

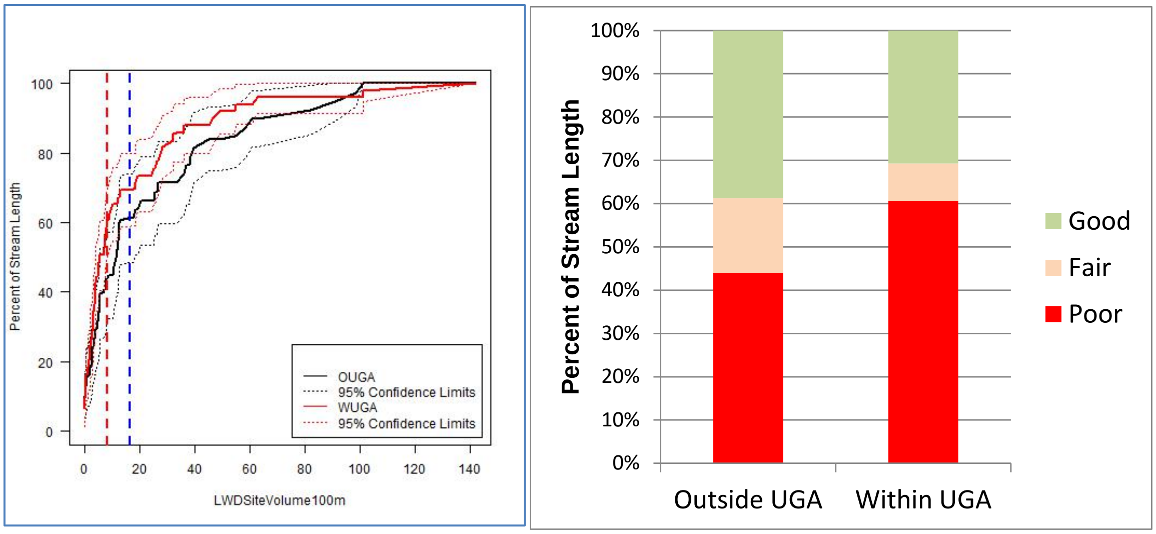


Figure 61. Volume of wood per 100 m reach length (LWDSiteVolume100m) cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.10.3 Residual Pool Area

Lower residual pool area values indicate less pool habitat. Pool habitat is an important rearing area for fish and is often related to the amount of stream woody debris (Bisson et al., 1987). Box plots and CDF plots of the residual pool area normalized to a 100 m reach length sampled within and outside UGAs indicate that residual pool area was very similar within and outside of UGAs (Figure 62 and Figure 63). The differences between the CDFs within and outside UGAs were not statistically significant (Table 19 and left panel of Figure 63). Based on least-disturbed reference condition, 29 percent of the stream length within UGAs was in poor condition with respect to residual pool area, while 16 percent of the stream length outside UGAs was in poor condition (Figure 63). A lesser proportion of stream length outside UGAs was in good condition (44 percent) relative to streams within UGAs (53 percent).

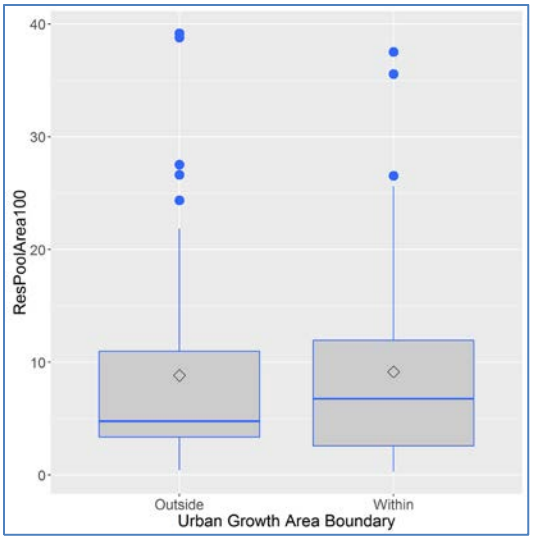


Figure 62. Residual pool area per 100 m reach length (ResPoolArea100) box plot for stream sites sampled outside and within Urban Growth Areas.

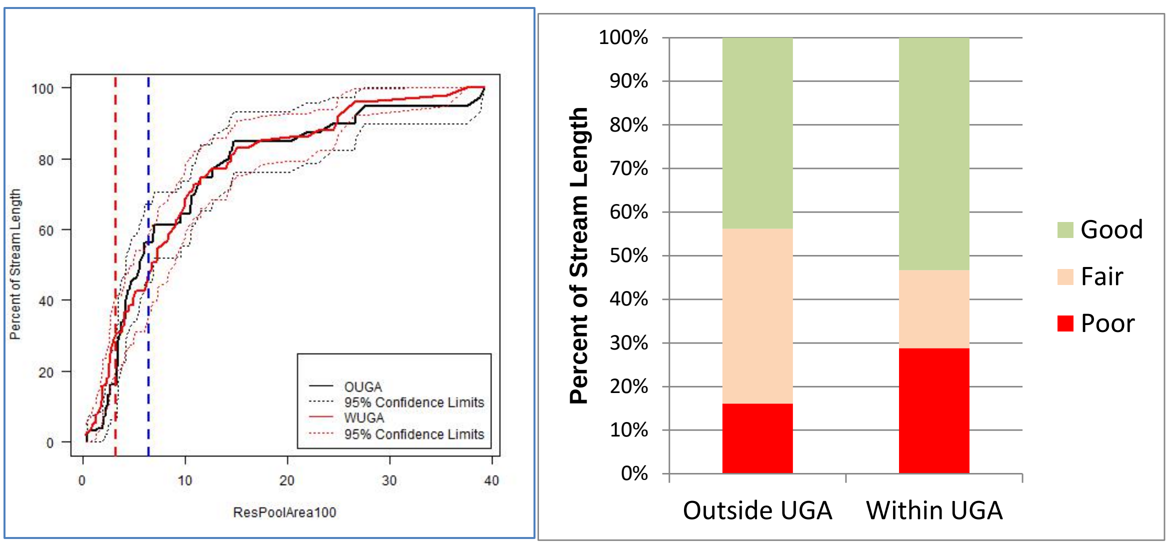


Figure 63. Residual pool area per 100 m reach length (ResPoolArea100) cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.10.4 Stream Substrate

Larger median diameter values define a coarser stream substrate, while lower values indicate finer stream substrate. Moderately coarse stream substrates (e.g., gravel, cobble, or small boulders) are considered to be a positive habitat attribute for anadromous fish spawning and healthy macroinvertebrate communities (Bryce et al., 2010). Box plots and CDF plots of the median diameter of stream substrate material within and outside UGAs indicate that median stream substrate diameter was typically greater outside UGAs (Figure 64 and Figure 65). The differences between the CDFs within and outside UGAs were statistically significant (Table 19 and left panel of Figure 65). Based on least-disturbed reference condition, 25 percent of the stream length within UGAs was in poor condition with respect to median stream substrate diameter, while 20 percent of the stream length outside UGAs was in poor condition (Figure 65). A much greater proportion of stream length outside UGAs was in good condition (44 percent) relative to streams within UGAs (13 percent).

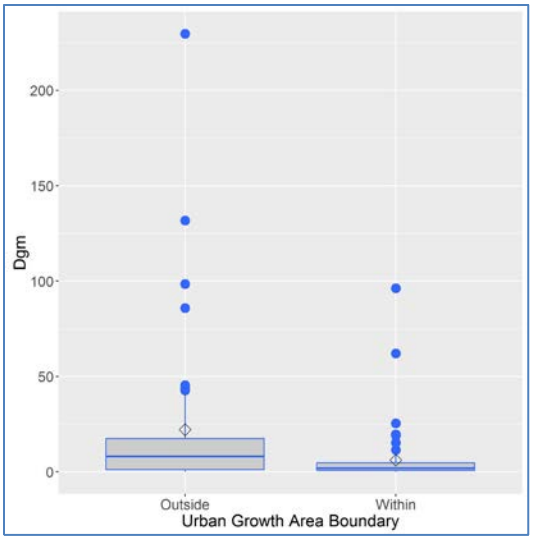


Figure 64. Median stream substrate particle diameter (Dgm) box plot for stream sites sampled outside and within Urban Growth Areas.

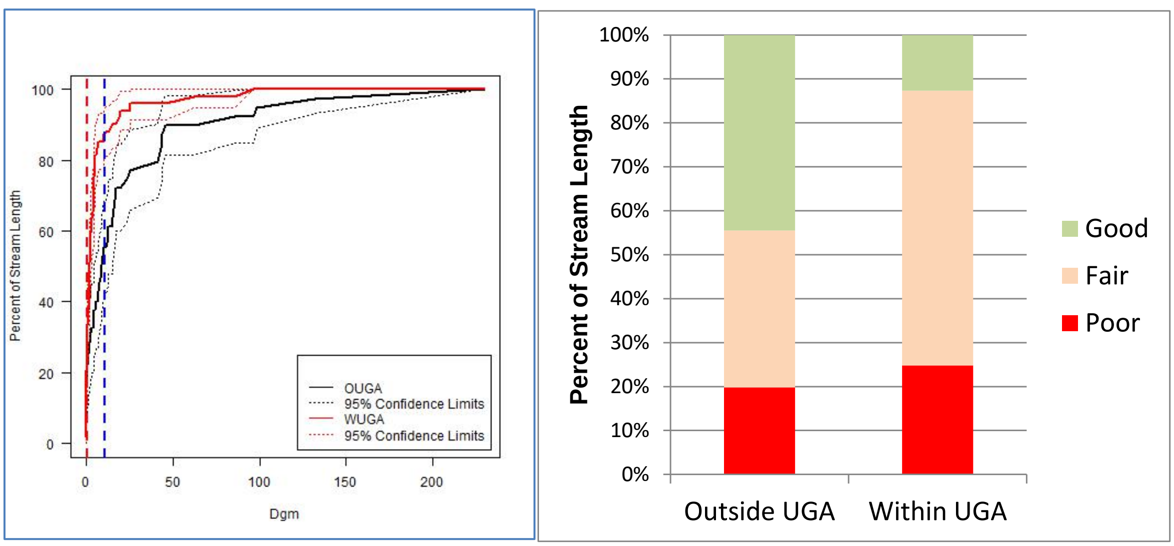


Figure 65. Median stream substrate particle diameter (Dgm) cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

Low gradient stream reaches in watersheds dominated by easily erodible fine sediment may result in naturally occurring fine sediment conditions in some streams. The distribution of the percent slope of small streams outside and within UGAs was very similar (although the CDFs were found to be significantly different; Appendix C8), so the difference in median particle diameter outside and within UGAs does not appear to be due to differences in stream gradient (Figure 66). Further investigation of the potential effect of reach slope and other factors such as potential sediment supply and channel transport capacity might provide additional information to explain the differences in sediment particle size distribution within and outside UGAs.

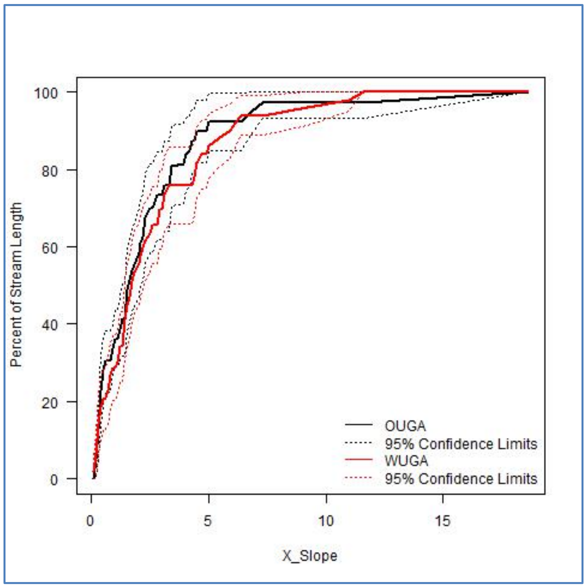


Figure 66. Cumulative distribution function (CDF) plot for percent slope for streams sampled outside and within Urban Growth Areas.

3.3.10.5 Bed Stability

Higher log relative bed stability values indicate greater bed stability (based on a combination of stream gradient and substrate median particle diameter). Greater bed stability is associated with better benthic invertebrate biological condition (Kaufmann et al., 2009). Bed stability appeared to be greater outside relative to inside UGAs (Figure 67 and Figure 68). The difference in bed stability within and outside UGAs was found to be statistically significant (Table 19 and left panel of Figure 68). Based on least-disturbed reference condition, 36 percent of the stream length within UGAs was in poor condition with respect to log relative bed stability, while 25 percent of the stream length outside UGAs was in poor condition (Figure 68). A much greater proportion of stream length outside UGAs was in good condition (64 percent) relative to streams within UGAs (26 percent).

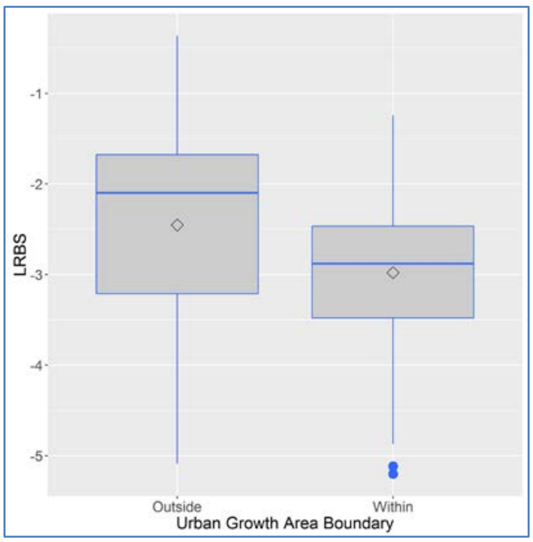


Figure 67. Logarithm of relative bed stability (LRBS) box plot for stream sites sampled outside and within Urban Growth Areas.

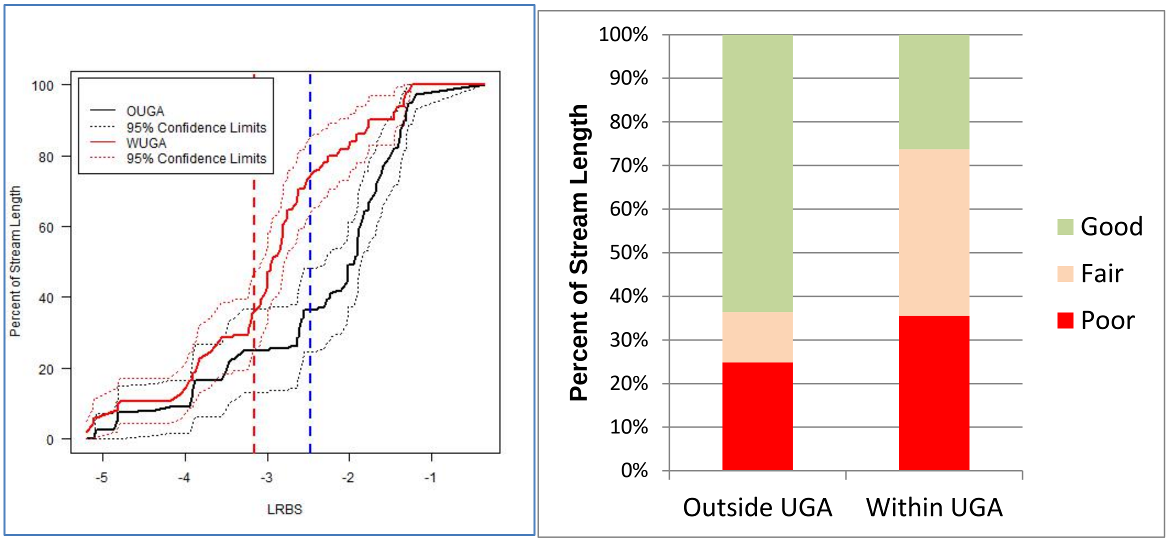


Figure 68. Logarithm of relative bed stability (LRBS) cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.11 All Other Stream Habitat Metrics

Habitat metrics that were measured, but not discussed in detail above, include measures describing bank quality and stability, channel dimensions, fish cover, habitat dimensions, large woody debris, riparian cover, disturbance, and vegetation structure, stream sinuosity, and bottom substrate.[19] Summary statistics for sites within and outside UGAs is provided in Appendix B Table B7. The summary of CDF results are provided in Appendix C7.

Of the 261 habitat metrics in Appendix C7 Table C7-3, the CDFs of 36 habitat metrics (out of the 256 not discussed above) were determined to be significantly different within and outside UGAs. Several of these were metrics for substrate particle size distribution including percent fines and percent sand+fines. Other metrics that were found to be significantly different included proximity weighted presence of all combined human influences on the riparian zone and stream embeddedness.

[19] See Janisch (2013) for a detailed description of habitat metrics measured as part of the Watershed Health monitoring protocols.

3.3.12 Landscape Data: Comparisons to reference conditions

Over 100 landscape metrics were calculated from available GIS data layers (Sheibley et al., 2017a). The landscape data were used to aid in an evaluation of the natural physical and human stressors that potentially affect observed stream biological conditions. Here we present the status evaluation of four landscape metrics that appeared to be associated with low B-IBI scores and/or with high TDI scores in the relative risk/attributable risk analysis presented in Section 3.4.2 below. These landscape metrics are:

- **Watershed Percent Urban Development:** The percent of the watershed upstream of the sampling point that is classified as low, medium, or high intensity development in the 2011 National Land Cover Database.
- **Watershed Percent Canopy Cover:** The percent of the watershed upstream of the sampling point that is covered by tree canopy as determined from the 2011 National Land Cover Database.
- **Riparian Percent Canopy Cover:** The percent of the riparian buffer upstream of the sampling point that is covered by tree canopy as determined from the 2011 National Land Cover Database.
- **Areal Nitrogen Loading Rate:** The annual areal loading rate of nitrogen from the watershed upstream of the sampling point, which the sum of estimated loading from wastewater, farm fertilizer, non-farm fertilizer, and manure.

3.3.12.1 Watershed Percent Urban Development

Consistent with the expectation that sites outside UGAs are less developed, the watershed percent urban development is typically much lower outside vs inside UGAs (Figure 69 and Figure 70). The difference is statistically significant (Table 19 and left panel of Figure 70). Based on reference site watershed percent development, 86 percent of stream length inside UGAs and 17 percent of stream length outside UGAs would be considered in poor condition (Figure 71). A much greater percentage of stream length outside UGAs would be considered in good condition (72 percent) compared to sites within UGAs (10 percent).

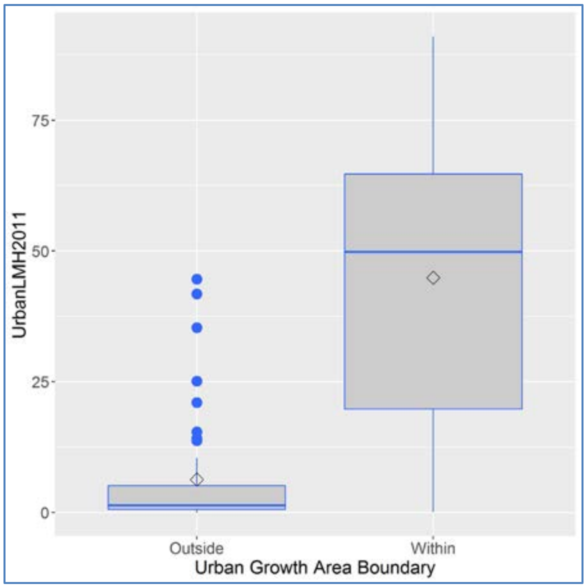
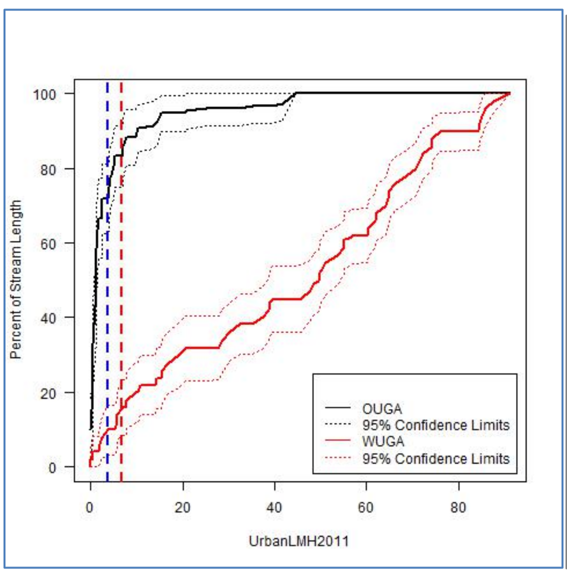


Figure 69. Watershed percent urban development (low, medium, and high intensity development) box plot for stream sites sampled outside and within Urban Growth Areas.



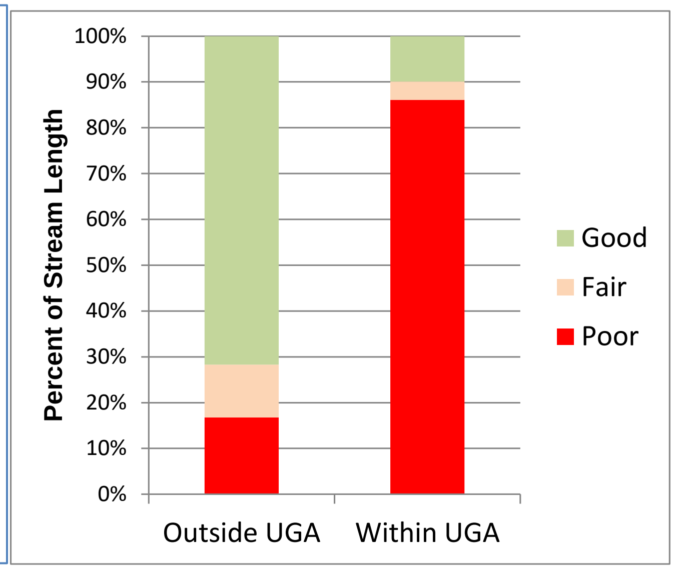


Figure 70. Watershed percent urban development (low, medium, and high intensity development) cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.12.2 Watershed Canopy Cover

Box plots and CDF plots indicate that watershed percent canopy cover is typically much lower inside vs outside UGAs (Figure 71 and Figure 72). The difference is statistically significant (Table 19 and left panel of Figure 72). Based on reference site watershed percent canopy cover, 94 percent of stream length inside UGAs and 40 percent of stream length outside UGAs would be considered in poor condition (Figure 72). A greater percentage of stream length outside UGAs would be considered in good condition (46 percent) compared to sites within UGAs (4 percent).

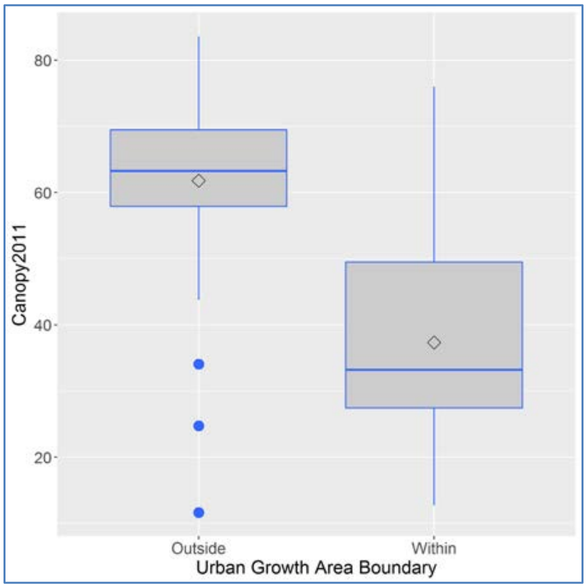


Figure 71. Watershed percent canopy cover box plot for stream sites sampled outside and within Urban Growth Areas.

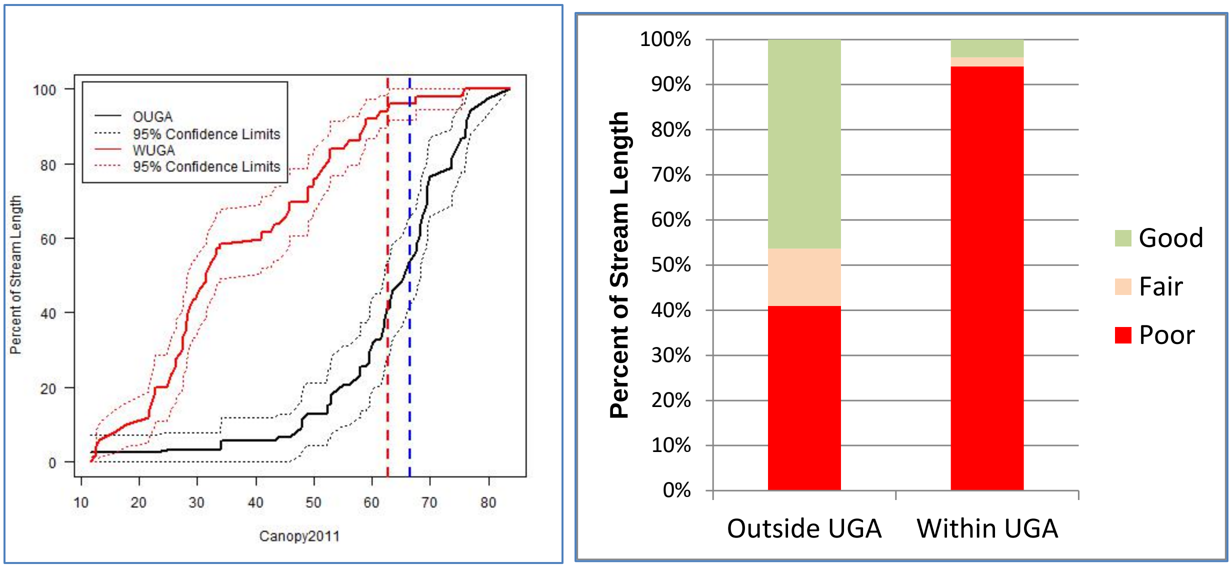


Figure 72. Watershed percent canopy cover cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.12.3 Riparian Canopy Cover

Box plots and CDF plots of riparian percent canopy cover within and outside UGAs indicate that riparian canopy cover is somewhat higher outside relative to inside of UGAs (Figure 73 and Figure 74). The difference was statistically significant (Table 19 and left panel of Figure 74). Based on reference site canopy cover, a greater percentage of stream length within UGAs is in poor condition (56 percent) relative to streams outside UGAs (29 percent) (Figure 74). A greater percentage of stream length outside UGAs was considered in good condition (57 percent) compared to sites within UGAs (21 percent).

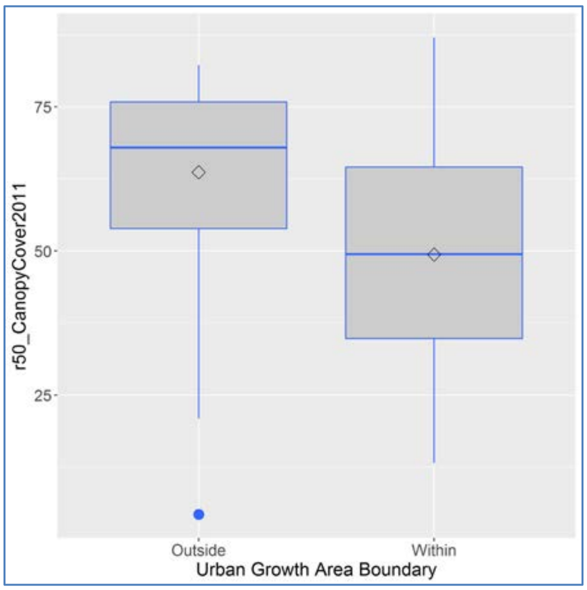
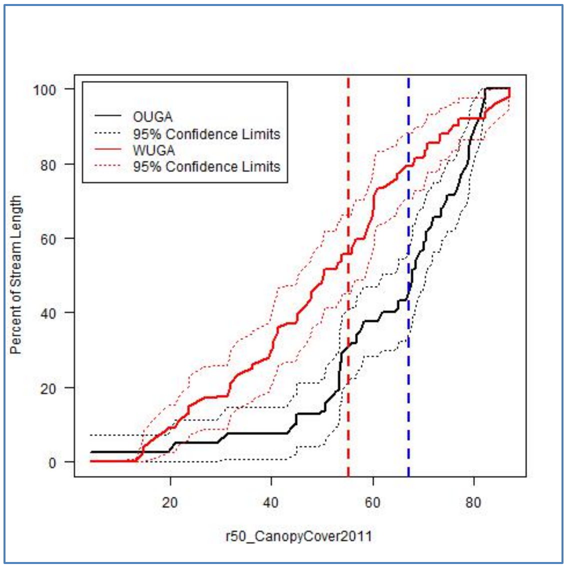


Figure 73. Riparian percent canopy cover box plot for stream sites sampled outside and within Urban Growth Areas.



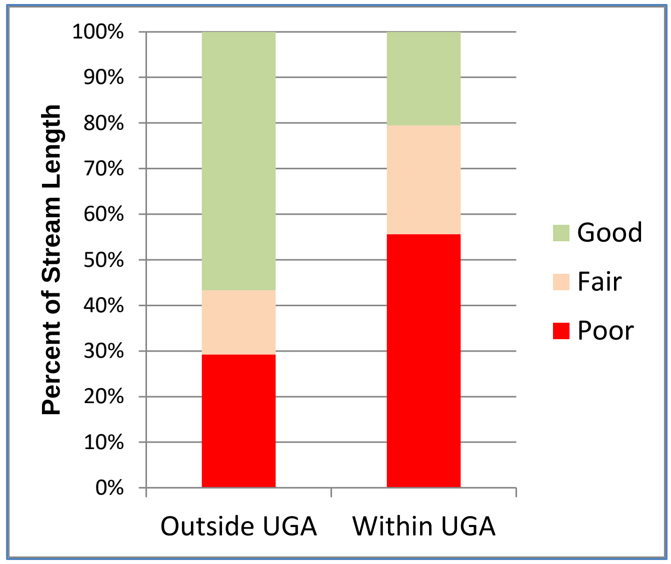


Figure 74. Riparian percent canopy cover cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.12.4 Areal Nitrogen Loading Rate

Box plots and CDF plots of the areal nitrogen loading rate within and outside UGAs indicate that nitrogen loading rate is somewhat higher inside relative to outside of UGAs (Figure 75 and Figure 76). The difference was statistically significant (Table 19 and left panel of Figure 76). Based on reference site nitrogen loading estimates, a greater percentage of stream length within UGAs is in poor condition (56 percent) relative to streams outside UGAs (16 percent) (Figure 76). A greater percentage of stream length outside UGAs was considered in good condition (56 percent) compared to sites within UGAs (12 percent). Although not shown here, the results of comparisons and status analysis for areal phosphorus loading rates are very similar to the results for nitrogen loading.

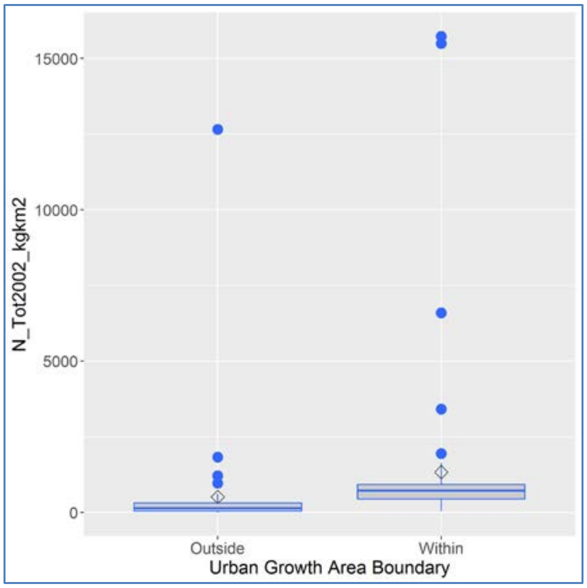
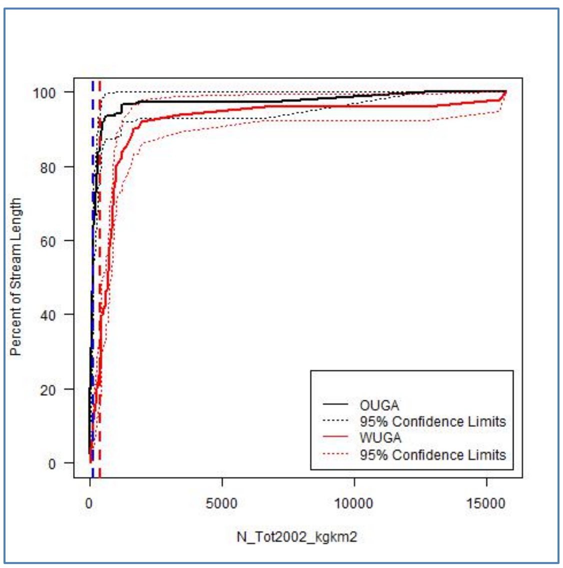


Figure 75. Areal nitrogen loading rate box plot for stream sites sampled outside and within Urban Growth Areas.



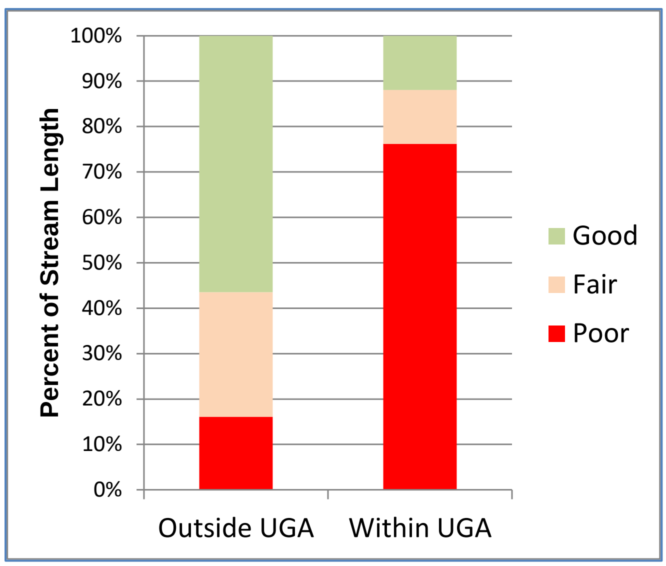


Figure 76. Areal nitrogen loading rate cumulative distribution function (CDF) plot (left) and categorical analysis bar plot (right) for streams outside and within Urban Growth Areas.

3.3.13 All Other Landscape Data

Other landscape data that were measured, but not discussed in detail above, include a variety of land cover data at the riparian and watershed scale as well as extent of surficial geological classes, precipitation statistics, additional areal nutrient delivery rates, population, road, and housing density, and watershed permeability (Sheibley et al, 2017a). Summary statistics for sites within and outside UGAs is provided in Appendix B Table B8. The summary of CDF results are provided in Appendix C8.

Of the 117 landscape metrics in Appendix C8 Table C8-3 not described above, the CDFs of 56 habitat metrics (out of 113) were determined to be significantly different within and outside UGAs. Although not inclusive, significant differences were found for site and mean watershed elevation (areas outside UGAs tend to be at higher elevations), monthly and annual watershed mean precipitation totals (more precipitation tends to fall at higher elevations), forest cover (higher cover outside UGAs), house and population density (higher within UGAs), impervious cover (higher within UGAs), areal phosphorus loading (higher within UGAs), and mean watershed slope (greater outside UGAs),

3.4 Risk Assessment: Identifying Natural and Human Stressors

As noted in Section 2.7.3, to be considered a reliable biological response indicator, the indicator should respond to natural and human stressors, but be relatively insensitive to natural background variability that is largely beyond human control (Dorfmeier, 2014). Examples of factors that are largely not under human control, but could potentially be factors that influence biological responses include watershed area, mean watershed elevation, and surficial geology (Dorfmeier, 2014). Biological response metrics that are not sensitive to natural background variables (or have been modified to remove their influence) can then be used to understand, if, and to what degree, natural and anthropogenic stressors influence these biological response metrics.

Two distinct approaches were used to evaluate relationships of natural and human stressors to biological response endpoints. Boosted regression trees were used to evaluate the potential influence of natural background variables as well as identify potentially important natural and human stressors.[20] Relative risk/attributable risk analysis was used to evaluate the potential importance of natural and human stressors. Here, we selected two key biological response metrics for analysis, B-IBI and TDI.

3.4.1 Boosted Regression Trees

Boosted regression tree (BRT) models were developed to help discover the most important variables across a wide range of categories that explain variability in biological responses. We used BRTs to evaluate the biological response variables across all small streams in the Puget Lowlands.[21] BRT models do not "prove" or "disprove" cause-effect relationships. However, they do provide insights into the relative importance of the predictor variables

[20] The distinction between natural and human stressors is described in Section 2.7.3.

[21] Initially, models were tested using within and outside UGAs as a categorical variable, but this classification was not a relatively important variable in any model. This variable was not included in the analysis.

tested to explain variation in the target response variables; in this case B-IBI and Trophic Diatom Index scores.

3.4.1.1 BRT Model for B-IBI Scores

The overall B-IBI response model included 14 variables and had a cross-validation correlation of 0.704.[22] The relative importance of these variables is shown in Figure 77 with the largest bars indicating the highest relative importance in the model explaining the variation in B-IBI scores.

The most important factor in the model was considered a natural background landscape factor; average (1982-2014) precipitation in December. The high importance of mean December precipitation is likely related to other human stressors that are potentially being confounded in the model. That this is the case is suggested by the fact that the model predicts higher B-IBI scores at sites with higher basin mean December precipitation. Although it is not possible to exclude a cause and effect relationship between basin mean precipitation and B-IBI scores, the high inverse correlation between basin mean precipitation and measures of urbanization and impervious cover suggest that the model is confounding precipitation with other measures of human disturbance. Similarly, the importance of longitude, another natural background variable in the model, is also likely due to the strong east-west development gradient across the Puget Lowlands with low levels of development and high levels of forest cover on the eastern flanks of the Olympic Mountains on the western side of the Puget Sound basin and similar low levels of development and elevated forest cover on the western flanks of the Cascade Mountains on the eastern side of the basin. Overall, we do not believe there is no compelling evidence that B-IBI is causally influenced by any of the natural background factors evaluated in this study, consistent with a previous evaluation conducted by Dorfmeier (2014).

[22] The cross-validation (CV) correlation is a measure of model fit. A CV correlation near 1.0 would be a perfectly fit model and a CV correlation near zero would be a poorly fit model.

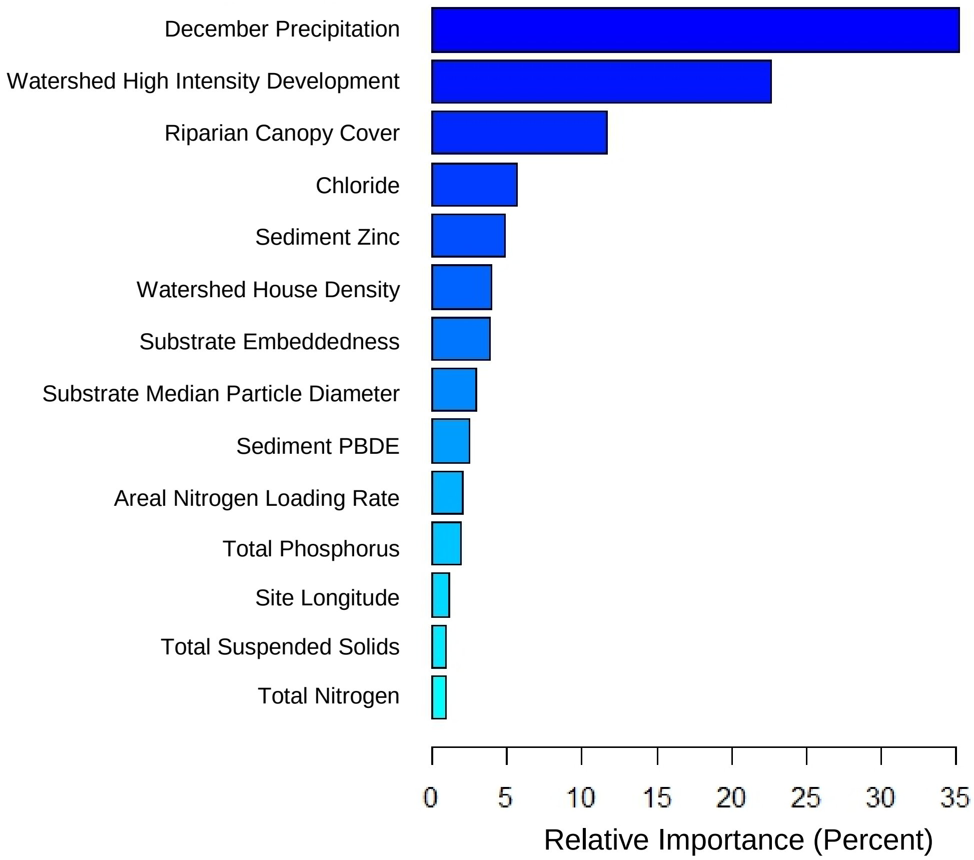


Figure 77. Relative importance of natural and human stressor variables in the final B-IBI boosted regression tree model.

All of the remaining important model variables are natural and human stressors. The next most important variable in the BRT model (Figure 77) was the percentage of high intensity development followed by riparian canopy cover in the watershed. The next seven most important stressors identified in the BRT model are considered stressors:

- Riparian canopy cover in the sample site buffer
- Chloride concentration in water (mean Aug-Oct)[23]
- Zinc concentration in sediment
- Housing density in the sampled site's watershed
- Stream substrate embeddedness
- Substrate median particle diameter

[23] Note that water concentrations were the average of samples collected in August-October. This averaging period was selected because although annual WQI scores were not substantially different within relative to outside UGAs, monthly WQI scores in August, September, and October had the largest relative differences with relatively lower WQI scores within UGAs (see Figure 23).

Although chloride and zinc occur naturally in the environment and stream embeddedness and substrate median particle diameter vary naturally among streams, these are considered stressors because research has shown that changes in these parameters are associated with human activity and urban development with adverse effects on benthic community structure (e.g., May et al., 1997; Morgan et al., 2007; Short et al., 2005). The remaining stressor variables in order of decreasing importance were sediment total PBDE concentration, watershed areal nitrogen loading rate, and mean August–October water concentrations of total phosphorus, total suspended solids and total nitrogen.

3.4.1.2 BRT Model for Trophic Diatom Index

The overall Trophic Diatom Index response model included 13 variables and had a cross-validation correlation of 0.761. The most important variable was total phosphorus concentration (mean Aug–Oct) (Figure 77). The remaining variables were relatively much less important. The next most important variable was the number of pieces of large woody debris per square meter followed by house density, stream total nitrogen concentration, stream chloride (mean Aug–Oct) concentration, sample site longitude, total nitrogen loading per square kilometer, rainfall erosivity, sediment copper concentration, sediment zinc concentration, watershed canopy cover, watershed mean (1982–2014) total annual precipitation, and stream total suspended solids concentration. The high relative importance of total phosphorus in the model is expected. The relative importance (and positive relationship) of the number of pieces of large woody debris with TDI is a bit surprising and may warrant further investigation as a potential stimulatory stressor (sensu Wagenhoff et al., 2011). Based on the BRT analysis, there is no compelling evidence that TDI is causally influenced by any of the natural background factors evaluated in this study.

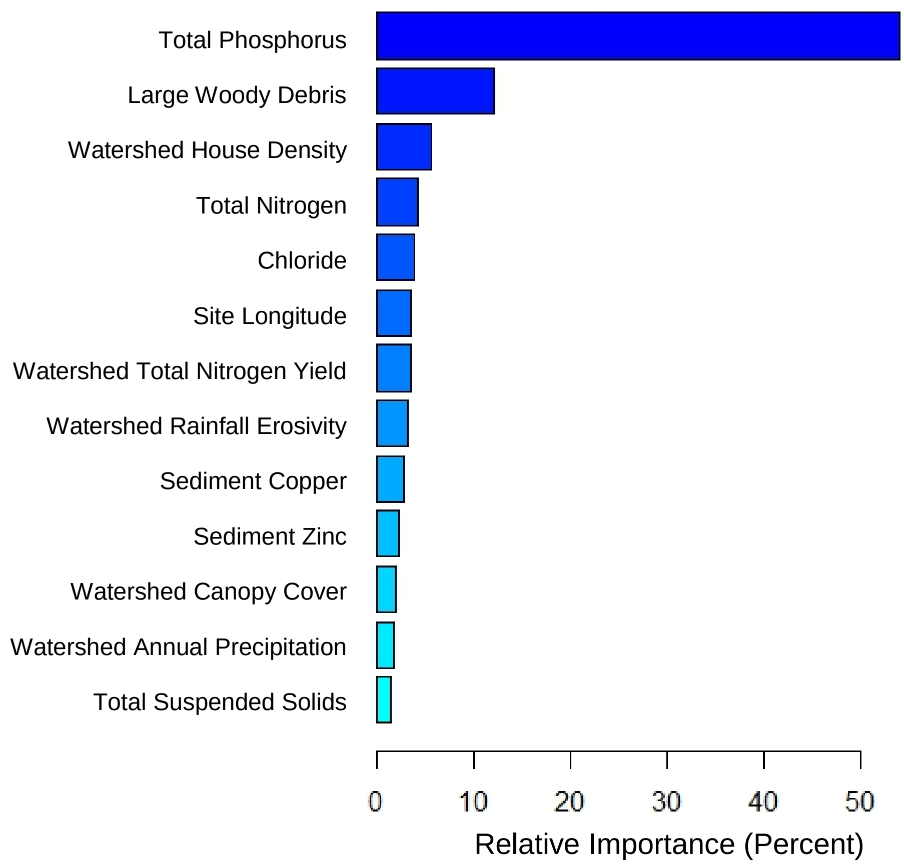


Figure 78. Relative importance of natural and human stressor variables in the final Trophic Diatom Index boosted regression tree model.

3.4.2 Relative Risk/Attributable Risk

Results from the RR/AR analysis are described for B-IBI and TDI scores in the following sections reflecting the three steps outlined in Section 2.7.3.2:

- Extent of poor condition
- Relative risk analysis
- Attributable risk analysis

3.4.2.1 Extent of Poor Condition

Across the study area, the extent of poor condition for B-IBI and the stressors selected for evaluation of relative risk/attributable risk (next section) is shown in Figure 79. The top three stressor variables with the greatest extent classified in poor condition were landscape metrics (Figure 79). Note that 40 percent of Puget Lowland stream length was estimated to be in poor biological condition based on B-IBI scores.

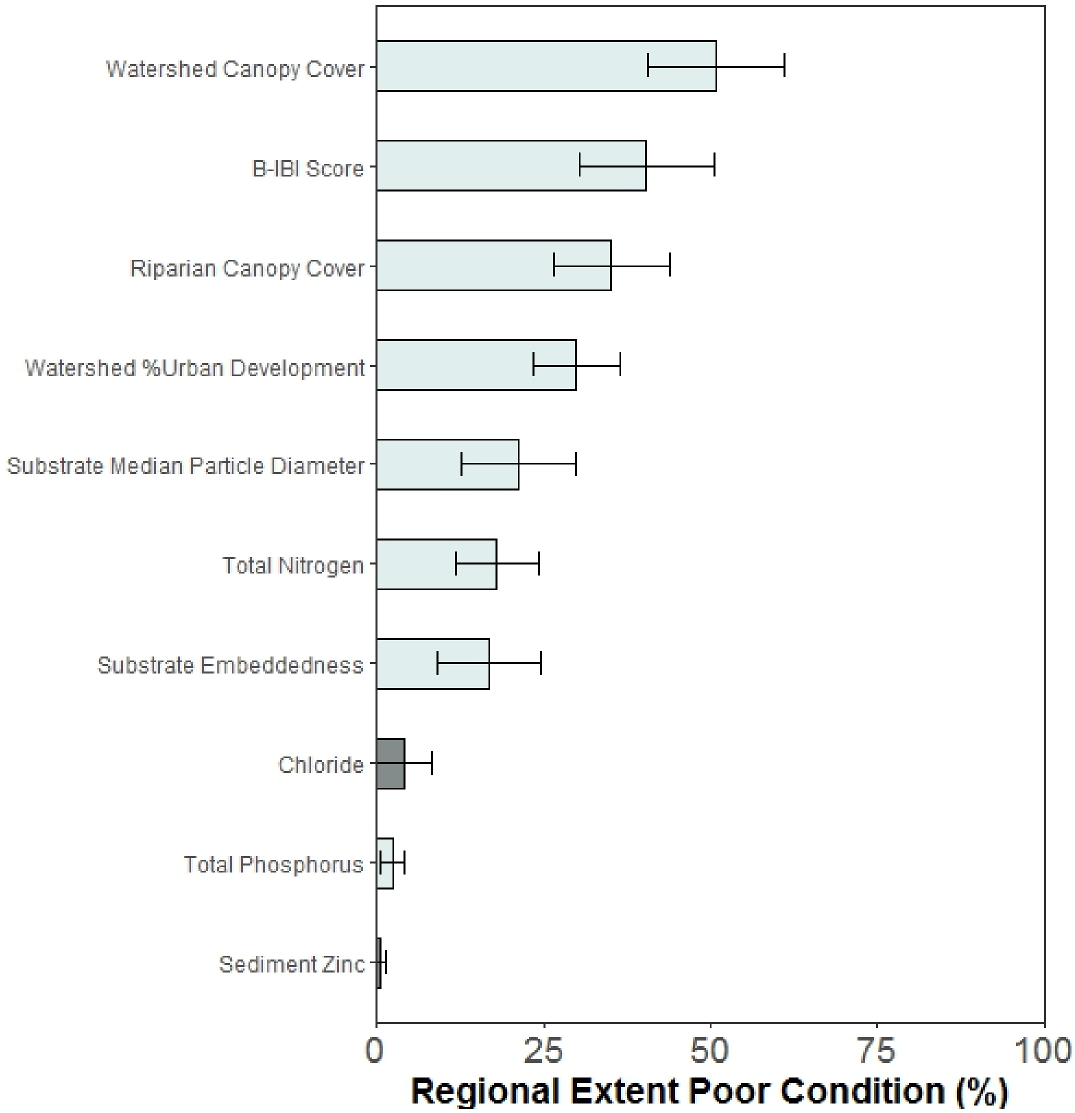


Figure 79. Relative extent of stream length classified in poor condition for the stressor metrics and B-IBI scores.

Note: Bars that are shaded dark gray indicate that the estimated extent of stream length in poor condition is not significantly different from zero. Error bars are 95% confidence limits.

With the exception of stream chloride concentrations (Aug-Oct mean concentrations), the extent of stream length classified in poor condition was greater within UGAs relative to outside UGAs (not shown).

3.4.2.2 Relative Risk Factors to B-IBI Scores

The factor with the highest relative risk of finding poor B-IBI scores was watershed canopy cover (RR= 3.8), followed by sediment zinc concentration (RR=2.5), riparian canopy cover (RR=2.5), and watershed percent urban development (RR=2.4) (Figure 80). The estimated relative risk for chloride concentration (RR=1.3) was not statistically significant –

essentially equivalent to a relative risk of 1.0, which means no increased risk of poor B-IBI scores. In other words, despite the finding above that about 4 percent of the region's streams are in poor condition due to chloride, low B-IBI scores are not closely associated with poor chloride condition. Whereas poor B-IBI scores are 3.8 times more likely to occur in regional streams with poor watershed canopy cover than in streams with good watershed canopy cover.

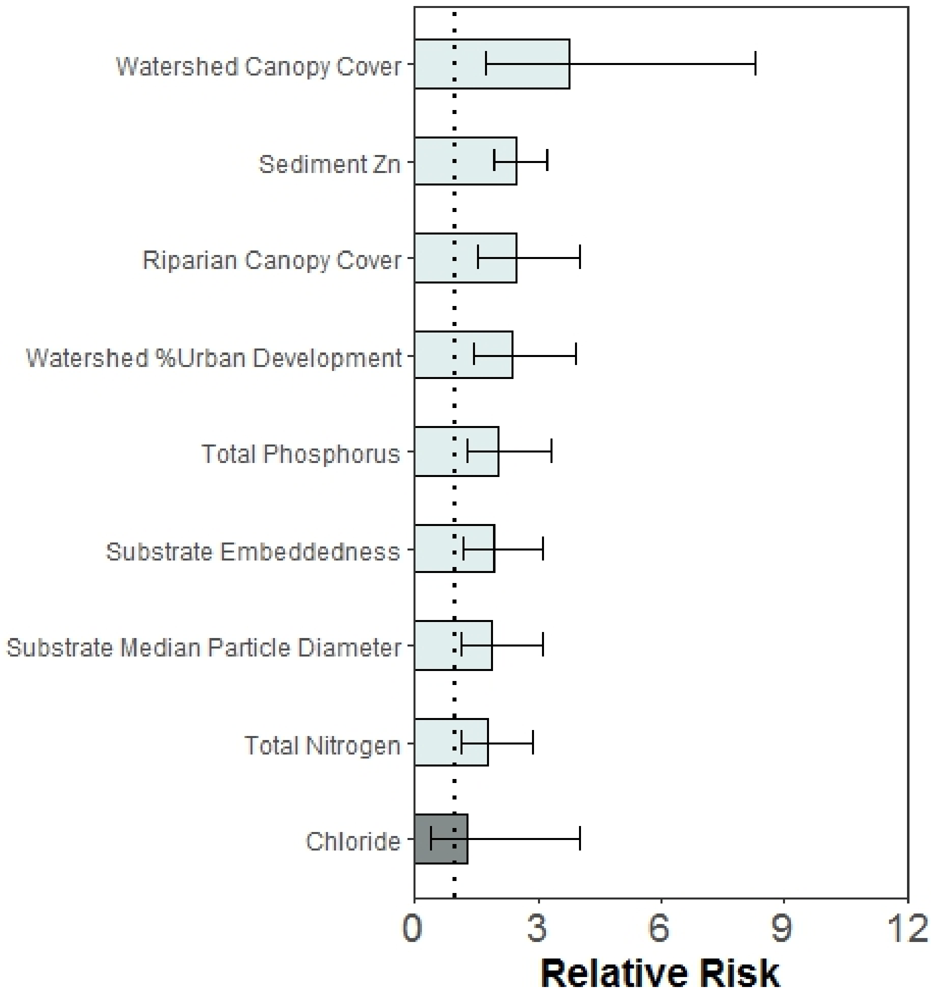


Figure 80. Relative risks to B-IBI scores and their 95 percent confidence intervals.

Note: Bars that are shaded dark gray indicate that the estimated relative risk is not significantly greater than one. Error bars are 95% confidence limits.

3.4.2.3 Attributable Risk Factors to B-IBI Scores

The greatest attributable risk to low B-IBI scores was associated with watershed canopy cover (AR=59 percent), followed by riparian canopy cover (AR=34 percent), then watershed percent urban development (AR=29 percent) (Figure 81). With the exception of total nitrogen, the remaining stressor attributable risk estimates were not statistically significant in this assessment. Although median particle diameter and substrate embeddedness were only nominally insignificant, these parameters were previously identified as having the highest attributable risk to B-IBI scores in western Washington streams (King County, 2014a) and in a Washington-wide analysis (Ecology, in preparation).

As an example of the interpretation of these results, if riparian canopy cover could be improved in reaches classified currently to be in poor riparian canopy cover condition, then a 34 percent reduction in the extent of stream reaches classified in poor B-IBI condition might be expected. Note that a large AR does not imply a causal relationship between the stressor and the biological response and interactions among stressors are not accounted for in the approach used in this study (Van Sickle and Paulsen, 2008; Van Sickle, 2013). Additional lines of evidence would provide stronger support for a causal relationship. For example, the positive association of riparian cover and B-IBI scores in Puget Lowland streams was highlighted by Morley and Karr (2002). The results of the AR analysis also do not ensure reversibility so it is recommended to interpret these results as best-case scenarios or to evaluate restoration priorities given the AR rankings (Van Sickle and Paulsen, 2008).

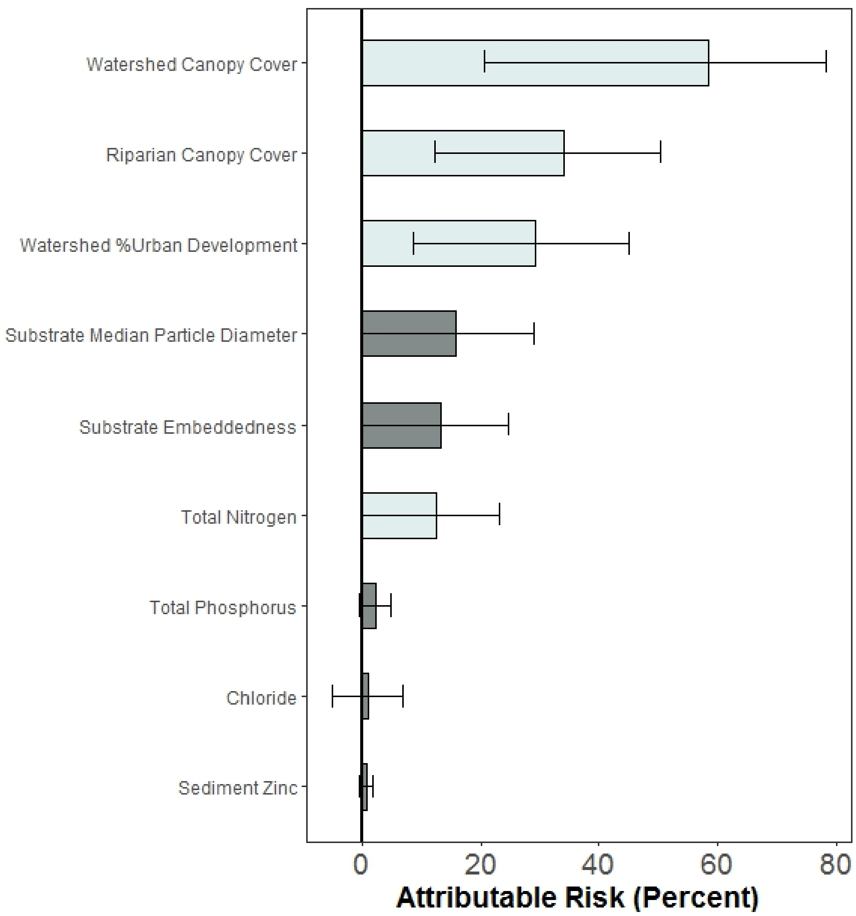


Figure 81. Attributable risks to B-IBI scores and their 95 percent confidence intervals.

Note: Bars that are shaded dark gray indicate that the attributable risk is not significantly different from zero. Error bars are 95% confidence limits.

3.4.2.4 Relative Risk/Attributable Risk for Trophic Diatom Index

Because the TDI was developed as a stream eutrophication indicator, the RR/AR analysis focused on landscape variables and stream nutrient concentrations. The relative extent of the stressors selected for evaluation of TDI scores is shown in Figure 82. The landscape metric, watershed percent urban development, was the variable with the greatest extent classified in poor condition; about 30 percent of the target stream length was classified in poor condition. The extent of stream length classified in poor condition based on stream total phosphorus and nitrogen concentrations (August-October means) was similar; about 20 percent of stream length was classified in poor condition. 36 percent of Puget Lowland stream length was estimated to be in poor biological condition based on the TDI.

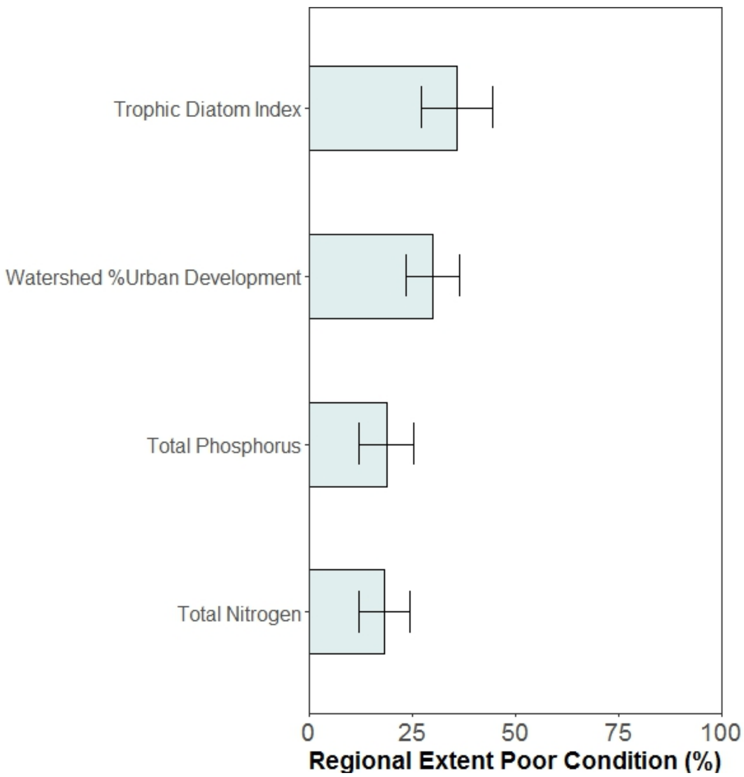


Figure 82. Relative extent of stream classified in poor condition for the stressor metrics (and Trophic Diatom Index) evaluated in the Trophic Diatom Index relative risk/attributable risk analysis.

Note: Error bars are the 95% confidence limits.

The highest relative risk (RR=3.5) for TDI scores was associated with total phosphorus, which is logical and confirming, followed by watershed percent urban development (RR=2.3) (Figure 83). Estimated relative risk for total nitrogen (RR=2.0) was also statistically significant, but slightly lower than that for watershed percent urban development.

The greatest estimated attributable risk was associated with total phosphorus (AR=31 percent) followed by watershed percent urban development (28 percent), both of which were statistically significant (Figure 84). The attributable risk estimate for total nitrogen (AR=15 percent) was also statistically significant. The interpretation of the total phosphorus results is that a 31 percent reduction in the extent of poor TDI condition could be achieved if stress due to elevated total phosphorus concentrations were be eliminated (by reducing summer total phosphorus concentrations below 0.050 mg/L) along with the caveats noted above for the B-IBI AR analysis.

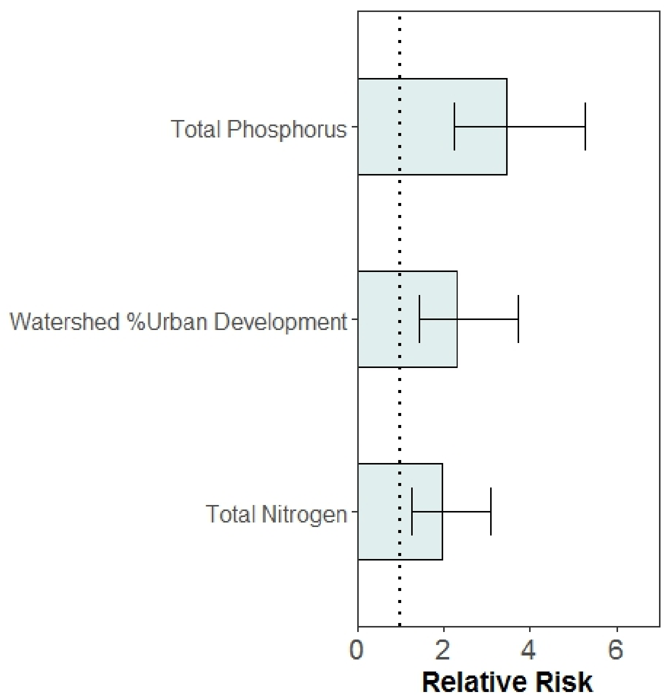


Figure 83. Relative risks to Trophic Diatom Index scores and their 95 percent confidence intervals.

Note: Error bars are the 95% confidence limits.

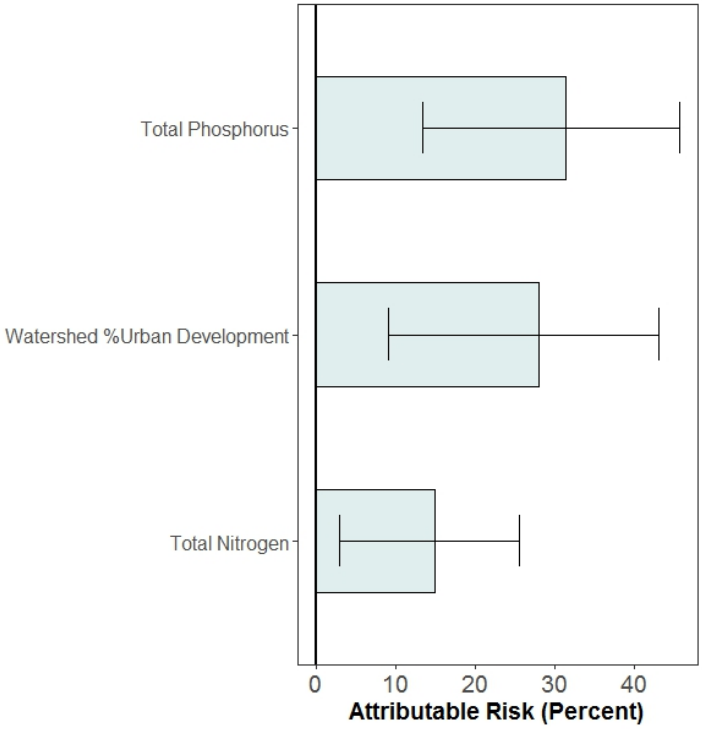


Figure 84. Attributable risks to Trophic Diatom Index scores and their 95 percent confidence intervals.

Note: Error bars are the 95% confidence limits.

4.0 COMPARISON OF SAM PLES TO OTHER PUGET LOWLAND MONITORING PROGRAMS

In this section we address the SWG request to compare SAM PLES study results to other similar regional and local Puget Lowland stream sampling programs. The main question being addressed in this section is:

- **Q4:** How do SAM PLES results compare to other stream monitoring programs in the Puget Lowlands?
    - **Q4.1:** Compare to other probabilistic sampling programs.
    - **Q4.2:** Compare to targeted sampling programs.

In general, there are two types of long-term environmental monitoring designs used to quantify status and trends resulting from local and upstream watershed changes. The first is based on the selection of sampling sites primarily using professional judgement (here called "targeted" designs). These designs generally focus on evaluating status and trends at these targeted sites. The second type of monitoring design is based on the random selection of sampling sites from a specified population of potential sampling sites with the goal of inferring conditions from the sampled sites to the population of all potential sampling sites as well as regional trends (here called "probabilistic" designs). The probabilistic design develops quantitative estimates of the extent and quality of the streams within the population of streams sampled as well as the statistical certainty of those estimates. With targeted designs, results cannot confidently be extrapolated across the region, although extrapolation could become possible through the development of a statistical or mechanistic model based on the targeted data or a combination of targeted and probabilistic data (e.g., Paul et al., 2008; Mass-Hebner et al., 2015).

In cases where comparisons between targeted and probabilistic study designs have been made, relatively significant errors in estimates of regional population statistics have been found when targeted study data are used to infer regional conditions (e.g., Peterson et al., 1999). However, differences appear to become smaller as the number of targeted sites increases and targeted sites are selected to represent the same strata or stressor gradient that is the focus of the probabilistic design (Collier and Olsen, 2013).

4.1 Programs Selected for Comparison

Based on stakeholder interest within the SWG, we compared the SAM PLES probabilistic design to other regional stream monitoring efforts that are based on targeted or probabilistic designs. For the targeted programs, we selected only the largest programs to alleviate some of the inherent error in small sample sizes. When possible, data collected during the same year as the SAM PLES study (2015) were compared.

4.1.1 Probabilistic Programs

Puget lowland stream data from six probabilistic data sets were compared. Comparisons were made between the Option 1 and Option 2 subsets and the overall SAM PLES study and three local probabilistic programs:

- SAM PLES (combined Option 1 and 2 data)
- SAM Option 1 (Option 1-pooled fund data)
- SAM Option 2: City of Redmond and Pierce County data
- Ecology's Watershed Health and Salmon Recovery (WHSR) Monitoring project[24] (Puget Lowland ecoregion only)
- King County's Stream Bug Monitoring program[25]
- King County's Water Resource Inventory Area (WRIA) 8 Status and Trends Monitoring study[26]

Ecology's WHSR program samples Washington State rivers and streams on a regional rotating basis. There are eight regions, including the non-federal portion of the Puget Sound basin. Ecology's WHSR Puget Sound basin includes not only the Puget Lowland ecoregion, but also the Cascades, North Cascades, and Coast Range ecoregions. Site selection was stratified on stream order with an unequal number of sample sites within each stream order stratum (0^{th}, 1^{st}, 2^{nd}, 3^{rd}, and 4^{th} order or greater).

Ecology sampled streams in the Puget Sound basin in 2009 and again in 2013. 30 sites were sampled in Puget Lowland stream sites in 2009 and 25 sites were sampled in 2013 (see Appendix D Figures D-1 and D-2). The suites of parameters/metrics that were measured as part of this program provide the largest variety of potential comparisons, including comparisons of biological, physical habitat, and water and sediment chemistry. This program is also the closest comparison to the SAM PLES study with respect to field methods, as much of the SAM PLES work was modeled from the methods developed for the WHSR program.

King County's Stream Bug Monitoring program was redesigned and expanded in 2002 and evolved from earlier King County aquatic macroinvertebrate monitoring efforts. Stream sampling sites were randomly selected with a sampling goal of 8 to 10 sites per subbasin (King County, 2002). In 2002, benthic invertebrate samples were collected from 148 sites in 20 subbasins in the Green-Duwamish and Greater Lake Washington watersheds (King County, 2004). Fewer than 10 sites were sampled in some basins due to lack of access and/or suitable sampling substrate. Over time, some sites have been dropped and others

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[24] https://ecology.wa.gov/Research-Data/Monitoring-assessment/River-stream-monitoring/Habitat-monitoring/Watershed-health
[25] http://www.kingcounty.gov/services/environment/data-and-trends/monitoring-data/stream-bugs.aspx
[26] http://www.kingcounty.gov/depts/dnrp/wlr/sections-programs/science-section/doing-science/wadeable-streams.aspx

added and in 2014 monitoring was expanded to include subbasins in the Snoqualmie River basin. In 2015, 141 sites were sampled as part of this program (see Appendix D Figure D-3).

King County's WRIA 8 Status and Trends study was a four year (2010-2013) pilot monitoring effort supported in part by a U.S. EPA grant (King County, 2015). Site selection focused on wadeable streams within three Chinook recovery tier areas. A total of 52 sites within WRIA 8 were sampled each year from 2010 to 2012 and 51 sites were sampled in 2013 for benthic invertebrates, fish, and physical habitat data. No water or sediment quality data were collected during this study (see Appendix D Figure D-4).

4.1.2 Targeted Programs

We solicited data from multiple jurisdictions for comparison to the 2015 SAM PLES study results. Targeted programs selected and the year the effort took place include:

- USGS's 2015 Pacific Northwest Stream Quality Assessment (PNSQA) in WA[27]
- King County's 2004-2012 Freshwater Sediment Monitoring program[28]
- King County's 2015 Stream Water Quality Monitoring program[29]
- Kitsap County's 2013-2016 Watershed Health Monitoring & Benthic programs[30]

The USGS 2015 Pacific Northwest Stream Quality Assessment sampled 88 stream sites in the Puget Lowlands and Willamette Valley; 33 sites were within the Puget Lowland ecoregion and suitable for comparison to the SAM PLES study (Sheibley et al., 2017b) (see Appendix D Figure D-5). Sites were sampled for water and sediment quality (and sediment toxicity), physical habitat, and benthic invertebrates, periphyton, and fish. The study design focused on sampling streams along a gradient of urbanization. The methods of the USGS study were generally similar to those of the SAM PLES study, including sediment sampling targeting depositional areas. Also consistent with the SAM PLES sediment sampling and analysis, sediment samples were sieved (<63 µm for metals and <2 mm for organic compounds) prior to laboratory analysis.

King County's freshwater sediment monitoring program, initiated in 2004, evolved from earlier efforts. The program includes a core set of sites in Puget Lowland streams within the Greater Lake Washington and Green-Duwamish basins. The core sites were sampled each year along with a rotating panel focusing on approximately three stream basins each year. Sediments were collected from depositional areas at each site and whole (not sieved) sediment was analyzed for a suite of metals (including acid volatile sulfide/simultaneously extracted metals) and organic contaminants, including PAHs, chlorinated pesticides, and Aroclor PCBs. Samples were also analyzed for ammonia nitrogen, total and soluble reactive phosphorus, pH, total sulfide, total organic carbon, and particle size distribution. For the comparisons presented here, a dataset was developed using the most recent sample from

[27] https://pubs.usgs.gov/fs/2015/3020/
[28] http://green2.kingcounty.gov/streamsdata/sediment.aspx
[29] http://green2.kingcounty.gov/streamsdata/
[30] http://www.cleanwaterkitsap.org/Pages/Watershed-Health-Monitoring.aspx

each site sampled through 2012 (funding for the program was suspended after 2012) providing data from 163 sites (sampled 2004-2012) for comparison to the 2015 SAM PLES data (see Appendix D Figure D-6).

King County's long-term stream water quality monitoring program was initiated in the early 1970s, evolving over time in response to changes in funding and design optimization efforts (e.g., Lettenmaier et al., 1984). In 2015, a network of 72 stream and river sites was sampled across King County on a monthly basis for stream water quality. 57 stream sites (river sites were excluded) were selected for comparison in this study (see Appendix D Figure D-7). Each month grab samples were collected and field instruments used to measure nutrient concentrations (dissolved and total forms of nitrogen and phosphorus), temperature, dissolved oxygen, pH, specific conductance, total alkalinity, total suspended solids, turbidity, and fecal coliform bacteria. The 2015 King County stream water quality monitoring data were compiled for use in the comparisons presented below.

Kitsap Watershed Health Monitoring Program includes stream flow, habitat, and benthic invertebrate monitoring. The stream habitat monitoring program began in 2012 and total of 33 stream sites have been sampled for habitat variables through 2016 (see Appendix D Figure D-8). The habitat metrics measured include several that are comparable to the ones measured as part of the SAM PLES study. The benthic invertebrate monitoring program began in 2010 targeting 51 stream sites through 2016. The sampling design involves a rotating panel with about half of the sites sampled every other year. The most recent benthic invertebrate sampling event from each of the 51 sites (sampled 2012-2016) was selected for comparison to the 2015 SAM PLES data (see Appendix D Figure D-9).

4.2 Methods for Comparison

Comparisons were based on treating both probabilistic and targeted programs as collections of sites, rather than including survey design weights available from the probabilistic programs to account for the unequal probability of selection of sites. This is an incorrect statistical analysis of the probabilistic study designs, but given the differences in spatial extent of many of the programs and the challenge of estimating design weights for targeted programs (Collier and Olsen, 2013), this approach provides a reasonable starting point for comparing results across several different monitoring programs.

The monitoring site location (latitude and longitude) from other programs were verified as existing in the Puget Lowlands for comparison with our study strata (within or outside UGAs) using ArcMap GIS tools. The within/outside UGA data was linked to the monitoring data in order to create box plots comparing the SAM PLES data (including comparisons of SAM PLES Option 1 and Option 2 data) to other program data.

We did not conduct a detailed evaluation of the comparability of each field and laboratory methods among the various programs, which could account for some of the differences observed (Roper et al., 2010; Sprague et al., 2017). We did identify important differences in methods when those differences were consistent with relatively large differences observed in results between programs.

4.3 Monitoring Indicators Selected for Comparison

A small set of the metrics and parameters measured as part of the SAM PLES were selected for comparison to data from other Puget Sound region monitoring programs. The metrics and parameters were chosen primarily on their presumed level of importance and management interest and the availability of comparable data from other monitoring programs. The following is a list of the metrics and parameters we selected for comparison:

- 0-100 scale B-IBI scores
- Water quality parameters: fecal coliform bacteria and total phosphorus
- Sediment quality parameters: copper and zinc
- Habitat parameters: substrate embeddedness and canopy closure (as measured with a densiometer)

A summary of the programs and parameters selected for comparison is provided in Table 29.

4.3.1 B-IBI

Figure 85 and Table 30 provide comparisons of 0-100 scale B-IBI scores for SAM PLES, SAM PLES Option 1 and Option 2, Ecology 2009 and 2013 Puget Sound region Watershed Health, King County Stream Bug Monitoring, and Kitsap County Benthic Invertebrate Monitoring data. Table 5 also includes the number of sites within and outside UGAs that were sampled by each monitoring program. Although there were differences in the median and range of scores among the various programs, there was a general pattern of lower scores within UGAs and higher scores outside of UGAs. Differences in scores among programs were likely due to differences in study designs (targeted vs probabilistic), the number of sites sampled, the type of streams targeted for sampling, and the geographic extent of the streams sampled. For example, Option 2 results were not as similar because only a small portion of Puget Lowland ecoregion streams were represented (i.e., only streams within the City of Redmond and unincorporated Pierce County).

Table 29. Summary of programs and metrics/parameters compared to the SAM PLES study, including the number of program sites within and outside UGA.

Program	Type of Site	B-IBI		Water Quality (FC and TP)		Sediment Quality (Cu and Zn)		Habitat Quality (Embeddedness and canopy closure)	
		OUGA	WUGA	OUGA	WUGA	OUGA	WUGA	OUGA	WUGA
SAM PLES	P	45	59	28	39	46	59	46	59
SAM PLES Option 1	P	37	48	22	28	37	48	37	48
SAM PLES Option 2	P	8	11	6	11	9	11	9	11
Ecology WHSR 2009	P	24	6	-	-	23	6	21	6
Ecology WHSR 2013	P	20	5	-	-	19	4	19	4
WRIA 8 S&T	P	22	29	-	-	-	-	22	29
King County benthos	P *	71	70	-	-	-	-	-	-
King County streams	T	-	-	22	35	-	-	-	-
King County sediment	T	-	-	-	-	58	105	-	-
Kitsap County	T	38	13	-	-	-	-	24	9
USGS 2015 PNSQA	T	-	-	-	-	21	25	-	-

P = Probabilistic design; T = Targeted design; FC = Fecal Coliform, TP = Total Phosphorus; Cu = Copper; Zn = Zinc

** The King County Stream Bug Program sites were selected randomly, but adjusted spatial weights are not currently available.*

Table 30. Comparison of median B-IBI scores outside and within Urban Growth Areas (UGAs) for selected Puget Sound monitoring programs.

Programs	Number of sites		Median	
	OUGA	WUGA	OUGA	WUGA
SAM PLES	45	59	72.4	39.3
SAM Opt1	37	48	73.9	40.1
SAM Opt2	8	11	63.7	36.9
EcoWHSR 2009	24	6	66.5	60.3
EcoWHSR 2013	20	5	58.3	35.3
King Co 2015	71	70	65.7	35.5
Kitsap Co 2010-2016	38	13	68.3	48.6
WRIA08 S&T 2013	22	29	82.0	34.3

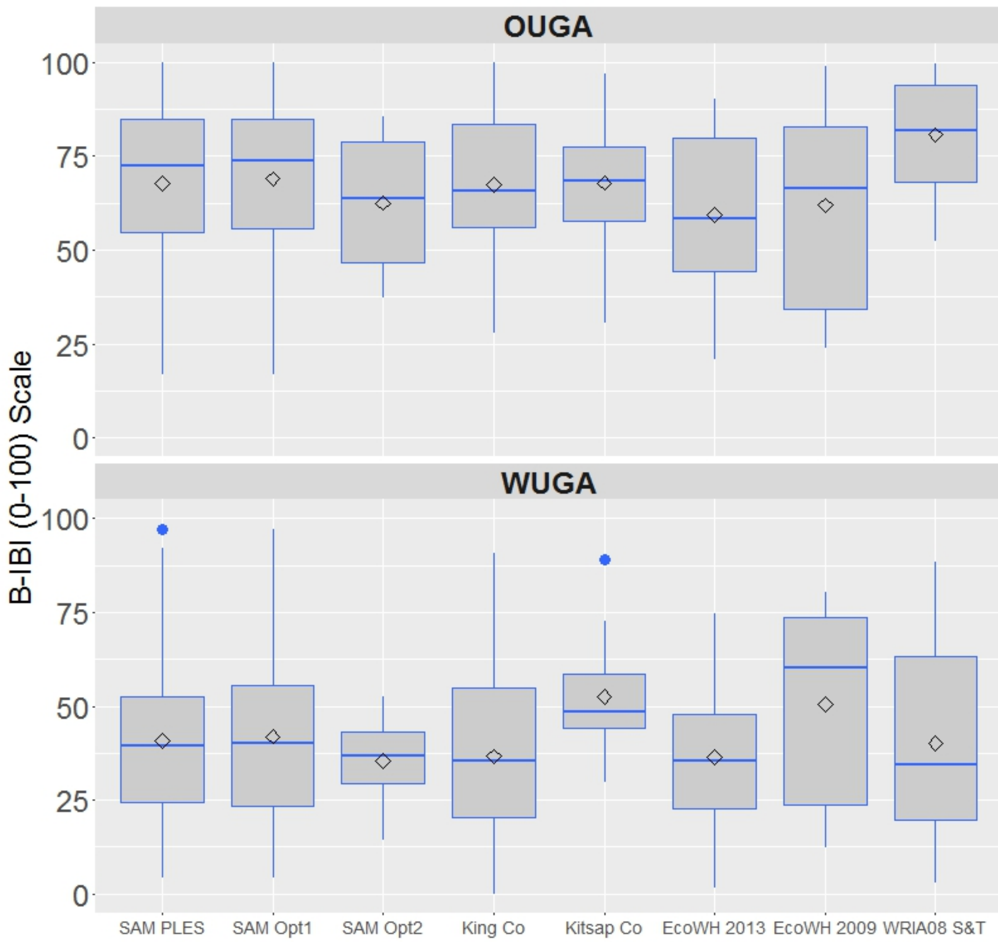

Figure 85. Box plot comparing 0-100 scale B-IBI scores for probabilistic and targeted sampling programs conducted in the Puget Sound region.

Note: See Section 2.7.2 for a detailed description of the information presented in a box plot.

4.4 Water Quality

Comparisons of water quality monitoring programs were made for geometric mean fecal coliform bacteria concentrations and mean total phosphorus concentrations based on annual monthly sampling conducted in 2015. Recall that the monthly water quality sampling for SAM regional sampling occurred during the 2015 calendar year, while monthly Option 2 sampling occurred during the 2015 water year (i.e., October 2014 to September 2015).

4.4.1 Fecal Coliform

Although there were differences in the medians and range of geometric mean fecal coliform concentrations measured in 2015 by the programs compared, the general pattern of lower concentrations outside UGAs and higher concentrations within UGAs was consistent among the SAM PLES (Figure 86 and Table 31). Median concentrations determined from the SAM PLES Option 2 and the King County programs were generally higher within and outside UGAs relative to SAMP PLES and SAM PLES Option 1. Absolute differences among the programs were likely due to a combination of factors including differences in study designs, sample sizes, and the spatial distribution and geographic extent of the sites sampled.

Table 31. Comparison of median geometric mean fecal coliform concentrations (cfu/100 mL) outside and within Urban Growth Areas (UGAs) for selected Puget Sound monitoring programs.

Programs	Number of sites		Median	
	OUGA	WUGA	OUGA	WUGA
SAM PLES	28	39	21	60
SAM Opt1	22	28	20	55
SAM Opt2	6	11	43	75
King Co 2015	22	35	44	86

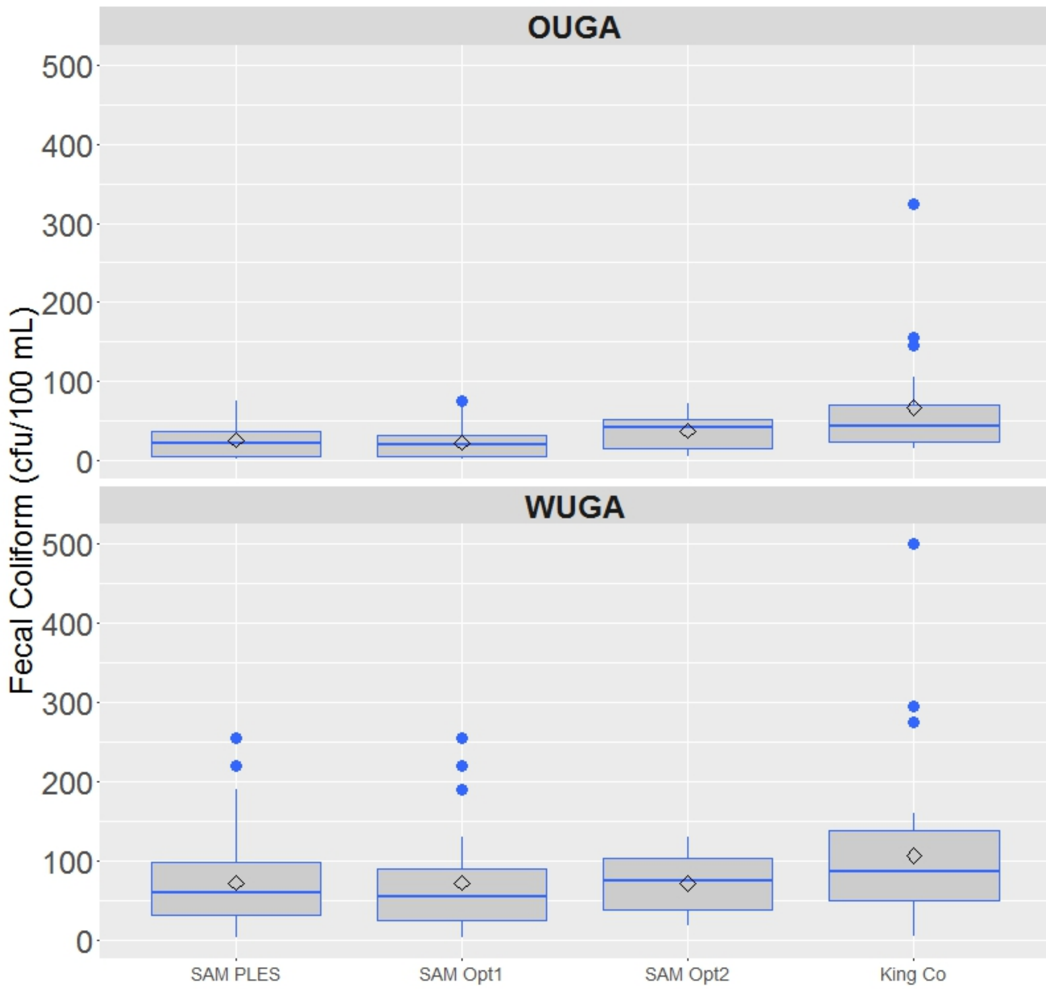

Figure 86. Box plot comparing geometric mean fecal coliform concentrations for probabilistic and targeted sampling programs conducted in the Puget Sound region.

Note: See Section 2.7.2 for a detailed description of the information presented in a box plot.

4.4.2 Total Phosphorus

Although the median and ranges of 2015 annual mean total phosphorus were different among the various programs, a general pattern of lower concentrations outside UGAs and higher concentrations within UGAs was observed (Figure 87 and Table 32). However, median concentrations determined from the SAM PLES Option 2 and the King County programs were generally higher outside UGAs relative to SAM PLES and SAM PLES Option 1.

Table 32. Comparison of annual mean total phosphorus concentrations (mg/L) outside and within Urban Growth Areas (UGAs) for selected Puget Sound monitoring programs.

	Number of sites		Median	
Programs	**OUGA**	**WUGA**	**OUGA**	**WUGA**
SAM PLES	28	39	0.026	0.043
SAM Opt1	22	28	0.021	0.043
SAM Opt2	6	11	0.035	0.043
King Co 2015	22	35	0.041	0.049

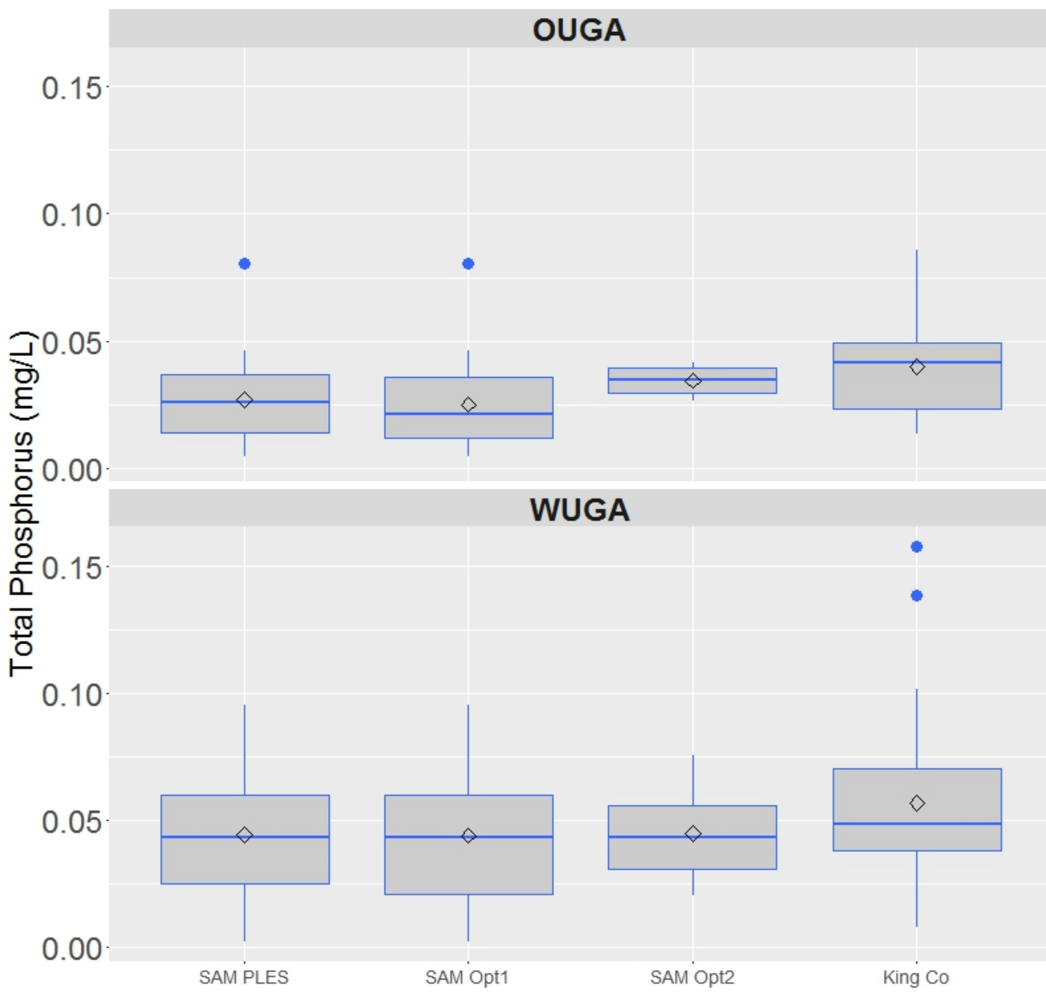

Figure 87. Box plot comparing annual mean total phosphorus concentrations for probabilistic and targeted sampling programs conducted in the Puget Sound region.

Note: See Section 2.7.2 for a detailed description of the information presented in a box plot.

4.5 Sediment Quality

Comparisons were made for SAM PLES, SAM PLES Option 1 and Option 2, Ecology 2009 and 2013 Puget Sound region Watershed Health, King County's Sediment Quality Monitoring Program (for sites sampled between 2004 and 2012), and the Puget Sound region sites sampled as part of the USGS PNSQA Program in 2015. Comparisons of sediment quality monitoring programs were made for copper and zinc concentrations based on annual grab samples. King County's sediment and Ecology's WHSR programs measured concentrations in whole sediment, while the SAM PLES and USGS PNSQA programs analyzed sieved sediment.

4.5.1 Copper

Sediment copper results among the programs compared were not as consistent as the various sampling program results for B-IBI or water quality. The overall SAM PLES, SAM Option 1, and USGS PNSQA results were most consistent, indicating little difference in sediment copper concentrations within vs outside UGAs (Figure 88 and Table 33). SAM Option 2 and the King County program indicated lower median sediment copper concentrations outside UGAs. The King County Sediment Monitoring Program sediment concentrations were typically lower within and outside UGAs relative to other programs. The lower King County sediment concentrations were likely due to the difference in analyzing whole vs. sieved samples. Metals and many other contaminants typically concentrate on the finest sediment particles if for nothing more than the relatively larger surface area provided for sorption, so sieved sediments tend to have higher concentrations than a whole sediment sample from the same site (Horowitz, 1991). The Ecology WHSR data from 2009 and 2013 suggested the opposite pattern of the SAM Option 2 and King County data; lower concentrations within UGAs vs outside UGAs.

Table 33. Comparison of sediment copper concentrations (mg/Kg) outside and within Urban Growth Areas (UGAs) for selected Puget Sound monitoring programs.

	Number of sites		Median	
Programs	**OUGA**	**WUGA**	**OUGA**	**WUGA**
SAM PLES	46	59	34.7	34.1
SAM Opt1	37	48	39.1	36.9
SAM Opt2	9	11	15.8	30.3
Ecology WHSR 2009	23	6	24.6	14.9
Ecology WHSR 2013	19	4	26.7	13.3
King Co 2004-2012	58	105	10.1	16.4
USGS PNSQA 2015	21	25	38.1	37.7

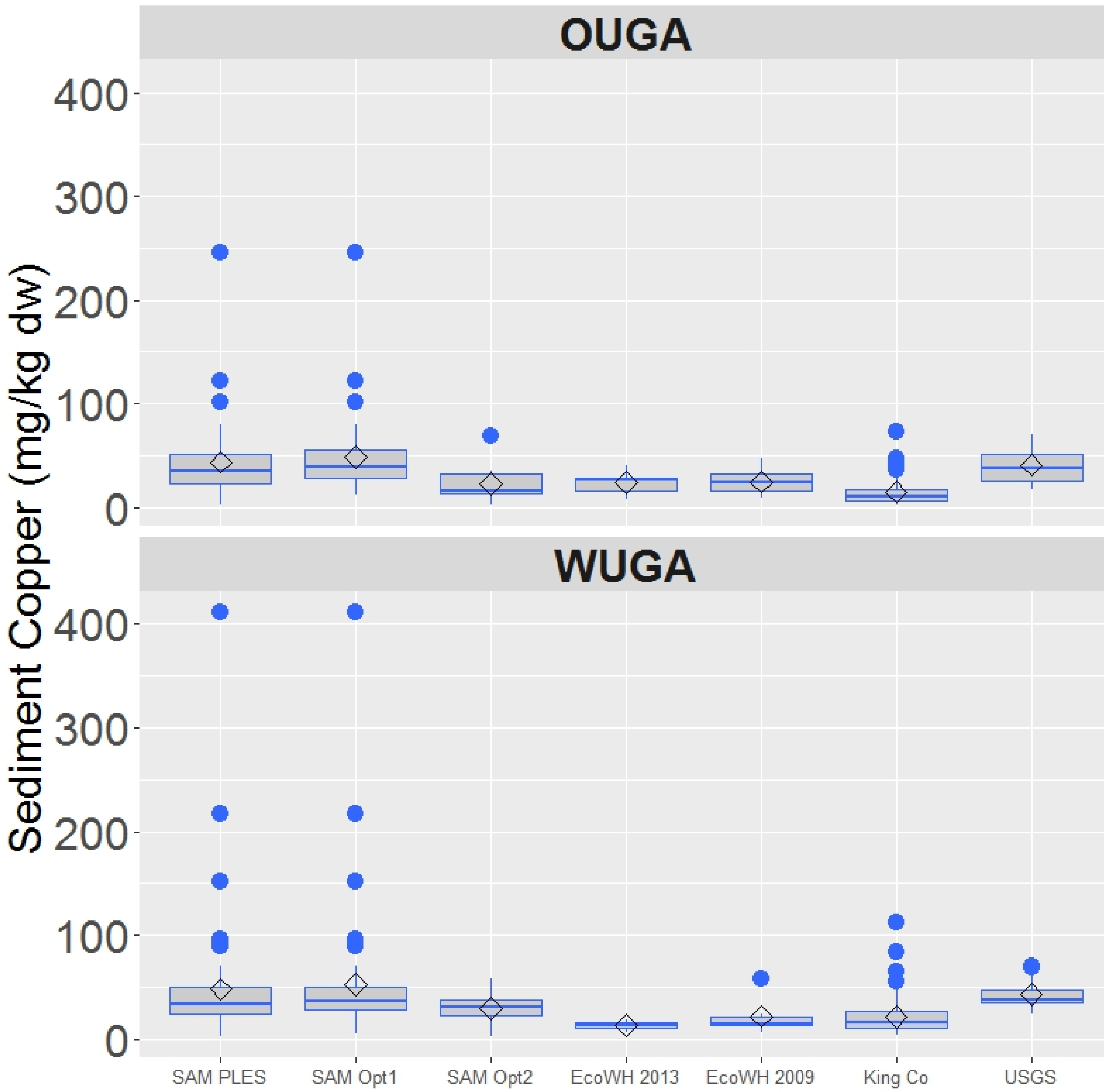

Figure 88. Box plot comparing sediment copper concentrations for probabilistic and targeted sampling programs conducted in the Puget Sound region.

Note: See Section 2.7.2 for a detailed description of the information presented in a box plot.

4.5.2 Zinc

Differences in sediment zinc results within and outside UGAs for each program were more consistent than for the sediment copper results. The SAM PLES, SAM Option 1, SAM Option 2, King County, and USGS PNSQA indicated higher zinc concentrations within vs outside UGAs (Figure 89 and Table 34). Similar to the sediment copper comparisons, King County sediment zinc concentrations were typically lower than the results from SAM PLES or USGS PNSQA programs, likely due to the analysis of whole vs sieved sediment. The Ecology WHSR 2009 and 2013 median zinc concentrations (also based on whole sediment analyses) were also lower than all but the King County program data. There was also a much smaller difference in median concentrations within and outside UGAs in the Ecology WHSR 2009 and 2013 data sets.

Table 34. Comparison of sediment zinc concentrations (mg/Kg) outside and within Urban Growth Areas (UGAs) for selected Puget Sound monitoring programs.

	Number of sites		Median	
Programs	**OUGA**	**WUGA**	**OUGA**	**WUGA**
SAM PLES	46	59	81.1	153
SAM Opt1	37	48	82.1	165
SAM Opt2	9	11	76.9	134
Ecology WHSR 2009	23	6	50.8	52.6
Ecology WHSR 2013	19	4	51.3	63.3
King Co 2015	58	105	36.7	91.9
USGS PNSQA 2015	21	25	107	188

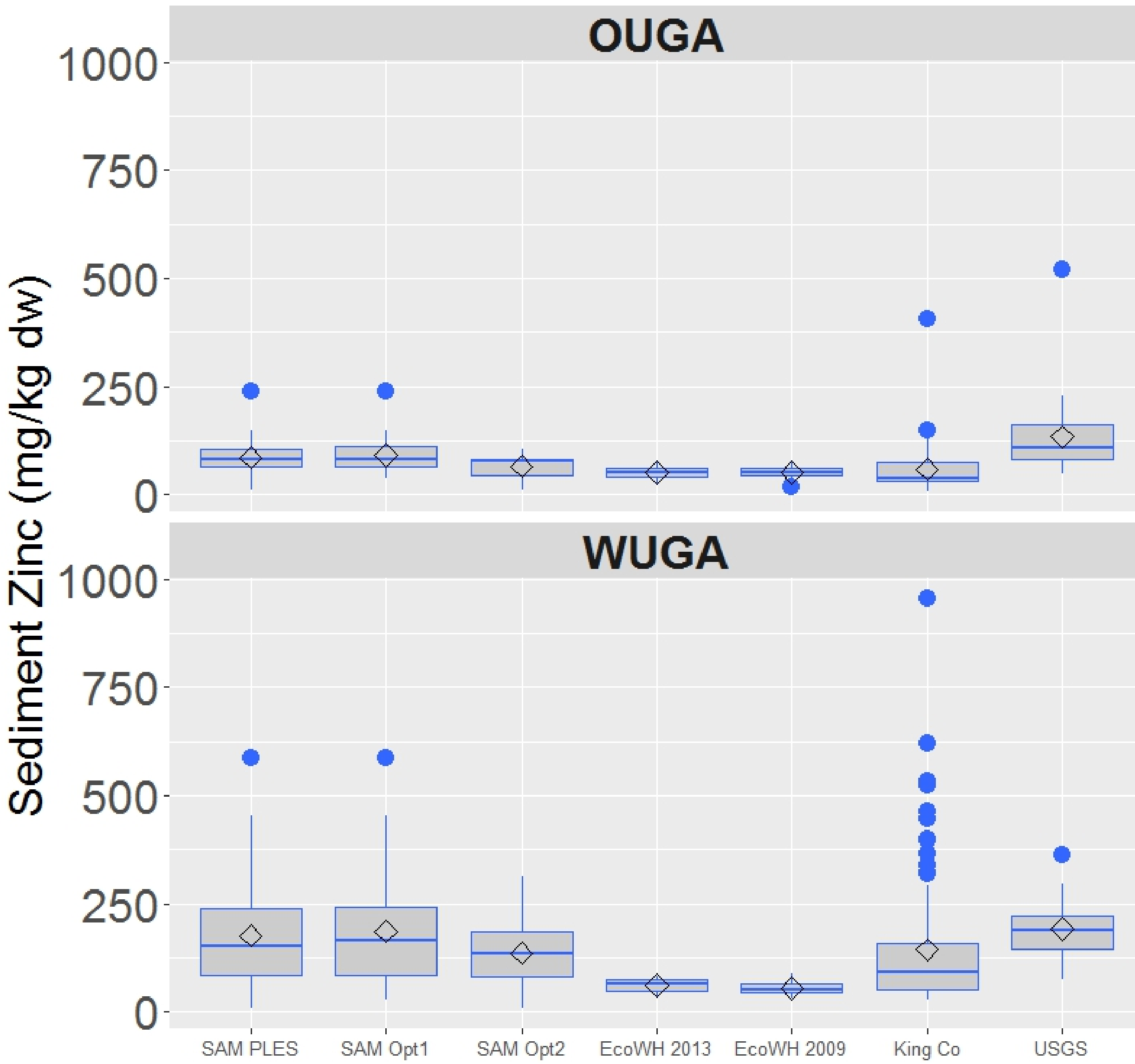

Figure 89. Box plot comparing sediment zinc concentrations for probabilistic and targeted sampling programs conducted in the Puget Sound region.

Note: See Section 2.7.2 for a detailed description of the information presented in a box plot.

4.6 Stream Habitat

Comparisons were made for SAM PLES, SAM PLES Option 1 and Option 2, Ecology 2009 and 2013 Puget Sound region WHSR, Kitsap County Watershed Health (for sites sampled from 2012–2016), and King County's WRIA 8 Status and Trends Study (for sites sampled in 2013). Comparisons of habit data from various monitoring programs were made for stream embeddedness and stream center canopy cover.

4.6.1 Percent Embeddedness

There were a number of differences in the median and range of scores among the various programs (Figure 90 and Table 35). The results from the SAM PLES and Ecology WHSR 2009 studies indicate less embeddedness in streams sites outside UGAs, while the Ecology WHSR 2013, Kitsap County Watershed Health, and WRIA 8 Status and Trends programs suggest the opposite. Differences in stream embeddedness estimates among programs were likely due to differences in study designs (targeted vs probabilistic), the number of sites sampled, type of streams targeted for sampling, and the geographic extent of the streams sampled.

Table 35. Comparison of stream embeddedness (percent) outside and within Urban Growth Areas (UGAs) for selected Puget Sound monitoring programs.

	Number of sites		Median	
Programs	**OUGA**	**WUGA**	**OUGA**	**WUGA**
SAM PLES	46	59	45.2	60.2
SAM Opt1	37	48	41.3	59.0
SAM Opt2	9	11	51.0	64.3
Ecology WHSR 2009	21	6	48.9	62.0
Ecology WHSR 2013	19	4	53.5	39.2
Kitsap Co 2012-2016	24	9	44.1	42.3
WRIA08 S&T 2013	22	29	63.2	58.5

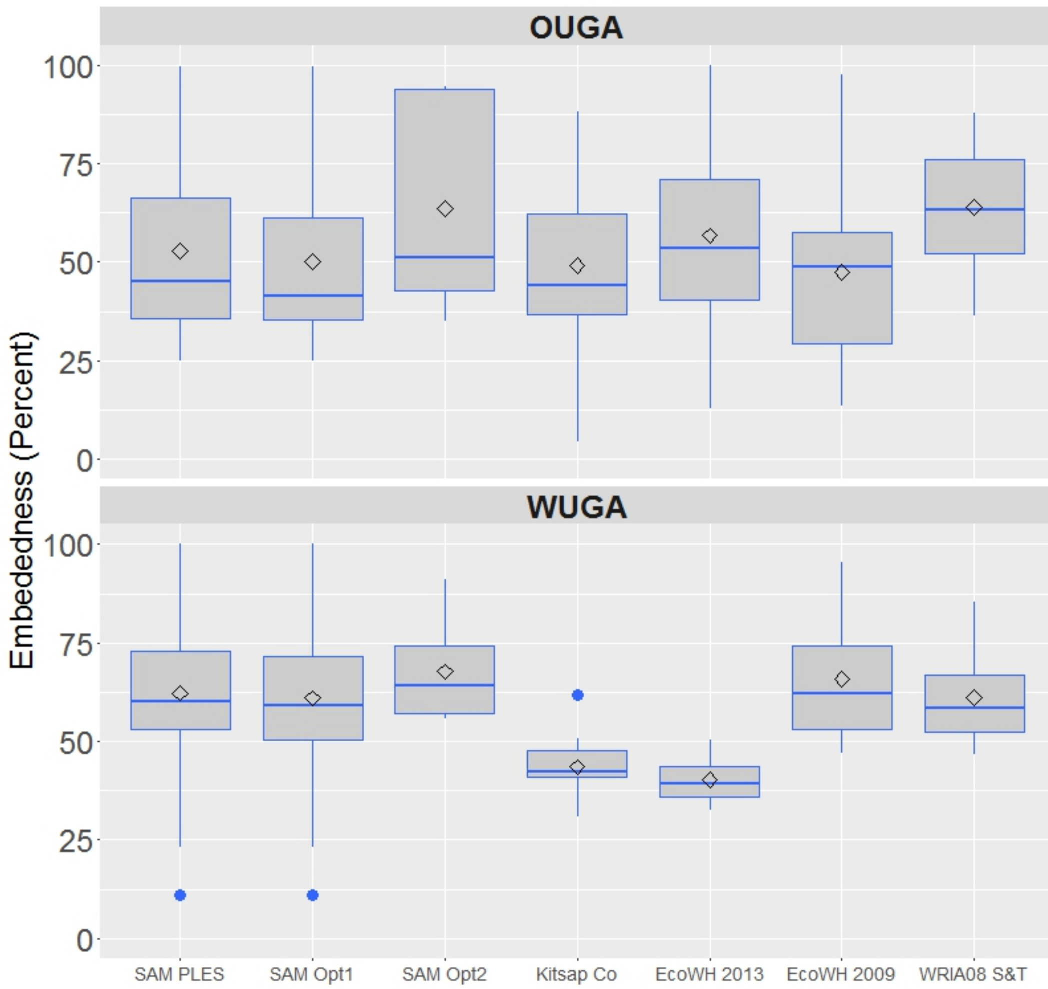

Figure 90. Box plot comparing stream embeddedness for probabilistic and targeted sampling programs conducted in the Puget Sound region.

Note: See Section 2.7.2 for a detailed description of the information presented in a box plot.

4.6.2 Percent Canopy Closure

With the exception of canopy closure measurements made as part of Ecology's WHSR 2013 study, median canopy closure measurements were similar (within about 5 percent or less) within and outside UGAs (Figure 91 and Table 36). The Puget Lowland data from Ecology's WHSR 2013 study percent canopy closure median was lower within UGAs relative to outside UGAs by about 30 percent. The range and distribution of stream canopy closure measurements were also quite different among the programs compared. Differences were likely due to differences in study designs (targeted vs probabilistic), the number of sites sampled, the type of streams targeted for sampling, and the geographic extent of the streams sampled.

Table 36. Comparison of stream center canopy closure (percent) outside and within Urban Growth Areas (UGAs) for selected Puget Sound monitoring programs.

	Number of sites		Median	
Programs	**OUGA**	**WUGA**	**OUGA**	**WUGA**
SAM PLES	46	58	90.0	90.6
SAM Opt1	37	48	86.0	91.4
SAM Opt2	9	10	95.7	85.2
Ecology WHSR 2009	17	5	83.0	86.1
Ecology WHSR 2013	11	3	84.4	65.5
Kitsap Co 2012-2016	24	9	86.0	90.2
WRIA08 S&T 2013	22	29	87.7	90.4

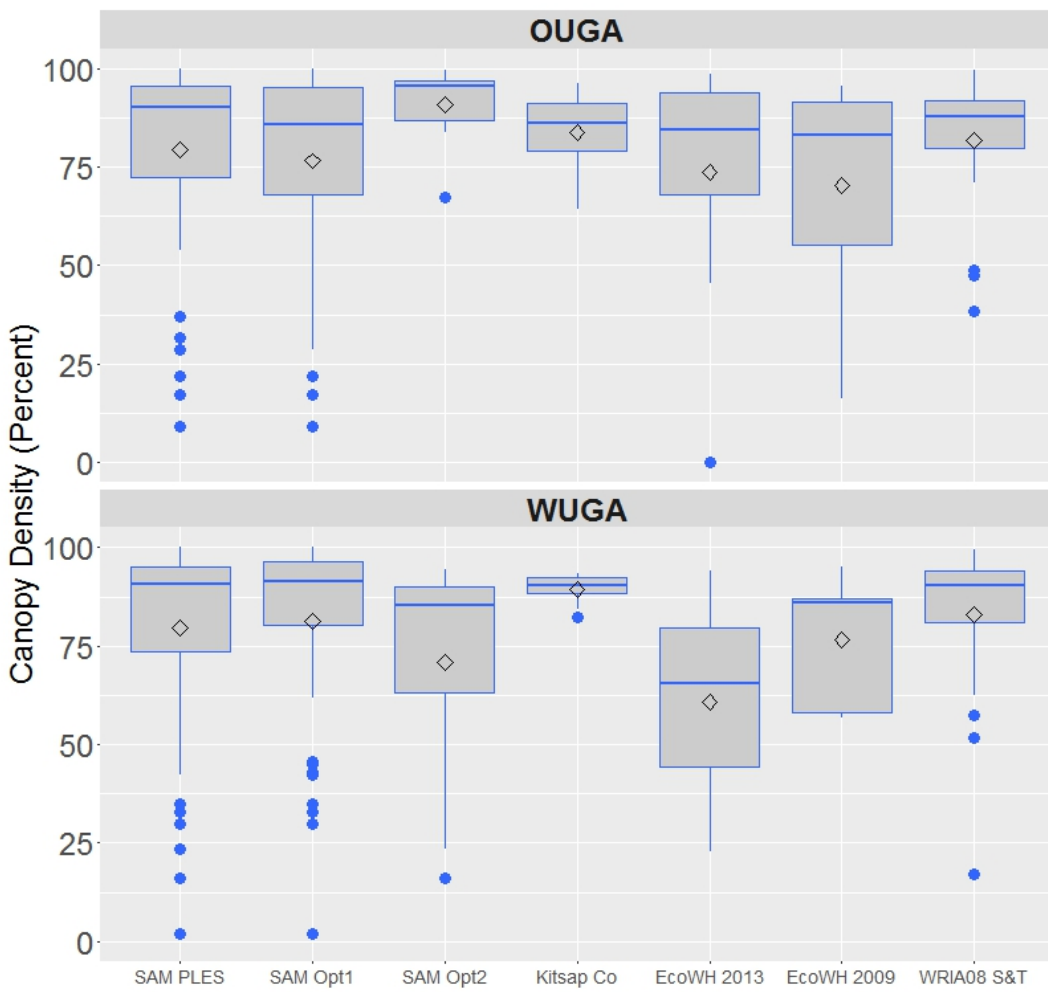

Figure 91. Box plot comparing stream center canopy closure for probabilistic and targeted sampling programs conducted in the Puget Sound region.

Note: See Section 2.7.2 for a detailed description of the information presented in a box plot.

4.7 Comparison of SAM PLES to Other Puget Sound Monitoring Programs Discussion

We anticipated that the programs would have slightly different median values for all the assessed parameters, which we found to be true. More importantly, we also found the range of values reported by the programs varied substantially. The smaller spatial extent of the studies or those with a smaller number of sites tended to produce the narrowest range of results.

We attribute variability in results between programs to differences in study designs (targeted vs probabilistic) and differences in methods. The larger differences are likely due to the distinct differences in methods used (e.g., analysis of whole vs. sieved sediment samples). The relative importance of the potential causes of differences identified above among habitat metric results (embeddedness in particular) cannot be determined for the large differences in metrics observed across programs. However, research has shown that some habitat metrics measured using similar (but not exactly the same) protocols may not produce comparable results across monitoring groups (Whiteacre et al., 2007; Roper et al., 2010).

Qualitatively, the B-IBI results appear to be most consistent across programs, particularly for programs with relatively large sample sizes (>30 sites within or outside UGAs). Although protocols for collecting benthic invertebrate data across programs differ to some degree (e.g., riffle-only sampling vs. random transects; sampling 3 vs 8 ft^2), evaluation of data collected with these different protocols have suggested that the data are fairly comparable (Gerth and Herlihy, 2006; Rehn et al., 2007; King County, 2014c).

Based on the relative similarity of the B-IBI results, one might conclude that there was no substantive difference between targeted and probabilistic programs. However, we do not believe that this is a reasonable conclusion that could be drawn from these comparisons. Probabilistic programs provide unbiased estimates of the status across the sample frame (in this case small streams distributed across the Puget Lowlands draining to Puget Sound). More localized programs are designed to address questions relevant to local monitoring and management objectives and are conducted on a more limited scale which precludes extrapolation of their results to the Puget Sound scale. The regional probabilistic sampling framework of the SAM small streams status and trends study also provides unbiased estimates of population statistics that can be used to put more local sampling information into a regional context. Ultimately, a regional Puget Lowland monitoring program is the only program that can reliably track regional (i.e., Puget Lowland) trends over time.

We also note that the probabilistic survey design provides a great opportunity to leverage other efforts nested within the same design framework (Larsen et al., 2008). It may also be possible to combine targeted non-probabilistic sampling data with data from probabilistic survey designs (Paul et al., 2008; Mass-Hebner et al., 2015). At a minimum, standardization of field and laboratory monitoring methods and a data sharing program would provide an even larger foundation of information that could be used to develop robust models (statistical and/or mechanistic) of biological stressor-response relationships (Isaak et al., 2014). The PSSB program and website is a good example of standardization and sharing of regional benthic invertebrate data, but standard field and laboratory protocols and data sharing across monitoring groups is more limited for flow, water quality, habitat, and sediment quality data.

5.0 REVIEW OF OTHER REGIONAL STATUS AND TRENDS STUDY DESIGNS

In this section we address the SWG's request to compare SAM PLES study design to other large-scale program study designs. The goal of this comparison is to learn from other large spatial scale designs at an opportune moment early in the SAM PLES long-term program.

We reviewed four large-scale programs for the study design (site selection, sample frequency, sample indicators/metrics) comparison:

- Lower Columbia habitat status and trends (LCHST) program
- Redmond Paired watershed study (RPWS)
- USGS Pacific Northwest Stream Quality Assessment (PNSQA)
- Southern California Stormwater Monitoring Coalition (SCSMC)

The Southern California Stormwater Monitoring Coalition program was the foundation for the original SAM PLES program, and was recently updated in 2015 for its next cycle of monitoring. These programs all had different goals. However, these programs share both unique and common features that will help to improve the SAM PLES for the future.

5.1 Lower Columbia Habitat Status and Trends Program

The LCHST program is designed to implement an integrated habitat and water quality status and trends monitoring program in the Lower Columbia River region in the near future. This program was initiated for two main reasons (1) to support recovery of salmonid species listed as threatened or endangered and (2) to assess the status and trends of urban streams in anticipation of future monitoring requirements under the National Pollution Discharge Elimination System (NPDES) municipal stormwater permits in southwest Washington. The program has not yet started, though the program design was completed in 2015 (Stillwater Sciences, 2015). The portion of this program related to assessing status and trends of urban streams is most relevant for comparing to the current SAM PLES program. It is anticipated to begin after the next permits are issued in 2019.

The program plans to use a probabilistic design for site selection using the Washington Master Sample list. Water quality, physical habitat, and biological sampling are proposed for the urban streams.

The sample frame for urban sites focused on small basins dominated by urban land use, which resulted in a small number of candidate sites (~30). Therefore, the entire population of urban stream water quality sites in NPDES permitted jurisdictions will be sampled on a rotating 5-year panel representing a true census sampling design. One fifth of the sites will be sampled every year as ‘trend’ sites. The remailing sites will be sampled with repeat visits every 5 years. The ‘trend’ sites sampled each year for water quality include legacy sites with previously collected data. This will allow for a quicker time frame for trend detection. For the habitat sites, the sample design calls for sampling 1/5th of the probabilistically selected sites each year with repeat visits every 5 years. This will allow a status assessment every year with trends determined from revisits on the 5-year interval.

The base program is a combination of discrete and continuous sampling. Discrete sampling is planned for sediment metals, sediment PAHs, bank full width, bankfull depth, and substrate size once per five years. Continuous sampling is planned for temperature, conductivity, and stage. In addition, wetted width will be measured for each visit, and benthic macroinvertebrate sampling done annually. There is a list of ‘extended indicators’ that include monthly grab sampling for parameters that stakeholders of the program also want that is contingent on additional funding. This list of ‘extended indicators’ include: dissolved oxygen, pH, turbidity, total suspended solids, total solids, total and dissolved nutrients, dissolved copper and zinc, and fecal coliform bacteria.

Twenty-one habitat indicators were selected and include: reach length, channel type, reach slope, sinuosity, bank modification, density of habitat types, bankfull width/depth, pools per unit depth, floodplain width, side channel habitat, flow category, benthic inverts, residual pool depth, bank stability, relative bed stability, density/distribution of instream wood, substrate particle size, shade, riparian canopy, riparian understory, temperature. The procedures for the habitat metrics follow that of Ecology’s Watershed Health and Salmon Recovery Monitoring program. Habitat sampling would take place from July 1 to September 30.

The study design also identifies that landscape conditions can be important and used to help stratify sample sites. Repeated analysis of land cover change over 5-year intervals at the subbasin and riparian zone scale is planned.

5.2 Redmond Paired Watershed Study

The RPWS is a SAM effectiveness monitoring study designed to evaluate rehabilitation (restoration and stormwater retrofits) of urban basins using a pseudo before-and-after-control-impact (BACI) design. Monitoring was initiated in 2016 following the development of the experimental design and QAPP (Herrera, 2017). The study question is: “How effective are watershed rehabilitation efforts at improving receiving water conditions at the watershed scale?”

A paired watershed approach is used to examine if stormwater management is improving water quality and habitat conditions of receiving waters. The study design includes sampling 7 watersheds for study: 3 application watersheds where rehabilitation will be focused, 2 reference watersheds where no rehabilitation will take place, and 2 control watersheds already impacted and not targeted for rehabilitation. Trends will be assessed for a number of metrics over a 10-year period at fixed stations. Effectiveness will be determined by roving stations that will be used to target more local changes as projects are established, started, and completed.

Status and trends monitoring for the RPWS includes continuous flow, temperature, conductivity, and turbidity; annual sediment chemistry; and storm and base flow grab samples for water quality (TSS, DOC, TP, TN, total and dissolved Cu and Zn, fecal coliform bacteria, hardness); annual benthic invertebrate (B-IBI and other metrics); and annual physical habitat measurements.

5.3 USGS Pacific Northwest Stream Quality Assessment

In 2015, the USGS conducted the PNSQA to investigate stream quality across the western part of the Pacific Northwest. The goal of the PNSQA was to assess the health of streams in the region by characterizing multiple water-quality factors that are stressors to in-stream aquatic life and by evaluating the relation between these stressors and the condition of biological communities. The effects of urbanization and agriculture on stream quality for the Puget Lowland and Willamette Valley Level III ecoregions were the focus of this regional study. A detailed report on the methods used in PNSQA was published by Sheibley et al. (2017b) and briefly summarized below.

A targeted approach was used to select sites within the region that ranged in levels of urban and agricultural development. A total of 47 sites were selected across the Puget Lowlands on streams that explicitly spanned a range of urban land use in their watersheds, and included streams in agricultural and reference watersheds. Depending on the type of land use, sites were sampled for contaminants, nutrients, and suspended sediment for either a 4- or 10-week period during April, May, and June 2015. This water-quality "index period" was immediately followed with an ecological survey of all sites that included stream habitat, benthic algae, benthic macroinvertebrates, and fish. Additionally, streambed sediment was collected during the ecological survey for analysis of sediment chemistry and toxicity testing.

5.4 Southern California Program

The SCSMC used a probabilistic study design to assess coastal watersheds in their region. The SAM PLES study was based off of the original design, and the program was updated in 2015 for its second round of sampling. The five year study looked at four ecological condition indicators: benthic invertebrates, diatoms, soft algae, and riparian habitat. The survey was designed to answer key questions that are essential to watershed management:

1) What is the biological condition of perennial streams in the region?
2) What stressors are associated with poor condition?
3) Are conditions changing over time?

The first question is addressed by estimating the extent of biologically intact streams, as determined by key biological indicators. The second question is addressed by estimating the extent of streams with stressors above key thresholds, and by associating stress levels with biological indicators through correlation and relative risk analyses. The third question is addressed by comparing condition across years of the survey.

The four biological condition endpoints mentioned above were sampled at each site, as well as water quality (in situ measures and 36 other parameters), toxicity, physical habitat, and landscape variables. Each selected site was sampled once per 5-year period unless one of the participating agencies was already sampling a selected site. Status was assessed every year of the 5 year project, and trends in biological condition were determined over the 5-year period.

SCSMC was unable to detect trends in biological condition because of the relatively short time frame of the survey (i.e., 5 years), as well as a study design that did not include site revisits over multiple years. These two characteristics of the program made it difficult to distinguish trends from natural variation driven by climate or other factors. For a trend at this regional scale to be evident, a longer time period would be required and/or site revisits.

In the second round of the program, the program now plans to monitor stage continuously, and include site revisits to increase power of trend detection. Each year, approximately 70 percent of the sites will be from a new sample draw while 30 percent will be revisits to previously sampled sites for trend estimates. Therefore during the 5-year period, they will have “condition” sites sampled 1 time each 5 years, and ‘trend’ sites sampled annually.

6.0 ABILITY TO DETECT LONG-TERM TRENDS

Environmental status and trends programs are intended to evaluate whether the aquatic resource conditions (i.e., status) are improving, declining, or maintaining current condition beyond the site scale (Larsen et al., 1995). A critical step in the development of a well-designed status and trends monitoring program is the evaluation of the components of variance of measured indicators. The relative magnitude of variance for an indicator affects uncertainty and statistical power to assess status, identify differences across strata, or detect trends over time.

Replicate (or revisit) measurements are needed to calculate residual error variance (σ^2_{res}). Residual error variance is the variance remaining after accounting for variance across sample sites, variance among years (across all sites collectively), and the year to year fluctuation among individual sampling sites. If multiple samples are collected throughout the year, the variance associated with within year sampling (across sites and years) must be accounted for in order to estimate residual variance.

Once these variance components have been estimated for a particular metric, it is possible to calculate indicators of precision such as signal to noise ratio (S:N = σ^2_{site}/ σ^2_{res}; Kauffman et al., 1999). Previous research indicates that S:N > 10 indicates negligible effects of noise, becoming minor through S:N of 6 and increasing to moderate as S:N reaches 2 (Kaufmann et al., 1999). As S:N approaches zero, noise becomes severely limiting and at 0, all variance is associated with noise. The signal to noise ratio provides an indication of the relative precision of a status indicator. The estimated components of variance can also be used to calculate the power to detect trends of a specified magnitude over a given length of time as a function of the number of sample sites and replicate measurements (Urquhart, 2012).

In the 2015 SAM PLES study, replicate samples were collected for sediment and water quality measures and periphyton. The level of replication was generally insufficient to accurately estimate the components of variance and associated signal to noise ratio for periphyton metrics, for which there were only 3 replicates. No replicate benthic invertebrate samples or physical habitat measurements were collected. Some metals and many of the organic contaminants were detected in less than 50 percent of the water and sediment samples analyzed. Many results below the detection limit compromise the ability to estimate components of variance.

6.1 Signal to Noise Ratio

The method used to estimate S:N requires the estimation of the components of variance of a particular measurement or indicator. The components of variance (σ^2) of the monthly water quality sampling can be described as follows:

$$\sigma^2_{Total} = \sigma^2_{Site} + \sigma^2_{Month} + \sigma^2_{Site:Month} + \sigma^2_{Residual}$$

Total variance	=	Population variance	+	Month variance	+	Interaction effects variance	+	Residual variance

Population variance describes the variance of a measurement made on a subsample of sites representing the population of interest during the year. In the absence of other sources of variance, these measurements would provide an estimate of status and associated variance for that year.

Month variance measures how much all sites (collectively) are higher or lower each month than the annual mean. This component of variance can be thought of as a common regional pattern of variance caused by regional-scale factors such as regional climate conditions and is sometimes referred to as temporal coherence (Larsen et al., 1995).

Site:Month interaction variance represents the month to month fluctuation among individual sampling sites. These fluctuations reflect responses to effects operating at the site level that are not already described by month effect described above. The Month and Site:Month variance can be separated by revisit samples collected at multiple sites each month.

Residual variance is the variance estimated from repeat sampling at multiple sites within a year. If residual variance of a particular measurement is relatively high, it may not be a useful indicator of status or trend. However, based on the information generated as part of the estimation of measurement variance, it may be possible to reevaluate and improve measurement methods. For example, residual variance might be reduced through sampling technology improvements, improved survey team training, or refinement of sampling protocols (Larsen et al., 1995).

Variance Components Analysis

Because the replicate design was not balanced (all sites were not revisited each month) we used a linear mixed-effects model to estimate the components of variance (Kincaid et al., 2004; Larsen et al., 2004). The model was of the form:

$$Y_{ijk} = \mu + S_i + T_j + ST_{ij} + I_{ijk}$$

where Y_{ijk} is the response for the kth visit to stream site *i* during month *j*, μ is the overall mean, S_i is the random effect due to stream site *i*, T_j is the random effect due to month *j*, ST_{ij} is the random effect due to the interaction of site *i* and month *j*, and I_{ijk} is the residual variation for the kth visit at site *i* during month *j*. Subscript *i* ranges from 1 to the number of stream sites in the survey, subscript *j* ranges from 1 to the number of months of data, and subscript *k* ranges from 0 to the number of site revisits during month *j* at site *i*.

The linear mixed effects model was fit using the lme4 R package (Bates et al., 2014) and took the form:

$$\text{lmer}(\text{RESULT} \sim 1\ +\ (1|\text{LOCATION\_ID})\ +\ (1|\text{MONTH})\ +\ (1|\text{LOCATION\_ID:MONTH}))$$

The variance components for sediment chemistry, which was sampled only once during the year (along with field replicates), was determined using a model that contained only the site term:

$$\text{lmer}(\text{RESULT} \sim 1\ +\ (1|\text{LOCATION\_ID})$$

If more than one year of data (ideally four or more years of data) including replicates were available, then it would be possible to estimate variance in sediment chemistry associated with Year and Site:Year interactions similar to the approach to monthly water quality variance components analysis described above:

$$\text{lmer}(\text{RESULT} \sim 1\ +\ (1|\text{LOCATION\_ID})\ +\ (1|\text{YEAR})\ +\ (1|\text{LOCATION\_ID:YEAR}))$$

Estimates of the signal to noise ratio were only calculated for the frequently detected water and sediment quality measurements (i.e., parameters that were detected in greater than 50 percent of samples). Because only one year of data have been collected so far, it is also not possible to estimate the year to year variance among individual sites or across all sites. Therefore, estimates of these variance components from other studies were used in a power analysis of trend detection for this program.

6.1.1 Water Quality

Estimated variance components of frequently detected water quality parameters measured using field instruments are presented in Table 37. All of the field measured water quality parameters in Table 37 had S:N ratios greater than 10. Dissolved oxygen and temperature had a relatively high month variance compared to variance across sites, which is consistent with the expected seasonal variation in these parameters. A relatively high month variance is likely an indication that monthly variation in a particular parameter might compromise trend detection power. Trend detection approaches that account for seasonality (i.e., systematic changes over the course of a year) may be needed to effectively detect trends in these parameters (Hirsch et al., 1991).

Table 37. Relative magnitude (percent) of four components of variance of frequently detected field-measured water quality parameters from the 2015 SAM PLES study.

PARAMETER	Site	Month	Site:Month	Residual	S:N RATIO
Conductivity	75.7%	7.2%	17.1%	0.0%	1612
Dissolved Oxygen	40.7%	32.8%	24.9%	1.5%	26
Temperature	9.2%	66.5%	24.1%	0.2%	54
pH	48.3%	12.8%	36.6%	2.4%	20

Estimated variance components of frequently detected water quality parameters measured in the laboratory are presented in Table 38. The S:N ratio of all but one of these parameters in Table 38 had ratios greater than 10. Fecal coliform bacteria measurements had a S:N ratio of 8.7.

Table 38. Relative magnitude (percent) of four components of variance of frequently detected laboratory-measured water quality parameters from the 2015 SAM PLES study.

PARAMETER	Site	Month	Site:Month	Residual	S:N RATIO
Chloride, Total	75.9%	3.3%	20.8%	0.1%	1380
DOC	50.4%	16.9%	32.3%	0.4%	139
Fecal coliform	16.8%	3.1%	78.1%	2%	8.7
Hardness, Total	75.2%	8.7%	16.0%	0.1%	651
Turbidity	7.8%	2.2%	89.9%	0.1%	108
Nitrite-Nitrate, Total	82.3%	2.2%	15.4%	0.1%	733
Total Phosphorus	25.8%	1.1%	72.6%	0.4%	64.1
Total Nitrogen	81.6%	3.8%	14.5%	0.1%	590

DOC = Dissolved Organic Carbon

Total and dissolved arsenic had S:N ratios greater than 10, which indicates negligible noise (Table 39). The remaining frequently detected metals (total and dissolved copper and total chromium) had signal to noise ratios between about 2 and 10 ranging from moderate to minor noise.

Table 39. Relative magnitude (percent) of four components of variance of select water quality metals data from the 2015 SAM PLES study.

PARAMETER	Site	Month	Site:Month	Residual	S:N RATIO
Arsenic, total	59.9%	1.5%	38.3%	0.3%	232
Arsenic, dissolved	76.3%	4.1%	18.4%	1%	69.8
Chromium, total	10.7%	2.7%	84.6%	2%	5.5
Copper, total	9.3%	7.4%	82.3%	1%	9.3
Copper, dissolved	23.4%	15.6%	48.9%	12%	1.9

A number of parameters had a relative Site:Month interaction variance greater than 50 percent. These parameters included turbidity, fecal coliform, total phosphorus, and total and dissolved chromium and copper. High Site:Month interaction variance is likely due to the influence of variation in flow on concentration. Such a relationship would vary from site to site over the course of the year. In such cases, improved trend detection power can be achieved by the use of statistical procedures that account for the relationship between concentration and flow (Hirsch et al., 1991; Vecchia, 2003; Vecchia et al., 2008). These techniques require a long and relatively complete record of daily discharge over the period being analyzed (Hirsch et al., 2010).

6.1.2 Sediment Quality

Estimated variance components of frequently detected sediment quality parameters are presented in Table 40. With the exception of sediment copper and zinc, all of the sediment quality parameters had S:N ratios greater than 10.

Table 40. Relative magnitude (percent) of four components of variance of select sediment metals and organic contaminant data from the 2015 SAM PLES study.

PARAMETER	Site	Residual	S:N RATIO
Arsenic	98%	2%	53.7
Cadmium	99%	1%	76.6
Chromium	93%	7%	12.5
Copper	69%	31%	2.2
Dichlobenil	99%	1%	88.9
Lead	93%	7%	13.4
Retene	96%	4%	25.2
Total PBDE	97%	3%	31.0
Total PCB	100%	0.0%	5450
Zinc	83%	17%	5.0

If more than one year of data (ideally four or more years of data) including replicates were available, then it would be possible to estimate variance in sediment chemistry associated with Year and Site:Year interactions similar to the approach to monthly water quality variance components analysis described above:

lmer(RESULT ~ 1 + (1|LOCATION_ID) + (1|YEAR) + (1|LOCATION_ID:YEAR))

6.1.3 Other Sources of Signal to Noise Estimates

Measures of S:N within a survey are useful for identifying metrics with the greatest potential for discriminating among sites and detecting trends, but S:N may not be useful for comparison to surveys in other regions because the absolute range of a metric may not be the same among regions. Nonetheless, S:N has been used extensively to evaluate the relative merit of many of the metrics measured in the 2015 SAM PLES study, including benthic invertebrate metrics, WH water quality measurements, and WH habitat metrics (Stillwater Sciences, 2015). Stillwater Sciences (2015) compiled S:N ratios from a number of studies and assigned a score or grade based on the letter grades provided in Merritt and Hartman (2012): A (>10), B (5-10), C (2-5), D (1-2), F (<1). Estimates of S:N for the 0-100 scale B-IBI and for WH habitat metrics based on four years of sampling are also available from King County (2015b). The S:N estimate for B-IBI was 16.1, which would be graded A.

S:N values from other studies provide estimates of the expected precision of metrics selected for use in a particular status and trends study. However, estimates of trend detection power based on specific study designs may be of more direct benefit to evaluating the utility of particular metrics. Potential approaches to evaluating the trend detection power of various sampling approaches based on available Puget Lowland data sets are described in the next section.

6.2 Trend Detection Power

The design of a status and trends monitoring program begins with an evaluation of the spatial and temporal components of the design as well as the selection of indicators to measure to meet the goals and objectives.[31] As part of this process, the various designs under consideration were evaluated for their ability to detect the amount of change of interest. This evaluation is termed trend detection power.

The power to detect trends in the data will depend on the study design, including the trend model selected, the statistical variance of the indicators sampled, the number of sites, the frequency that sites are sampled, and the intended timeframe expected to yield a trend (Larsen et al., 2004). Because the SAM PLES study was for a single year, estimates of variance were obtained from similar surveys with multiple years of data as described below. Data sets with multiple years of data provide essential information regarding inter-annual variability that was not available from the single initial year of sampling conducted for the SAM PLES study. While use of data from these similar surveys introduces uncertainty into our estimate of trend detection power, this approach provides a reasonable first approximation of trend detection power of potential monitoring designs.

Although evaluating a large number of monitoring network designs is beyond the scope of this report, two approaches to evaluating trend detection power of regional monitoring programs are demonstrated. The methods used to evaluate the potential power of various sampling designs are described below.

The first approach to evaluation of trend detection power allows for the evaluation of various once a year revisit and/or panel designs that are the foundation of U.S. EPA and Ecology's status and trends monitoring programs. Regional trends (i.e., mean trend across all sites) can be evaluated by using a linear mixed effects model of the form:

$$Y_{ijk} = \mu + S_i + T_j + \beta_j + ST_{ij} + I_{ijk}$$

This model is similar to the variance components model above, but this model includes the parameter (β_j) that represents the average slope or trend over all sampling sites (Urquhart et al., 1998; Anlauf et al., 2011; Urquhart, 2012). The remaining parameters are as follows. Y_{ijk} is the response for the kth visit to stream site *i* during year *j*, μ is the overall mean, S_i is the random effect due to stream site *i*, T_j is the random effect due to year *j*, ST_{ij} is the random effect due to the interaction of site *i* and year *j*, and I_{ijk} is the residual variation for the kth visit at site *i* during year *j*. Subscript *i* ranges from 1 to the number of stream sites in the survey, subscript *j* ranges from 1 to the number of years of data, and subscript *k* ranges from 0 to the number of site revisits during year *j* at site *i*.

[31] Pacific Northwest Aquatic Monitoring Partnership Monitoring Advisor: http://www.monitoringadvisor.org/design/

The trend detection power calculations were performed for B-IBI scores using variance estimates available from a four year status and trend study conducted in WRIA 8 (King County, 2015b). The calculation method was based on the approach described by Urquhart (2012) using functions written in R provided by Tom Kincaid (personal communication, U.S. EPA Corvallis, 8 January 2015). In addition to the survey design and parameter variance components, a site correlation across years of 1.0 and a year autocorrelation of 0.0 was used (these are the default parameters).

A second approach to estimating trend detection power was developed for evaluation of sub-annual (i.e., seasonal or monthly data) sampling. This approach is similar to that used by Jabusch et al. (2016). Jabusch et al. (2016) evaluated the power of the San Francisco Estuary Institute program to detect regional nutrient trends. The method relies on water quality monitoring data from a long-term data set collected from 25 stations in King County.

The method is a simulation approach that randomly samples the long-term monthly fecal coliform concentrations at 25 King County stations. The sampling approach is capable of simulating any length of time. As an example, 10 year-long monthly data sets were simulated with the same fixed trends imposed for each of the 25 stations. A power curve was generated by conducting many simulations (typically 1,000 simulations) for each fixed trend and evaluating the regional trend in annual geometric mean fecal coliform concentration for statistical significance (using a p-value of 0.05). The power curve represents the power to detect a regional change in fecal coliform concentration based on sampling a 25 station network at a monthly sampling frequency. Although linear mixed-effects models can be used to detect trends and evaluate trend detection power in long-term surveys that include sub-annual sampling, more commonly a non-parametric regional trend test (regional Kendall test) is used, particularly for the detection of water quality trends (Helsel and Frans, 2006). The power analysis was conducted using R and the regional Kendall trend (rkt) package (Marchetto et al., 2013; Marchetto, 2017).

6.2.1 B-IBI

Data from a four year status and trends study conducted in WRIA 8 were used to represent expected variance (site, year, site-year, and residual variance) for B-IBI scores (King County, 2015b). The power analysis calculations assumed annual sampling at 50 sites (Design 2 in Table 41). The power to detect a 1, 2 or 3 percent change per year over a 20-year period in the overall mean B-IBI score of 40 is presented in Figure 92. A mean value of 40 was similar to the estimated mean B-IBI score within UGAs in the SAM PLES study. The lower the mean value, the lower the trend detection power for a given expected percent change. For the smallest incremental change in B-IBI scores (1 percent per year) a power of 0.8 (a typical minimum target; Lenth, 2001) was not reached at the end of a

Table 41. Schematic of four revisit panel survey designs.

		Time periods (years)								
Panel	Size	1	2	3	4	5	6	7	8	…
		Design 1: Always revisit 25 sites								
1	25	X	X	X	X	X	X	X	X	…
		Design 2: Always revisit 50 sites								
1	50	X	X	X	X	X	X	X	X	…
		Design 3: Always revisit 100 sites								
1	100	X	X	X	X	X	X	X	X	…
		Design 4: Augmented serially alternating (200 sites per cycle)								
1	40	X				X				
2	40		X				X			
3	40			X				X		
4	40				X				X	…
Common	10	X	X	X	X	X	X	X	X	…
		Design 5: Augmented serially alternating (56 sites per cycle)								
1	12	X				X				
2	12		X				X			
3	12			X				X		
4	12				X				X	…
Common	2	X	X	X	X	X	X	X	X	…

Note: Adapted from Urquhart (2012)

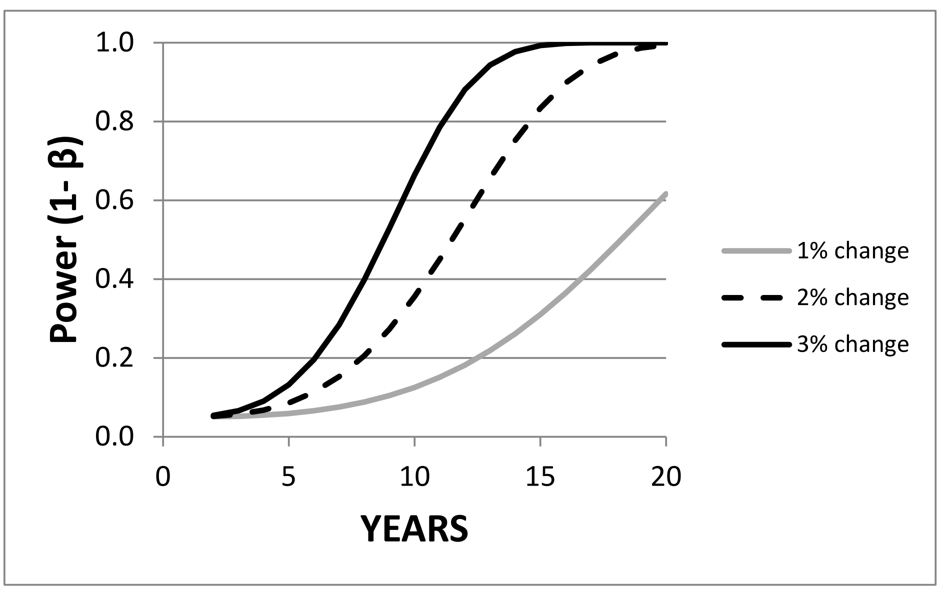

Figure 92. Plot illustrating the power to detect a 1, 2, or 3 percent change (average trend) in B-IBI scores over a 20-year period based on an annual repeat visit sampling design of 50 sites sampled every year (Design 2 in Table 41).

Note: Power analysis based on variance estimates from a four year status and trends monitoring study conducted in WRIA 8 (King County, 2015b) assuming an initial mean 0-100 scale B-IBI score of 40.

20-year sampling period.[32] Trend detection power reaches 0.8 in about 12 to 13 years for a rate of change of 2 percent and in about 9 to 10 years for a 3 percent annual rate of change.

The estimates of when a trend detection power of 0.8 would be reached would extend out further in time if all 50 sites were sampled every second year, every third year, etc. If sampling were conducted only once every 5 years as may have been envisioned by the original design, the numbers on the x-axis would be replaced with 25, 50, 75 and 100 years.

Other factors, such as number of sites, will also have an effect on trend detection power. For example, there would be a relatively small change in the power to detect a 2 percent annual change if the number of sites sampled each year was decreased to 25 or increased to 100 (Figure 93; Designs 1 and 3 in Table 41). Although it might be tempting to decrease the number of sites sampled each year due to the expected small effect on trend detection power, such a reduction would have a potentially larger negative effect on the confidence (i.e., precision) in a status assessment based on fewer sites.

Because an assessment of status is essentially a proportion (e.g., proportion of stream miles in poor B-IBI condition), the estimate of precision can be determined from the number of sites and the expected proportion.[33] The estimates of precision in Table 42 provide an indication of the tradeoff between precision estimates of status and the number of sites sampled. In summary, The power to detect a trend in B-IBI scores may not be significantly reduced by sampling 25 rather than 50 sites, but the precision of a status assessment (with 95 percent confidence) would decrease from 5 to 6 percent.

Various rotating panel designs could also be considered for the SAM PLES effort, including designs that include some fixed sites sampled each year and other sites that rotate over a fixed cycle known as an augmented serially alternating design. As an example, the B-IBI trend detection power of a design that includes 10 fixed sites and 40 rotating sites that cycle every 4 years (Design 4 in Table 41) was evaluated using the estimated variance components from the WRIA 8 status and trends study (King County, 2015b). Figure 94 illustrates that there would be little expected sacrifice in trend detection power with an substantial increase in the robustness of a 4 year status assessment with a rotating panel design equivalent to sampling the same 50 sites every year. The increase in the precision of the status assessment is due to a change from sampling the same 50 sites each year to 200 sites sampled every 4 years.

[32] Note that a power of 0.8 (and a significance level of 0.05) are presumed here, although the choice of target power and significance level is typically part of the design process. For example, a selection of a significance level of 0.1 may provide a reasonable tradeoff of Type I and Type II errors, while reducing the costs of a particular monitoring program (Levine et al., 2014).
[33] Aquatic resource monitoring FAQs – Survey Design: https://archive.epa.gov/nheerl/arm/web/html/surdesignfaqs.html#manysamples

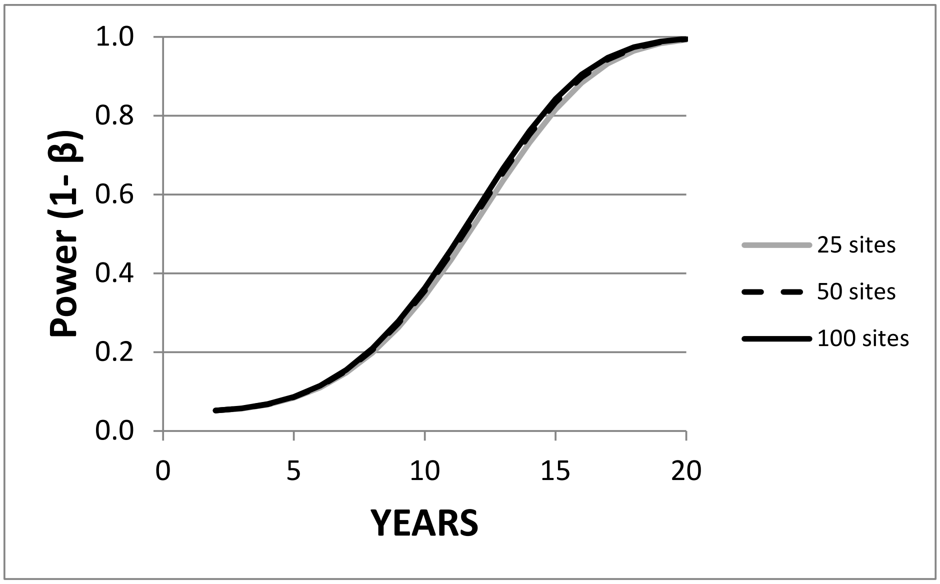

Figure 93. Plot illustrating the power to detect a 2 percent change (average trend) in B-IBI scores over a 20-year period based on an annual repeat visit sampling design of 25, 50, and 100 sites.

Note: Power analysis based on variance estimates from a four year status and trends monitoring study conducted in WRIA 8 (King County, 2015b) assuming an initial mean 0-100 scale B-IBI score of 40.

Table 42. Confidence limits (i.e., precision) of an estimated proportion (20 and 50% in good/poor condition) based on a simple random survey.

Assumed Percent in Good (or Poor) Condition	Precision with 90% Confidence for alternative sample sizes			Precision with 95% Confidence for alternative sample sizes		
	25	50	100	25	50	100
20%	±13	±9	±7	±16	±11	±8
50%	±16	±12	±8	±20	±14	±10

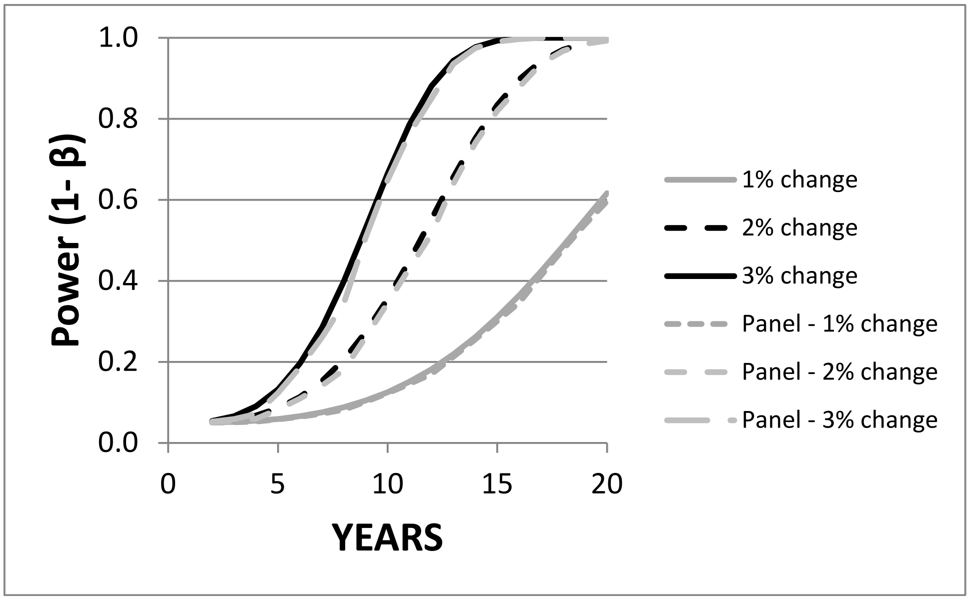

Figure 94. Plot illustrating the power to detect a 1, 2, and 3 percent change (average trend) in B-IBI scores over a 20-year period based on an annual repeat visit sampling design of 50 sites and a panel design of 10 fixed sites and 40 sites rotating over a 4-year period for 20 years.

Note: Power analysis based on variance estimates from a four year status and trends monitoring study conducted in WRIA 8 (King County, 2015b assuming an initial mean 0-100 scale B-IBI score of 40.

Figure 95 shows that a more substantial decrease in power would be expected if the rotating panel was reduced to 2 sites visited each year with 12 rotating panel sites every 4 years (the revisits, i.e., site replicates, within a year were also reduced from 5 to 2; a total of 56 sites visited over 4 years; Design 5 in Table 41). Such a reduction in sampling effort, from 50 to 14 sites per year with no substantial sacrifice in the ability to assess the status over 50 sites at the end of each sampling cycle, resulted in a moderate reduction in expected trend detection power.

Although not analyzed here, revisit designs that only visit the same sites only a few times and then never again could also be considered. A design of this type might better accommodate difficulties in maintaining land owner access permission over long periods of time. A design of this type also might more easily account for changes in the population composition (e.g., changes in UGA designations) (McDonald, 2003).

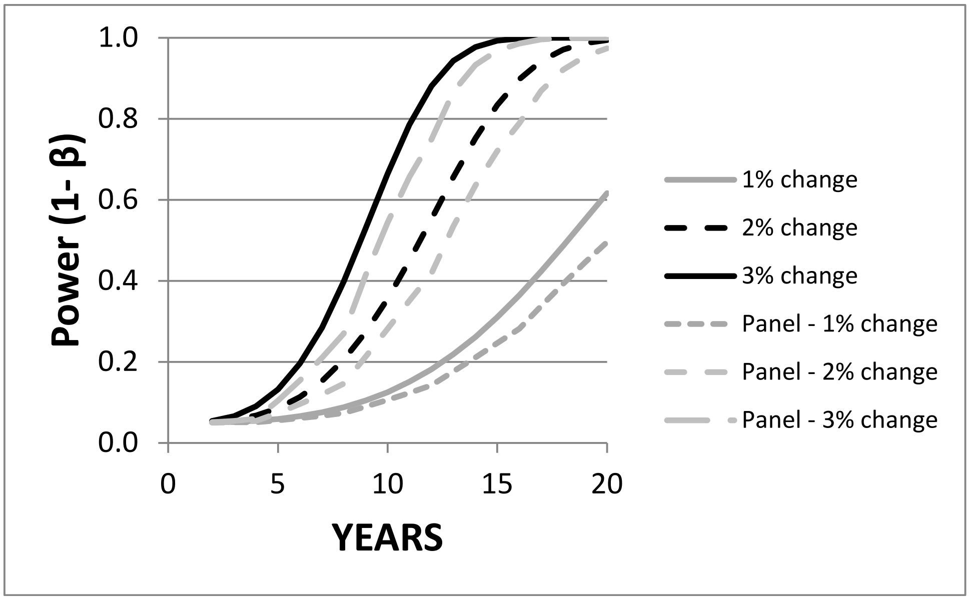

Figure 95. Plot illustrating the power to detect a 1, 2, and 3 percent change (average trend) in B-IBI scores over a 20-year period based on an annual repeat visit sampling design of 50 sites and a panel design of 2 fixed sites and 12 sites rotating over a 4-year period for 20 years.

Note: Power analysis based on variance estimates from a four year status and trends monitoring study conducted in WRIA 8 (King County, 2015b) assuming an initial mean 0-100 scale B-IBI score of 40.

The desired objectives of the SAM PLES program going forward and available resources will provide the information needed to compare the tradeoffs of specific designs. Desired monitoring objectives include specification of acceptable power, including acceptable probabilities of Type I and Type II errors, as well as the desired amount of change that should be detectable by this program. [34]

6.2.2 Water Quality

An example approach for monthly or sub-annual water quality sampling is provided below using long-term water quality data from 25 sites in King County to represent potential water quality variability across sites, months, and years. Evaluating the trend detection power of a status and trends monitoring program that includes sub-annual (e.g., seasonal or monthly) sampling is more complicated than the evaluation of annual repeat visit or panel designs. Although linear mixed-effects models can be used to detect trends and evaluate trend detection power in long-term surveys that include sub-annual sampling, more commonly a non-parametric regional trend test (regional Kendall test) is used, particularly for the detection of water quality trends (Helsel and Frans, 2006). A method to evaluate the trend detection power of a monthly water quality sampling effort is illustrated in this section using fecal coliform data from a long-term data set collected from 25 stream stations in King County.

[34] Type I and Type II errors are falsely identifying a trend when there isn't one (Type I error) and falsely rejecting a trend when there is one (Type II error).

The method is similar to that used by Jabusch et al. (2016) to evaluate the power of the San Francisco Estuary program to detect regional nutrient trends. The method is a simulation approach that relies on randomly sampling the long-term monthly fecal coliform concentrations at each King County site. The sampling approach is capable of simulating any length of time. As an example, 10 year-long monthly data sets were simulated with fixed trends imposed for each of the 25 stations. By generating many simulations (typically 1,000 simulations) for each fixed trend and evaluating the trend in annual geometric mean fecal coliform concentration, a power curve can be constructed for the power to detect a regional change in fecal coliform concentration based on sampling a 25 station network at a monthly sampling frequency.

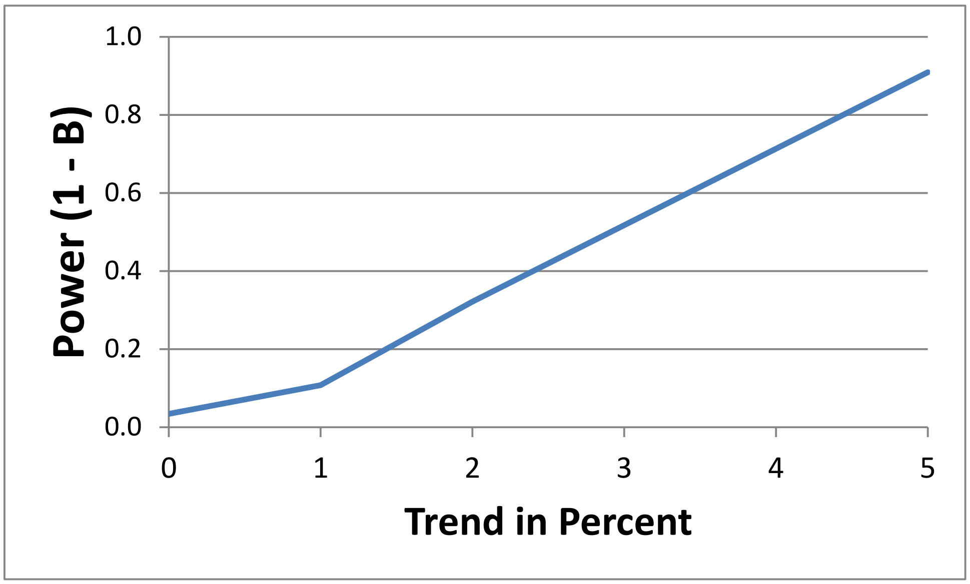

Figure 96. Plot illustrating the power to detect a 1 to 5 percent per year change (average trend) in annual geometric mean fecal coliform concentrations measured at 25 stations over a 10-year period based on data from King County's long-term monitoring program.

Note: Power analysis based on variance estimates from a 25 King County long-term status and trends monitoring stations and a Type I error rate (alpha) of 0.05.

This power analysis example indicates that regional trend detection power of 0.8 will likely only be achieved with a network of at least 25 sites sampled monthly every year for a period of 10 years if (and only if) the regional trend is greater than 4 percent per year.

The simulation approach also provides an estimate of at-site power to detect trends of the specified magnitudes. Interestingly, the simulations indicate that there is relatively low power to detect a trend at any particular site for even the highest simulated trend of 5 percent change per year (estimated power between 0.09 and 0.34, depending on the station). It is the consistent increase (albeit noisy) at all sites that would result in the power to detect regional trends.

In general, the approach illustrated above (or some modification of this approach) could be used to evaluate the power of other sub-annual sampling designs, including other monitoring parameters. This would include designs which focus on sampling during a specific season or for storm and non-storm sampling as the King County long-term program has included storm and non-storm sampling for several years at these 25 long-term monitoring locations. The use of the King County data set, although likely fairly representative of regional stream water quality, introduces an additional degree of uncertainty to any power analysis based on these data.

7.0 SAM PLES TREND PROGRAM RECOMMENDATIONS

We believe the first round of monitoring was a success and established the foundation of a regional lowland streams monitoring program. The existing program achieved several goals: (1) established the status of small streams in the Puget Lowlands, (2) presented stream conditions in two categories of urbanization across sampled watersheds, and (3) allowed for the identification of natural and human factors leading to poor biological condition.

In this section we discuss recommendations under two scenarios for SAM streams trend program, and a short list of considerations for shorter term SAM streams study ideas that merit exploration, but not as part of the core program. Some recommendations apply to either trend program scenario.

The first scenario assumes that the SAM PLES is largely kept the same and uses the within and outside UGA study design used in 2015. We use the experiences and findings from the 2015 SAM PLES monitoring effort, and information learned during the review of other programs conducting similar regional sampling to address Question 5.

- **Q5**: What water, sediment, biological and habitat parameters would be carried forward for status and trend assessment of SAM PLES monitoring in the future, and at what timing and frequency?
- And, how can we capture a more stormwater focused story?

With some small changes to the existing program, such as dropping monthly water quality sampling (and the emphasis on the WQI), and adding continuous monitoring of stage (water level), the current program will be able to address the study questions, albeit with less direct focus on stormwater impacts. Repeating more or less the same study a second time may be important due to the drought year, although there is also a potential to favor the initial design as a result of sunk cost bias. Sunk cost bias is the difficulty of objectively weighing the value of a future investment against costs that have already been incurred and cannot be recovered. We appreciate the desire to evaluate trends as soon as is possible, but encourage decisions regarding changes to the initial design be made assuming that a well-designed long-term monitoring effort will be needed to detect relevant environmental changes.

In the second scenario we propose a minor re-design of the strata from the binary within/outside of UGA to a gradient of urbanization conditions where stormwater management occurs. Under this second scenario we also propose modifications to the program's questions and design that we believe will improve the program's ability to meet its trend objectives of tracking if stream quality is changing over time. Some of the recommendations presented here would need further work in advance of the next round of sampling.

Some good recommendations for monitoring streams that could improve our understanding of impacts from stormwater can be accommodated in "spin-off" or "short term" studies at the SAM streams sites. We've made a list of these ideas in Section 7.3.

The following sections describe results-based recommendations for improving what we did this round, based on the study questions from the QAPP (Ecology, 2014; Appendix A).

- What are the status and trends of instream water quality, biological, and habitat conditions for 1st, 2nd, and 3rd order (small) streams in Puget Lowlands?
- What are the status and trends of the water quality, biological, and habitat conditions for 1st, 2nd, and 3rd order (small) streams in Puget Lowlands, both inside and outside of City/UGAs?

For this report, we refined the first 3 of the 5 questions presented previously to gain more specificity on how to present the data as follows:

Q1: What percent of streams meet biological, water, and sediment quality standards for beneficial uses within and outside urban growth areas (UGAs)?

Q2: What natural variables correlate with the status of streams within and outside the UGA?

Q3: What human variables correlate with the status of streams within and outside the UGA?

7.1 Minimum changes scenario

In making recommendations for the minimum changes scenario, we focused on what worked well and lessons learned from what did not work well.

7.1.1 Recommendations under a minimum changes scenario

1. Modify questions 1, 2, and 3 to focus status assessments on reference or least disturbed conditions and specific biological-effects thresholds rather than focus on water and sediment quality standards and beneficial uses.
2. Reevaluate existing site list to eliminate nested basins and evaluate the watershed characteristics above each point to make sure they represent the strata label.[35] Reassign some sites located at the outer edges of the UGA to the outside UGA stratum that the sites actually represent with their watershed drainage areas.

[35] The recommendation to eliminate nested basins is based on analysis of monthly mean flow (Konrad and Voss, 2012) and SAM PLES monthly water quality data (Sheibley, in preparation) collected from nested Puget Sound basins. The correlation of mean monthly data between nested sites increases as the sites share an increasing percentage of upstream drainage area. It should be noted that eliminating nested basins may preclude population estimates based on a survey design as traditional survey designs require that all streams in the target population have a positive chance of being selected in the sample.

3. Define how to address UGA boundaries changing over time, both the process to adaptively balance the number of streams in each strata and how to approach the data analysis.
4. Better define data needs for human and natural variables that are not directly collected as part of this monitoring effort. For example, the land use, precipitation record, and beneficial uses.
5. Drop monthly water quality sampling and the WQI. Keep limited sampling of water quality parameters with a moderate or better signal to noise ratio and established relationship to stormwater discharges or impacts. Consider an index period (more detail provided in Section 7.2.1.4 below) for water quality sampling.
6. Add an emphasis to assess certain stressors that are more related to stormwater management, given the acknowledgement that broadly observed changes over time will reflect multiple stressors. For example, much of stormwater management is flow control, yet there are no hydrological study questions and only limited flow data (monthly instantaneous values) were gathered in the current design.

7.2 Modified design scenario

This second scenario presents our preferred and scientific recommendations to change the study questions to better align with trend program questions in an adaptive management framework. This section also presents a few options for consideration on how to do this and meet scientific rigor and power of the design.

The main premise for the following recommendations and options are that the Puget Lowland land use exhibits a gradient of development conditions, and we acknowledge that a single study design cannot separate stormwater specific management from other human activities (e.g., riparian clearing/restoration, habitat restoration, etc.) occurring on the landscape. Therefore the design must capture and track to the extent affordable and reasonable, factors that influence non-point and point source stormwater discharge impacts to stream condition; namely hydrologic, chemical, and benthic habitat. The primary factors are understood to be urbanization and agriculture in the Puget Lowlands. Given stormwater permittees are funding this study design as part of the permit, we understand the focus to be urbanization. Below are the design considerations for looking at an urbanization gradient, since that answers slightly different goals and questions than was addressed in this round of sampling.

The remainder of this section will focus on specific recommendations for the next round of SAM PLES monitoring. These recommendations will focus on sample design, sample frequency, and measurements to add, delete, or expand.

7.2.1 Study Design: Sampling frame and site selection

7.2.1.1 Sampling Frame

1. We recommend that the SAM streams status and trends program continue using a ***regional scale probabilistic study design*** in order to make inferences across the Puget Lowland ecoregion. Targeted and small scale probabilistic studies produce information to address local problem or effectiveness questions but are not representative of conditions at the regional scale.

2. ***Target population*** recommend a changed from inside and outside the UGA as follows:
    a. Eliminate nested basins to ensure independence of observations at sampled sites.
    b. Sampled sites represent watersheds above the site that are small to medium (2.5-50 km^2 area) in size to ensure sampling of wadeable perennial streams where stormwater impacts are realized and monitorable.
    c. Use gradient of development conditions that reflects landscape and provides information for stormwater managers.
        - Decide on which factors to stratify sites across an urbanization or development gradient (*e.g.,* percent development, percent impervious surface, watershed canopy).
        - Consult a statistician to discuss how to distribute sites across the gradient; for example evenly in different categories, more for a certain density, or randomly across the gradient.
        - Consult a statistician to discuss how to assign weights and adjust weights for stream reaches versus points.
        - Help decision makers understand the statistical implications of each site selection approach (e.g., equal or unequal probability sampling) described above.

3. ***Reference Sites***. In order to determine status, develop thresholds, and conduct relative risk/attributable risk analysis, it is recommended that reference sites be sampled as part of the small streams program. These reference sites would be sampled for the same parameters as the probabilistic sites to establish least impacted conditions, and account for natural and climatic variability.
    a. Develop a consistent set of SAM reference sites (approximately 15% of total number of sites). Start with the existing 16 Puget Sound reference sites currently used by Ecology's ambient monitoring program that match SAM target site criteria.
        - Ensure the data collected and procedures used by Ecology matches the data collected for SAM PLES. If there is divergence, collaborate with Ecology to establish consistent datasets.
        - Reference sites would represent a similar watershed size within the Puget Lowland ecoregion and meet other SAM site selection criteria.

   b. Reference sites are recommended to have low percent developed land cover in their contributing watersheds and high percent forest cover. The details would be provided in the next QAPP.

4. ***Analysis of land cover and other GIS metrics*** is recommended to be repeated every 5 years to look at changes over time.

7.2.1.2 Site Selection

5. In the site selection, criteria clearly articulate the reach associated with the master sample point.

7.2.1.3 Frequency for Sampling

6. Additional work is needed to finalize the details of the next round of the SAM PLES program. We recommend the SAM streams team members meet with other SAM nearshore scientists to take the next steps in shaping the final frequency details with consideration given to a gradient of urban conditions and parameters to track over time.

7. Sampling frequency is not tied to the permit cycle but is defined with consideration given to the parameters being sampled, signal to noise, or other utility concerns such as cost.

8. We recommend that trend is the primary focus for SAM's stream monitoring, but not exclusively and periodic larger conditions assessments are made to ensure representativeness of the sites and relevant parameters are being assessed.
   a. Repeat visits to a core set of randomly selected sites we recommend 100 to 150, but need a minimum of 75 sites each cycle. This list will include all sites from the 2015 list that meet site selection criteria.
   a. In addition to the repeat visit sites, single visits to a number of randomly selected new sites will ensure regional representativeness of the sample. We recommend 50 to 100 but need a minimum of 25 each cycle.

9. We recommend a rotating panel design to ease implementation of the program without jeopardizing statistical analysis. The sample will repeat once 250 total sites have been visited.

7.2.1.4 Parameters

10. **Add** continuous monitoring of stage at all sites.

11. Consistent with our findings from the 2015 SAM PLES sampling and analysis we recommend, that these measures be **continued at each site**:
    a. Sample in the summer for watershed health, includes:
       - Watershed Health parameters per Ecology protocol – which includes, benthic invertebrates, periphyton, habitat, and water quality.

b. Sieved sediment (<2mm only) per USGS protocol for total organic carbon, grain size, nutrients, PAHs, phthalates, select metals, and pesticide analyte list that includes dichlobenil. Keep grain size for sieved portion.
   - For PAHs, use method that can achieve lower detection limits (selective ion monitoring or SIM).

12. **Drop** monthly water quality sampling and WQI calculation

13. **Consider** creating an index period(s) for water quality (timeframe(s) to be determined) for the following variables
    - Fecal coliform
    - Nutrients
    - Conventionals (temperature, dissolved oxygen, conductance, and pH)
    - total suspended solids and turbidity
    - Metals (lead, zinc, copper, arsenic, cadmium, chromium)
    - PAHs
    - Pesticides

14. **Consider** continuing to measure for PBDEs in sediment.

7.3 Short-term Study Ideas to Leverage Trend Program

Our goal with this section is to capture additional work that could be done along with SAM streams, but as shorter term studies or studies at a subset of sites. We recommend that initially the focus be on defining the core trend streams program and then consider some of these ideas when funding or capacity allows. This following list is not in priority order.

1. Consider targeting storm and base flow sampling to better capture stormwater-related variation in the high priority parameters identified above.
2. Revisit some or all of the parameters that were dropped once every 10 to 15 years at a subset of sites to reevaluate their detection frequency and detected concentration levels.
3. Consider sampling in special studies:
   a. PCBs in sediment
   b. Surface water pesticides detected in the recent regional USGS PNWQA study.
   c. Sediment pesticides including pyrethroids (e.g., bifenthrin) and phenyl pyrazole (e.g., fipronil) insecticides identified as potential stressors in other regional studies.
   d. Special sampling approaches to improve our understanding of their occurrence and importance to biological impacts.

4. As a special study, analyze a set of hand-picked sites to represent a range of development and located at existing flow gages. Such an effort should build off of suggestions from previous gage network analyses (Konrad and Voss, 2012; Konrad and Sevier, 2014).

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Appendix A: Detectioin Frequency of Water and Sediment Chemistry Parameters

Appendix A is also provided as a Microsoft Excel file available for download at: http://your.kingcounty.gov/dnrp/library/2018/kcr2968/kcr2968-digital-appendices.zip

Table A1: Water Chemistry

Table A2: Sediment Chemistry-Polycyclic Aromatic Hydrocarbons (PAHs), Phthalates, and Pesticides

Table A3: Sediment Chemistry-Polychlorinated Biphenyl and Polybrominated Diphenyl Ether Congeners

Table A1: Detection frequency of water chemistry parameters (see **Section 2.7.1** for Case definitions).

Parameter	Fraction Analyzed	Case	n	% detected	% not-detected
1-Methylnaphthalene (ug/L)	Total	C	479	0%	100%
2-Methylnaphthalene (ug/L)	Total	C	740	0%	100%
Acenaphthene (ug/L)	Total	C	740	0%	100%
Acenaphthylene (ug/L)	Total	C	740	0%	100%
Ammonia (mg/L)	Total	B	1006	47%	53%
Anthracene (ug/L)	Total	C	741	0%	100%
Arsenic (ug/L)	Dissolved	A	743	88%	12%
Arsenic (ug/L)	Total	A	744	90%	10%
Benz(a)anthracene (ug/L)	Total	C	741	0%	100%
Benzo(a)pyrene (ug/L)	Total	C	741	0%	100%
Benzo(b)fluoranthene (ug/L)	Total	C	480	0%	100%
Benzo(g,h,i)perylene (ug/L)	Total	C	741	1%	99%
Benzo(k)fluoranthene (ug/L)	Total	C	480	0%	100%
Benzofluoranthenes, Total (ug/L)	Total	C	261	1%	99%
Cadmium (ug/L)	Dissolved	C	743	1%	99%
Cadmium (ug/L)	Total	C	744	1%	99%
Calcium (ug/L)	Total	A	261	100%	0%
Carbazole (ug/L)	Total	C	479	0%	100%
Chloride (mg/L)	Total	A	1044	100%	0%
Chromium (ug/L)	Dissolved	B	743	47%	53%
Chromium (ug/L)	Total	A	744	63%	37%
Chrysene (ug/L)	Total	C	741	0%	100%
Copper (ug/L)	Dissolved	A	743	72%	28%
Copper (ug/L)	Total	A	744	79%	21%
Dibenzo(a,h)anthracene (ug/L)	Total	C	741	0%	100%
Dibenzofuran (ug/L)	Total	C	479	0%	100%
Dissolved Organic Carbon (mg/L)	Dissolved	A	744	90%	10%
Fecal coliform (cfu/100mL)	Total	A	990	93%	7%
Fluoranthene (ug/L)	Total	C	741	2%	98%
Fluorene (ug/L)	Total	C	740	0%	100%
Hardness as CaCO3 (mg/L)	Total	A	1004	100%	0%
Indeno(1,2,3-cd)pyrene (ug/L)	Total	C	741	1%	99%
Lead (ug/L)	Dissolved	C	743	7%	93%
Lead (ug/L)	Total	B	744	35%	65%
Magnesium (ug/L)	Total	A	261	100%	0%
Naphthalene (ug/L)	Total	B	740	24%	76%
Nitrite-Nitrate (mg/L)	Total	A	1006	100%	0%
Ortho-phosphate (mg/L)	Dissolved	A	1005	99%	1%
PCN-002 (ug/L)	Total	C	479	0%	100%

Parameter	Fraction Analyzed	Case	n	% detected	% not-detected
Phenanthrene (ug/L)	Total	C	741	1%	99%
Pyrene (ug/L)	Total	C	741	0%	100%
Retene (ug/L)	Total	C	480	3%	97%
Silver (ug/L)	Dissolved	C	743	0%	100%
Silver (ug/L)	Total	C	744	0%	100%
Total Benzofluoranthenes (ug/L)	Total	C	741	0%	100%
Total Nitrogen (mg/L)	Total	A	1044	100%	0%
Total Phosphorus (mg/L)	Total	A	1042	96%	4%
Total Suspended Solids (mg/L)	Total	A	1040	87%	13%
Zinc (ug/L)	Dissolved	B	743	22%	78%
Zinc (ug/L)	Total	B	744	34%	66%

Table A2: Detection frequency of sediment metals, PAHs, phthalates, and pesticides (see **Section 2.7.1** for Case definitions).

Parameter	Case	n	% detected	% not-detected
1-Methylnaphthalene (ug/Kg)	C	111	1%	99%
2-Methylnaphthalene (ug/Kg)	C	111	1%	99%
2,4-D (mg/Kg)	C	111	0%	100%
Acenaphthene (ug/Kg)	C	111	2%	98%
Acenaphthylene (ug/Kg)	C	111	1%	99%
Anthracene (ug/Kg)	C	111	12%	88%
Arsenic (mg/Kg)	A	111	100%	0%
Benz(a)anthracene (ug/Kg)	B	111	21%	79%
Benzo(a)pyrene (ug/Kg)	B	111	27%	73%
Benzo(b)fluoranthene (ug/Kg)	C	70	10%	90%
Benzo(g,h,i)perylene (ug/Kg)	C	111	16%	84%
Benzo(k)fluoranthene (ug/Kg)	C	70	10%	90%
Benzofluoranthenes, Total (ug/Kg)	A	41	59%	41%
Bis(2-Ethylhexyl) Phthalate (ug/Kg)	B	111	46%	54%
Butyl benzyl phthalate (ug/Kg)	C	111	7%	93%
Cadmium (mg/Kg)	A	111	98%	2%
Carbaryl (mg/Kg)	C	111	0%	100%
Carbazole (ug/Kg)	C	111	11%	89%
Chlorpyrifos (mg/Kg)	C	111	0%	100%
Chromium (mg/Kg)	A	111	100%	0%
Chrysene (ug/Kg)	B	111	34%	66%
Copper (mg/Kg)	A	111	100%	0%
DCPMU (mg/Kg)	C	111	1%	99%
Di-N-Octyl Phthalate (ug/Kg)	C	111	0%	100%
Dibenzo(a,h)anthracene (ug/Kg)	C	111	4%	96%
Dibenzofuran (ug/Kg)	C	111	2%	98%
Dibutyl phthalate (ug/Kg)	C	111	1%	99%
Dichlobenil (mg/Kg)	A	111	73%	27%
Diethyl phthalate (ug/Kg)	C	111	5%	95%
Dimethyl phthalate (ug/Kg)	C	111	0%	100%
Diuron (mg/Kg)	C	111	2%	98%
Fluoranthene (ug/Kg)	B	111	41%	59%
Fluorene (ug/Kg)	C	111	2%	98%
Indeno(1,2,3-cd)pyrene (ug/Kg)	C	111	20%	80%
Lead (mg/Kg)	A	111	100%	0%
Naphthalene (ug/Kg)	C	111	2%	98%
PCN-002 (ug/Kg)	C	111	0%	100%
Phenanthrene (ug/Kg)	B	111	31%	69%

Parameter	Case	n	% detected	% not-detected
Pyrene (ug/Kg)	B	111	40%	60%
Retene (ug/Kg)	A	111	65%	35%
Silver (mg/Kg)	A	111	57%	43%
Solids (%)	A	151	100%	0%
Total Benzofluoranthenes (ug/Kg)	B	111	28%	72%
Total Organic Carbon_SIEVE-0.063MM (%)	A	83	100%	0%
Total Organic Carbon_SIEVE-2.0MM (%)	A	110	100%	0%
Total PAH (ug/Kg)	B	111	43%	57%
Total PBDE (ug/Kg)	A	110	100%	0%
Total PCB (ug/Kg)	A	111	100%	0%
Triclopyr (mg/Kg)	C	111	0%	100%
Zinc (mg/Kg)	A	111	100%	0%

Table A3: Detection frequency of sediment PCB and PBDE congeners (see **Section 2.7.1** for Case definitions).

Parameter	Case	n	% detected	% not-detected
PCB-001 (pg/g)	B	110	28%	72%
PCB-002 (pg/g)	B	111	41%	59%
PCB-003 (pg/g)	B	111	38%	62%
PCB-004 (pg/g)	B	111	46%	54%
PCB-005 (pg/g)	C	111	9%	91%
PCB-006 (pg/g)	B	111	49%	51%
PCB-007 (pg/g)	C	111	9%	91%
PCB-008 (pg/g)	A	111	59%	41%
PCB-009 (pg/g)	B	111	31%	69%
PCB-010 (pg/g)	C	111	7%	93%
PCB-011 (pg/g)	B	111	43%	57%
PCB-012/013 (pg/g)	B	111	50%	50%
PCB-014 (pg/g)	C	111	4%	96%
PCB-015 (pg/g)	A	111	57%	43%
PCB-016 (pg/g)	B	111	45%	55%
PCB-017 (pg/g)	B	111	43%	57%
PCB-018/030 (pg/g)	B	111	44%	56%
PCB-019 (pg/g)	B	111	27%	73%
PCB-020/028 (pg/g)	B	111	45%	55%
PCB-021/033 (pg/g)	B	111	44%	56%
PCB-022 (pg/g)	A	111	59%	41%
PCB-023 (pg/g)	C	111	9%	91%
PCB-024 (pg/g)	C	111	18%	82%
PCB-025 (pg/g)	A	111	58%	42%
PCB-026/029 (pg/g)	A	111	54%	46%
PCB-027 (pg/g)	A	111	57%	43%
PCB-031 (pg/g)	B	111	41%	59%
PCB-032 (pg/g)	B	111	38%	62%
PCB-034 (pg/g)	C	111	19%	81%
PCB-035 (pg/g)	A	111	63%	37%
PCB-036 (pg/g)	B	111	25%	75%
PCB-037 (pg/g)	A	111	89%	11%
PCB-038 (pg/g)	B	111	33%	67%
PCB-039 (pg/g)	B	111	39%	61%
PCB-040/041/071 (pg/g)	A	111	77%	23%
PCB-042 (pg/g)	A	111	86%	14%
PCB-043 (pg/g)	B	111	43%	57%
PCB-044/047/065 (pg/g)	A	111	82%	18%
PCB-045/051 (pg/g)	B	111	46%	54%

Parameter	Case	n	% detected	% not-detected
PCB-046 (pg/g)	A	111	65%	35%
PCB-048 (pg/g)	A	111	66%	34%
PCB-049/069 (pg/g)	A	111	83%	17%
PCB-050/053 (pg/g)	A	111	62%	38%
PCB-052 (pg/g)	A	111	56%	44%
PCB-054 (pg/g)	C	111	17%	83%
PCB-055 (pg/g)	C	111	14%	86%
PCB-056 (pg/g)	A	111	90%	10%
PCB-057 (pg/g)	C	111	6%	94%
PCB-058 (pg/g)	C	111	6%	94%
PCB-059/062/075 (pg/g)	A	111	78%	22%
PCB-060 (pg/g)	A	111	79%	21%
PCB-061/070/074/076 (pg/g)	A	111	80%	20%
PCB-063 (pg/g)	A	111	67%	33%
PCB-064 (pg/g)	A	111	75%	25%
PCB-066 (pg/g)	A	111	85%	15%
PCB-067 (pg/g)	A	111	51%	49%
PCB-068 (pg/g)	A	111	57%	43%
PCB-072 (pg/g)	A	111	51%	49%
PCB-073 (pg/g)	C	111	4%	96%
PCB-077 (pg/g)	A	111	92%	8%
PCB-078 (pg/g)	C	111	4%	96%
PCB-079 (pg/g)	A	111	68%	32%
PCB-080 (pg/g)	C	111	5%	95%
PCB-081 (pg/g)	B	111	32%	68%
PCB-082 (pg/g)	A	111	86%	14%
PCB-083/099 (pg/g)	A	111	94%	6%
PCB-084 (pg/g)	A	111	92%	8%
PCB-085/116/117 (pg/g)	A	111	94%	6%
PCB-086/087/097/108/119/125 (pg/g)	A	111	95%	5%
PCB-088/091 (pg/g)	A	111	92%	8%
PCB-089 (pg/g)	B	111	41%	59%
PCB-090/101/113 (pg/g)	A	111	85%	15%
PCB-092 (pg/g)	A	111	92%	8%
PCB-093/095/098/100/102 (pg/g)	A	111	95%	5%
PCB-094 (pg/g)	B	111	29%	71%
PCB-096 (pg/g)	B	111	46%	54%
PCB-103 (pg/g)	B	111	45%	55%
PCB-104 (pg/g)	C	111	10%	90%
PCB-105 (pg/g)	A	111	96%	4%

Parameter	Case	n	% detected	% not-detected
PCB-106 (pg/g)	C	111	2%	98%
PCB-107/124 (pg/g)	A	111	85%	15%
PCB-109 (pg/g)	A	111	90%	10%
PCB-110/115 (pg/g)	A	111	95%	5%
PCB-111 (pg/g)	C	111	15%	85%
PCB-112 (pg/g)	C	111	5%	95%
PCB-114 (pg/g)	A	111	70%	30%
PCB-118 (pg/g)	A	111	95%	5%
PCB-120 (pg/g)	B	111	50%	50%
PCB-121 (pg/g)	C	111	5%	95%
PCB-122 (pg/g)	A	111	59%	41%
PCB-123 (pg/g)	A	111	75%	25%
PCB-126 (pg/g)	A	111	59%	41%
PCB-127 (pg/g)	B	111	25%	75%
PCB-128/166 (pg/g)	A	111	90%	10%
PCB-129/138/160/163 (pg/g)	A	111	95%	5%
PCB-130 (pg/g)	A	111	85%	15%
PCB-131 (pg/g)	B	111	50%	50%
PCB-132 (pg/g)	A	111	91%	9%
PCB-133 (pg/g)	A	111	64%	36%
PCB-134/143 (pg/g)	A	111	70%	30%
PCB-135/151/154 (pg/g)	A	111	94%	6%
PCB-136 (pg/g)	A	111	90%	10%
PCB-137 (pg/g)	A	111	82%	18%
PCB-139/140 (pg/g)	A	111	66%	34%
PCB-141 (pg/g)	A	111	90%	10%
PCB-142 (pg/g)	C	111	1%	99%
PCB-144 (pg/g)	A	111	74%	26%
PCB-145 (pg/g)	C	111	12%	88%
PCB-146 (pg/g)	A	111	94%	6%
PCB-147/149 (pg/g)	A	111	98%	2%
PCB-148 (pg/g)	B	111	32%	68%
PCB-150 (pg/g)	B	111	33%	67%
PCB-152 (pg/g)	B	111	26%	74%
PCB-153/168 (pg/g)	A	111	95%	5%
PCB-155 (pg/g)	B	111	27%	73%
PCB-156/157 (pg/g)	A	111	87%	13%
PCB-158 (pg/g)	A	111	87%	13%
PCB-159 (pg/g)	A	111	56%	44%
PCB-161 (pg/g)	C	111	1%	99%
PCB-162 (pg/g)	B	111	42%	58%
PCB-164 (pg/g)	A	111	86%	14%

Parameter	Case	n	% detected	% not-detected
PCB-165 (pg/g)	C	111	2%	98%
PCB-167 (pg/g)	A	111	93%	7%
PCB-169 (pg/g)	C	111	1%	99%
PCB-170 (pg/g)	A	111	95%	5%
PCB-171/173 (pg/g)	A	111	91%	9%
PCB-172 (pg/g)	A	111	90%	10%
PCB-174 (pg/g)	A	111	98%	2%
PCB-175 (pg/g)	A	111	58%	42%
PCB-176 (pg/g)	A	111	82%	18%
PCB-177 (pg/g)	A	111	96%	4%
PCB-178 (pg/g)	A	111	90%	10%
PCB-179 (pg/g)	A	111	93%	7%
PCB-180/193 (pg/g)	A	111	96%	4%
PCB-181 (pg/g)	B	111	47%	53%
PCB-182 (pg/g)	B	111	35%	65%
PCB-183/185 (pg/g)	A	111	94%	6%
PCB-184 (pg/g)	B	111	24%	76%
PCB-186 (pg/g)	C	111	3%	97%
PCB-187 (pg/g)	A	111	95%	5%
PCB-188 (pg/g)	B	111	41%	59%
PCB-189 (pg/g)	A	111	68%	32%
PCB-190 (pg/g)	A	111	85%	15%
PCB-191 (pg/g)	A	111	68%	32%
PCB-192 (pg/g)	C	111	7%	93%
PCB-194 (pg/g)	A	111	95%	5%
PCB-195 (pg/g)	A	110	92%	8%
PCB-196 (pg/g)	A	111	93%	7%
PCB-197/200 (pg/g)	A	111	86%	14%
PCB-198/199 (pg/g)	A	111	95%	5%
PCB-201 (pg/g)	A	111	87%	13%
PCB-202 (pg/g)	A	111	96%	4%
PCB-203 (pg/g)	A	111	97%	3%
PCB-204 (pg/g)	C	111	19%	81%
PCB-205 (pg/g)	A	111	74%	26%
PCB-206 (pg/g)	A	112	96%	4%
PCB-207 (pg/g)	A	111	76%	24%
PCB-208 (pg/g)	A	111	95%	5%
PCB-209 (pg/g)	A	133	98%	2%
PBDE-007 (pg/g)	B	110	20%	80%
PBDE-008/011 (pg/g)	B	110	24%	76%
PBDE-010 (pg/g)	C	110	1%	99%
PBDE-012/013 (pg/g)	B	110	38%	62%

Parameter	Case	n	% detected	% not-detected
PBDE-015 (pg/g)	A	110	90%	10%
PBDE-017/025 (pg/g)	A	110	87%	13%
PBDE-028/033 (pg/g)	A	110	87%	13%
PBDE-030 (pg/g)	C	110	2%	98%
PBDE-032 (pg/g)	C	110	8%	92%
PBDE-035 (pg/g)	A	110	55%	45%
PBDE-037 (pg/g)	B	110	45%	55%
PBDE-047 (pg/g)	A	110	81%	19%
PBDE-049 (pg/g)	A	110	95%	5%
PBDE-051 (pg/g)	A	110	72%	28%
PBDE-066 (pg/g)	A	110	91%	9%
PBDE-071 (pg/g)	A	110	75%	25%
PBDE-075 (pg/g)	A	110	52%	48%
PBDE-077 (pg/g)	B	110	49%	51%
PBDE-079 (pg/g)	A	110	68%	32%
PBDE-085 (pg/g)	A	110	78%	22%
PBDE-099 (pg/g)	A	109	73%	27%
PBDE-100 (pg/g)	A	110	81%	19%
PBDE-105 (pg/g)	C	110	0%	100%
PBDE-116 (pg/g)	C	110	5%	95%
PBDE-119/120 (pg/g)	B	110	47%	53%
PBDE-126 (pg/g)	C	110	5%	95%
PBDE-128 (pg/g)	C	110	2%	98%
PBDE-138/166 (pg/g)	B	110	44%	56%
PBDE-140 (pg/g)	B	110	33%	67%
PBDE-153 (pg/g)	A	110	88%	12%
PBDE-154 (pg/g)	A	110	86%	14%
PBDE-155 (pg/g)	A	110	54%	46%
PBDE-181 (pg/g)	C	110	16%	84%
PBDE-183 (pg/g)	A	110	87%	13%
PBDE-184 (pg/g)	B	110	32%	68%
PBDE-190 (pg/g)	B	111	23%	77%
PBDE-191 (pg/g)	C	109	19%	81%
PBDE-203 (pg/g)	A	110	75%	25%
PBDE-206 (pg/g)	A	110	81%	19%
PBDE-207 (pg/g)	A	110	81%	19%
PBDE-208 (pg/g)	A	110	81%	19%
PBDE-209 (pg/g)	A	110	83%	17%

Appendix B: Statistical Summary of Biological, Water, Sediment, Habitat and Landscape Data Collected Within and Outside Urban Growth Areas

Appendix B is a Microsoft Excel file available for download at:
http://your.kingcounty.gov/dnrp/library/2018/kcr2968/kcr2968-digital-appendices.zip

Table B1: Benthic Macroinvertebrate Metrics

Table B2: Periphyton Metrics

Table B3: In Situ Data

Table B4: Water Chemistry Data

Table B5: Water Quality Index

Tale B6: Sediment Chemistry Data

Table B7: Habitat Metrics

Table B8: Physical Landscape and Land Cover Data

Appendix C: Summary of Cumulative Distribution Frequency Analysis of Biological, Water, Sediment, Habitat and Landscape Data Collected Within and Outside Urban Growth Areas

Appendix C is a set of Microsoft Excel files available for download at:
http://your.kingcounty.gov/dnrp/library/2018/kcr2968/kcr2968-digital-appendices.zip

Table C1: Benthic Macroinvertebrate Metrics

Table C2: Periphyton Metrics

Table C3: In Situ Data

Table C4: Water Chemistry Data

Table C5: Water Quality Index

Tale C6: Sediment Chemistry Data

Table C7: Habitat Metrics

Table C8: Physical Landscape and Land Cover Data

Appendix D: Maps Showing Sampling Locations of Other Puget Lowland Monitoring Programs Compared to the 2015 SAM PLES Study

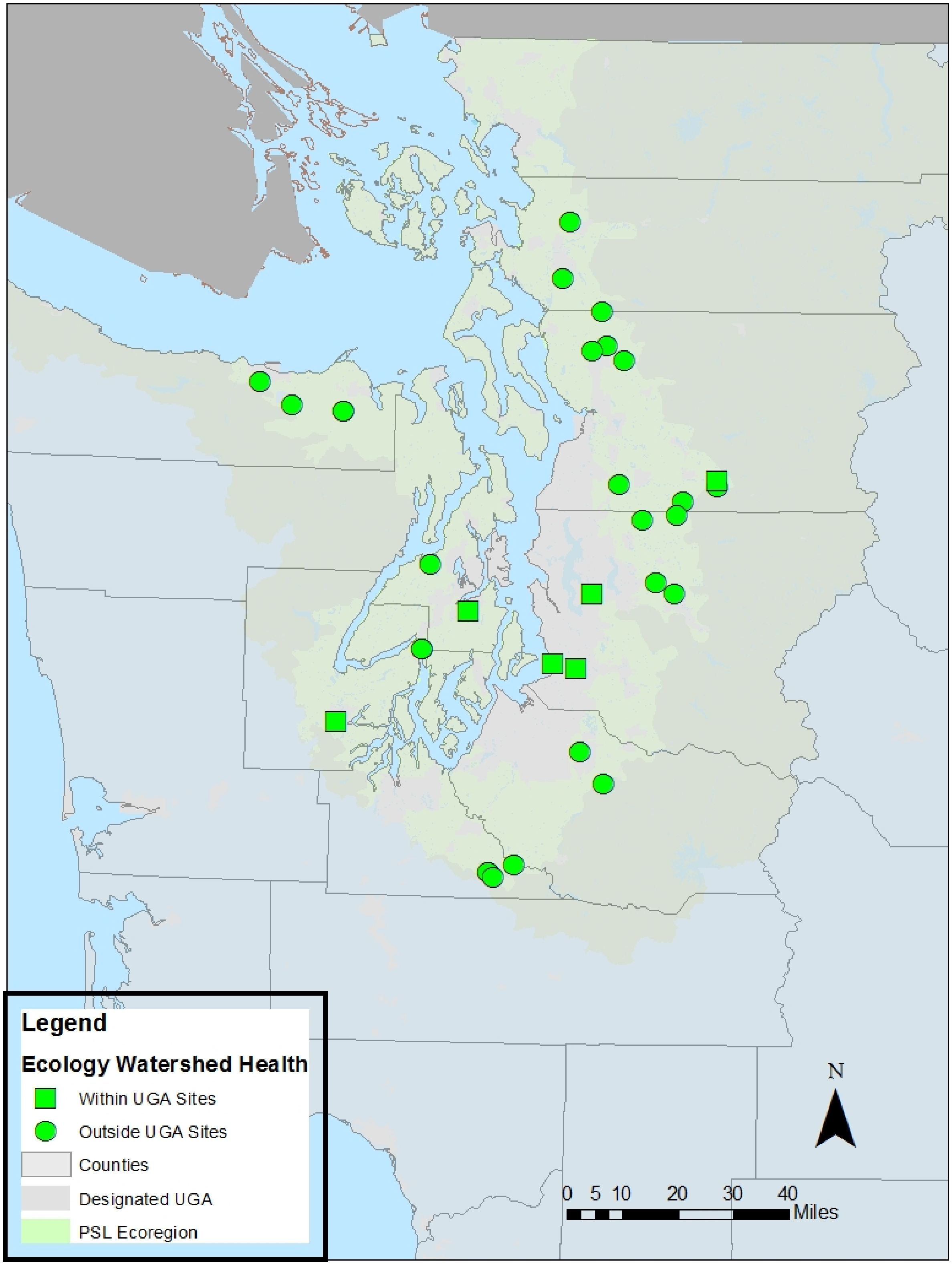

Figure D-1. Ecology Watershed Health and Salmon Recovery stream sites sampled in 2009.

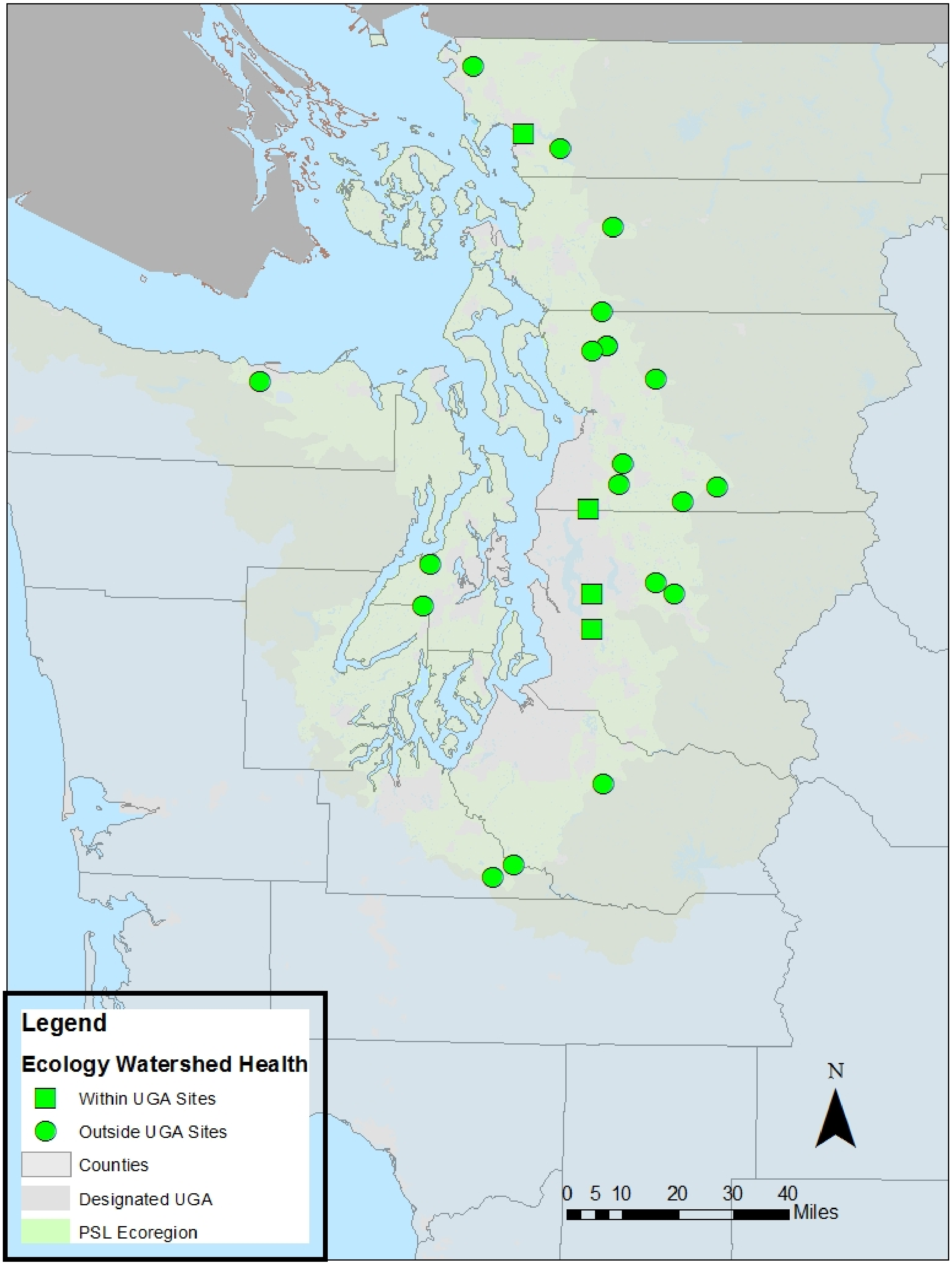

Figure D-2. Ecology Watershed Health and Salmon Recovery stream sites sampled in 2013.

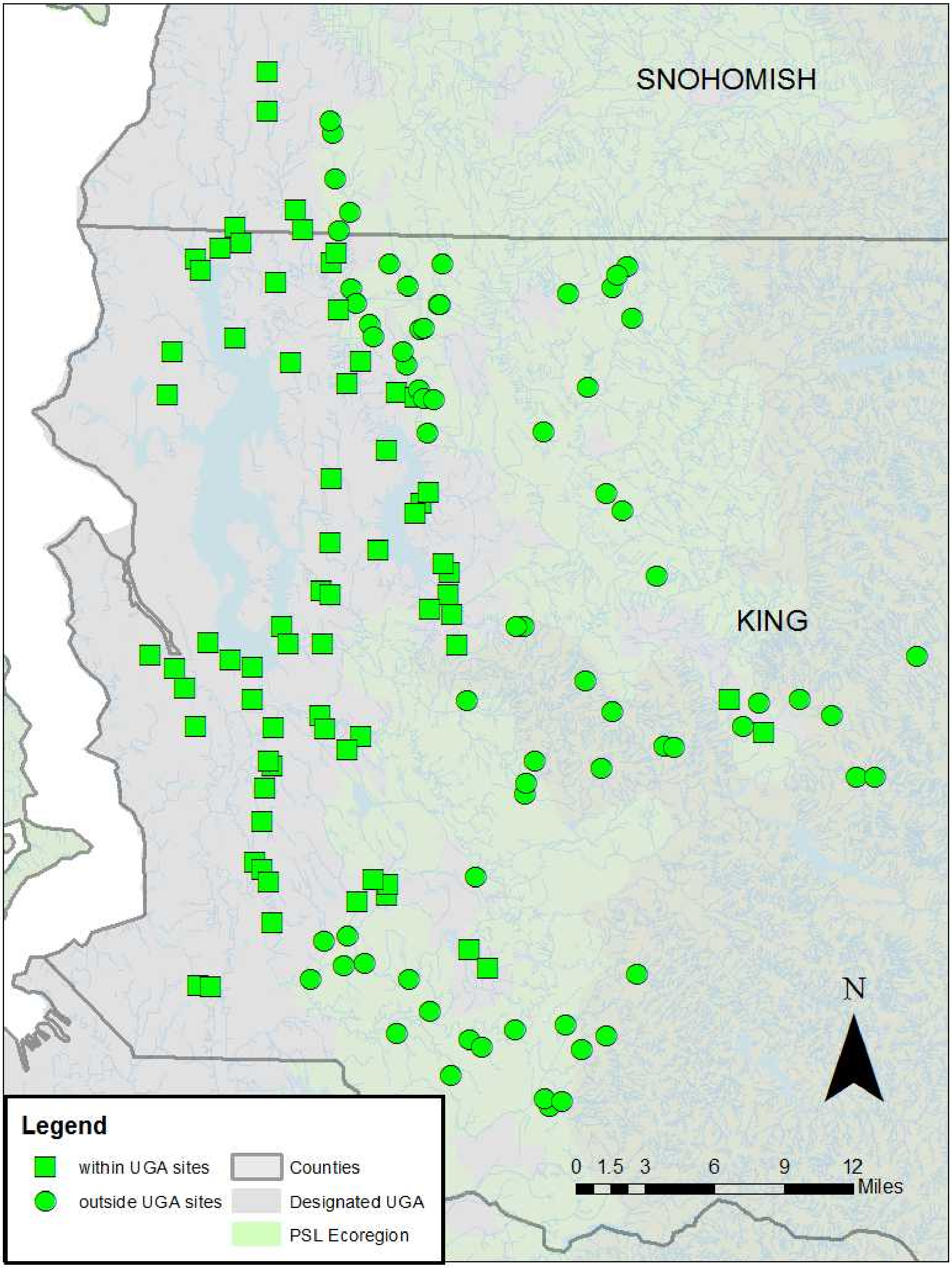

Figure D-3. King County stream benthos sites sampled in 2015.

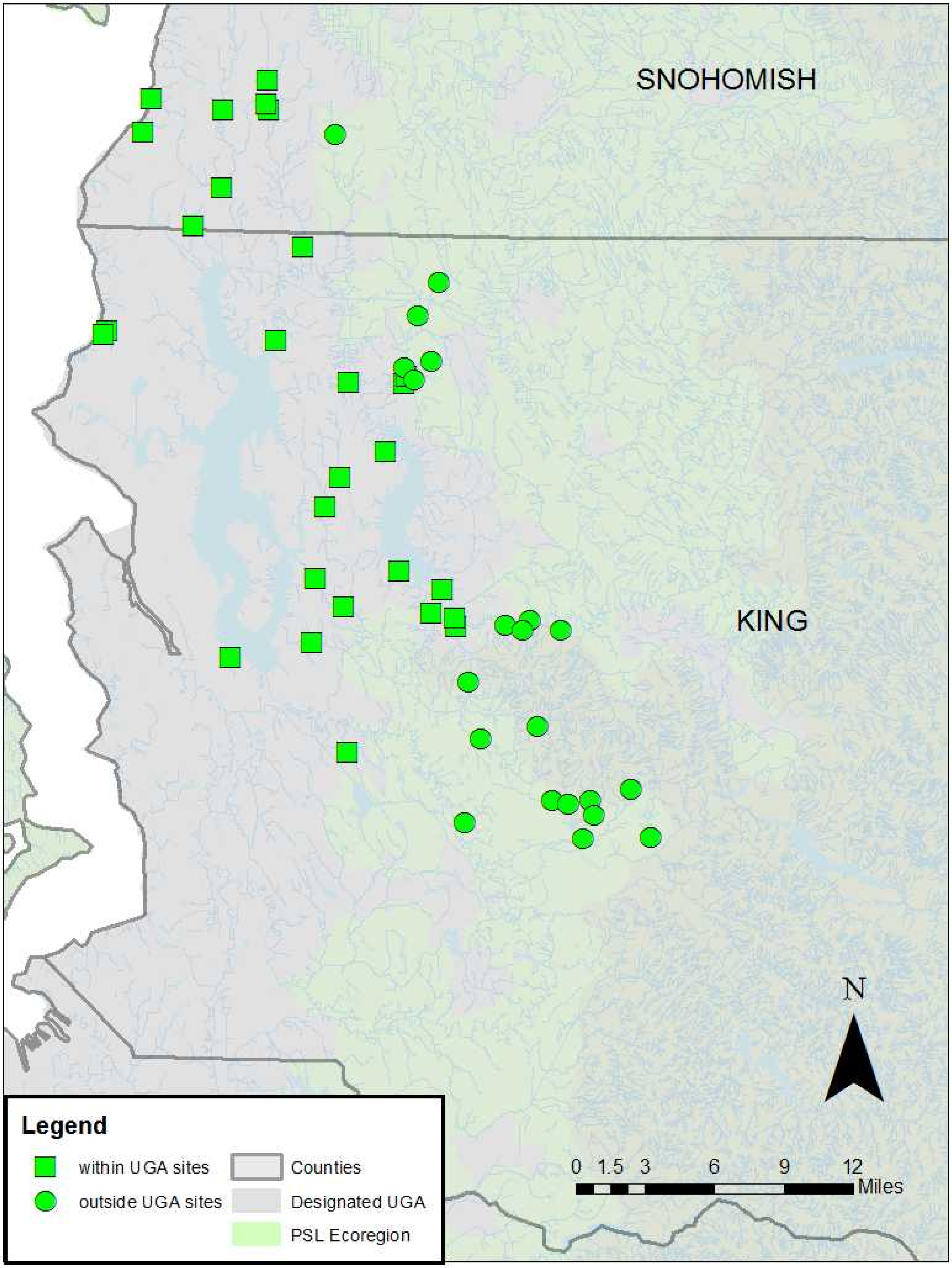

Figure D-4. King County WRIA 8 Status and Trends sites sampled in 2010-2013.

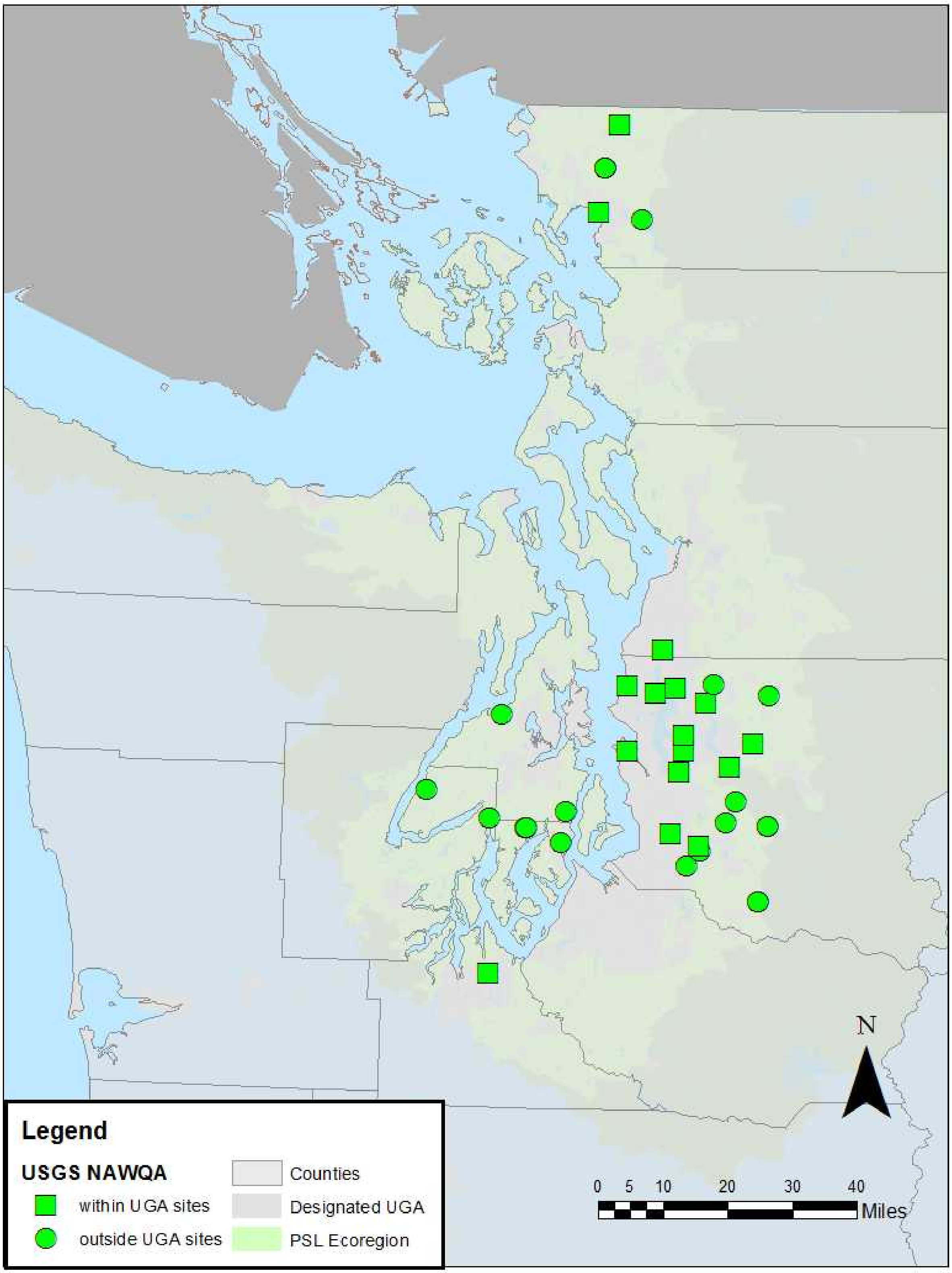

Figure D-5. USGS Pacific Northwest Stream Quality Assessment sites sampled for sediment chemistry in 2015.

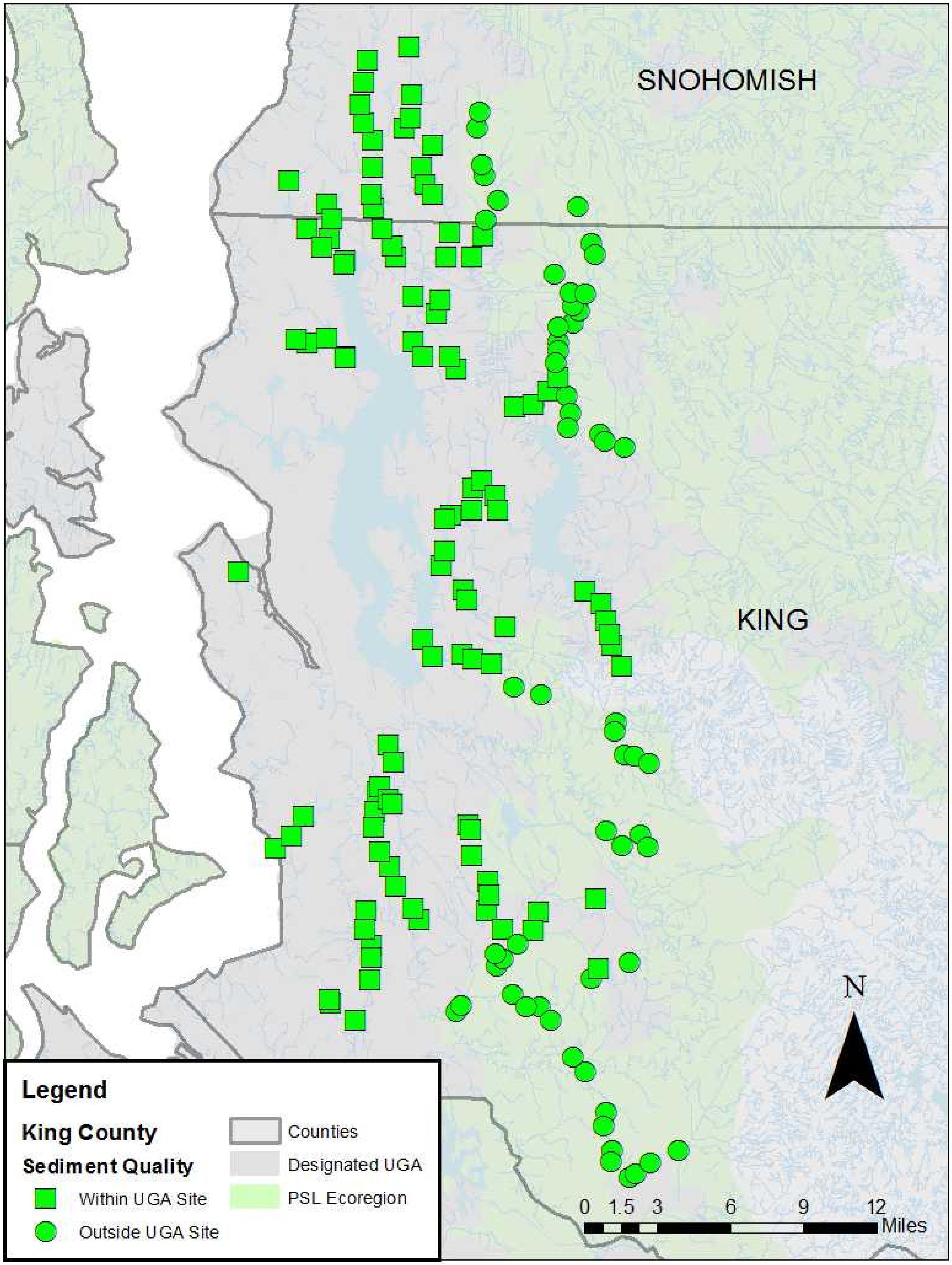

Figure D-6. King County stream sites sampled for sediment chemistry in 2004-2012.

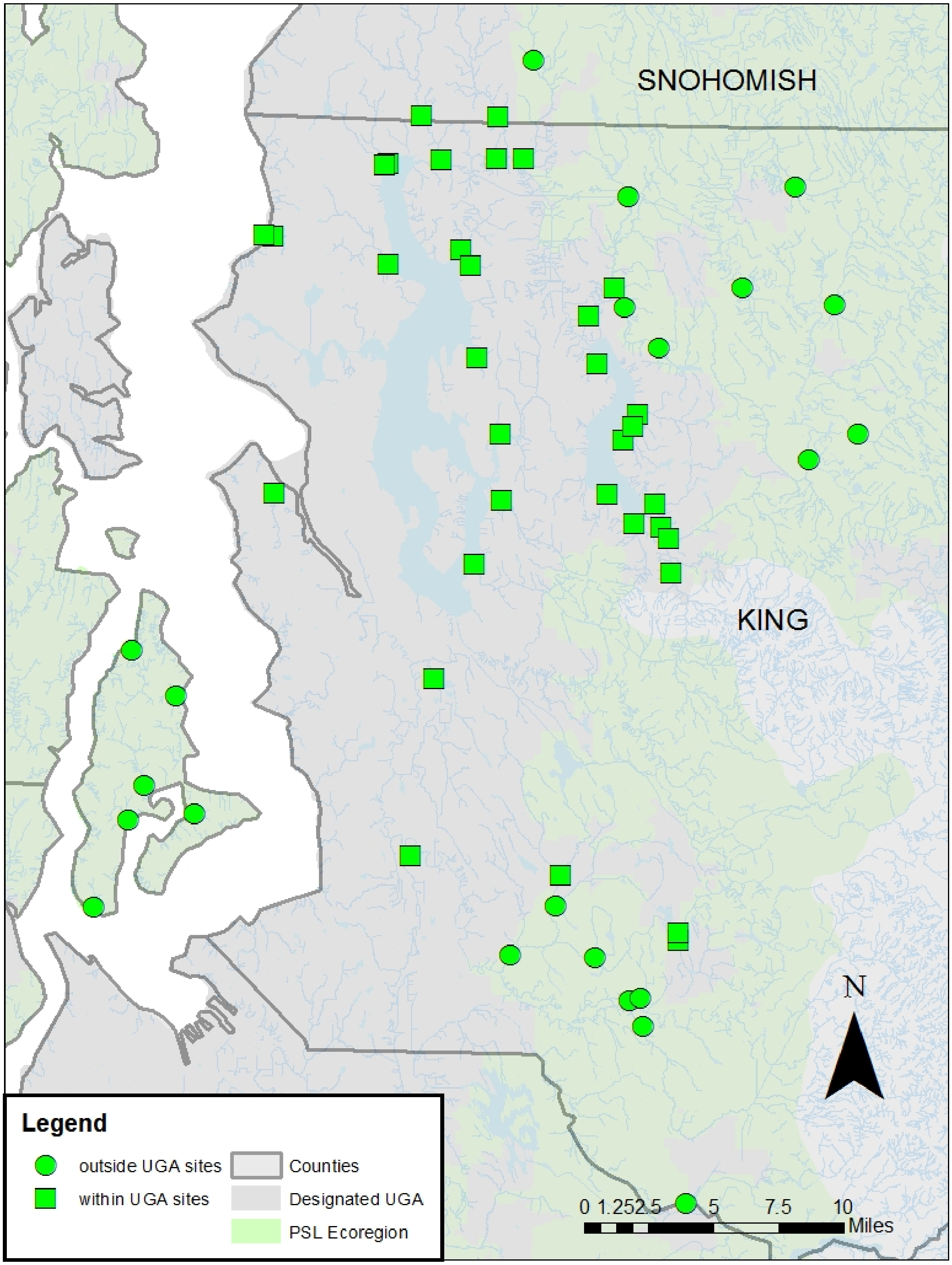

Figure D-7. King County stream sites sampled for water quality in 2015.

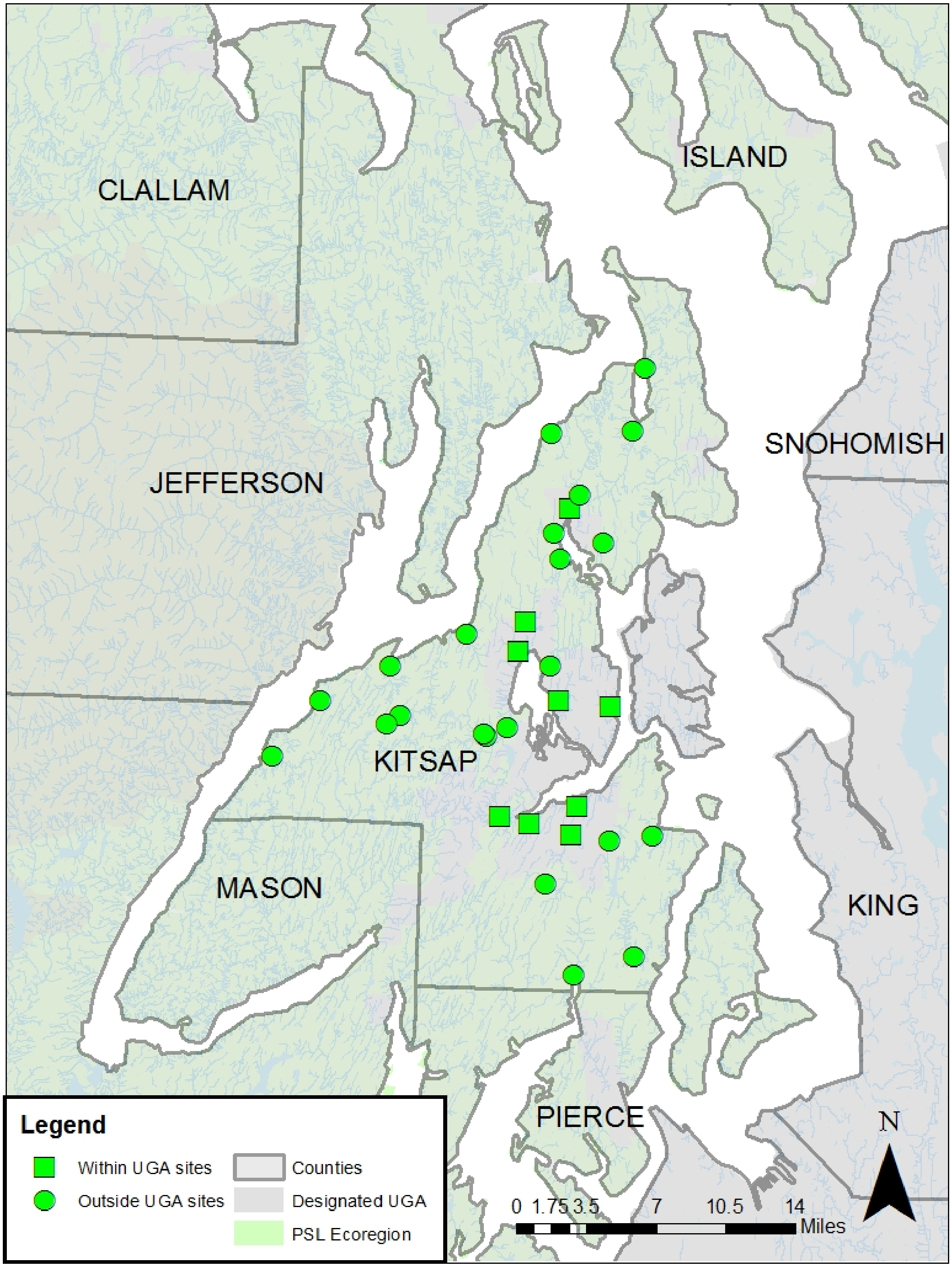

Figure D-8. Kitsap County stream habitat sites sampled in 2012-2016.

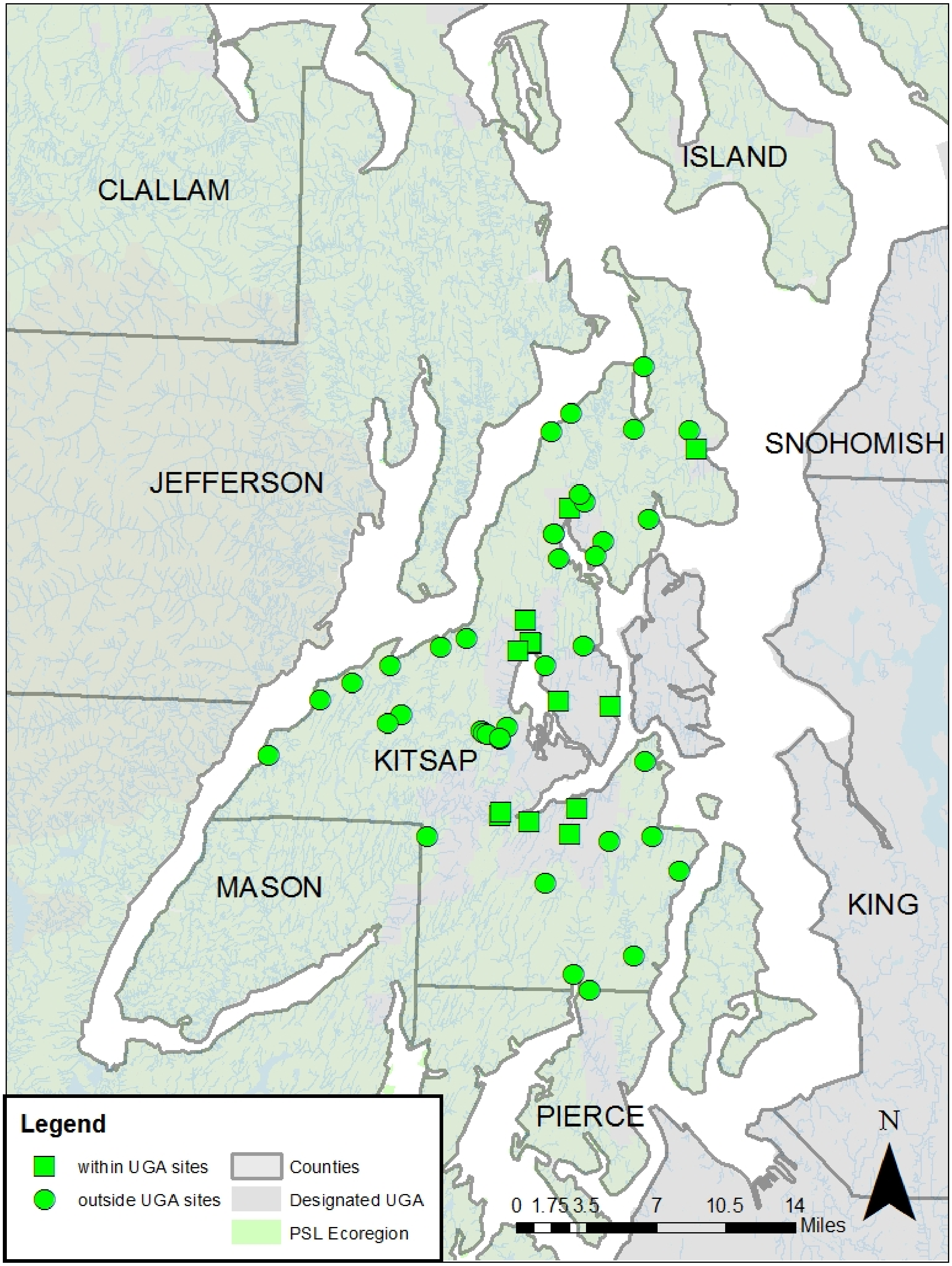

Figure D-9. Kitsap County stream benthos sites sampled in 2012-2016.