Trilogy and Redmond Ridge Urban Planned Development (UPD) Natural Resources Monitoring Midpoint Review

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King County

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1.0 Executive Summary

The Trilogy and Redmond Ridge Urban Planned Development (UPD) Natural Resources Monitoring Midpoint Review report summarizes the natural resource monitoring that was a condition of development permits. The primary purpose of the report is to present the extent to which the developments have affected ecosystem function by evaluating biological, physical, and chemical responses of some of our most sensitive and valuable natural resources to changing landcover conditions related to the developments. This report focuses only on the natural resources. However, it is part of a larger mid-point review of project impacts as a whole, including those with human social implications such as changes to traffic routing and volumes, etc.

The monitoring techniques and protocols implemented in this study were specified in the original monitoring plans (King County 1999, 2001). This report presents data on changes to affected natural resources, if those changes exceed permitted or expected conditions, and suggestions on how to proceed during continuing phases. In general, despite some changes in ecosystem conditions, our analyses indicate that the permit conditions have been largely met due to the natural resources protections that were put in place. However, one key finding is that because our monitoring efforts were largely not set up in a hypothesis testing framework, it is difficult or impossible to say with statistical certainty, that natural resources protective measures have been adequate. Hopefully, the results of this study can inform how potential future impacts to natural systems are avoided or mitigated in other developments within the County.

1.1 Summary of Key Findings

Macroinvertebrates

Aquatic macroinvertebrate populations appear to be in decline at several locations within creeks draining from the UPDs. Some of these declines are beyond the range of variability established during three years of pre-development data from 1999 to 2001. In addition, declines exceed established threshold criteria in two locations. However, one year of post-development data is insufficient to define a trend. These streams should continue to be monitored closely, but no corrective actions are warranted at this time. Most basins should be sampled through at least 2008 or 2009. In addition, to ensure at least three years of post-development data to compare with pre-development data monitoring must continue through at least 2008 at all locations.

Amphibians

Amphibian populations in the UPD wetlands appear to fluctuate greatly from year to year. Factors contributing to amphibian declines could include hydrologic excursions, changes in plant community structure, introduction of predators, epidemics or parasites, meteorological conditions including precipitation and climate change, and /or water quality problems. Some of these factors could be attributed to the increasing urbanization of the basin (e.g., hydrological changes that influence plant community structure and available habitat and water quality problems), but many cannot. Given the sensitivity of amphibians to environmental change, these populations should continue to be monitored closely. However, at this time, no corrective actions are warranted.

Fish

Fish populations fluctuate naturally and population estimates are inherently variable relying on numerous assumptions. In addition, the population estimates that were calculated for this study reflect a single moment in time for the specific reaches sampled. No corrective actions are recommended at this time. Nevertheless, the declines in Colin South, Evans East, and Adair cutthroat trout populations should be monitored closely for continued changes over time and perhaps expanded to test if the reaches sampled are reflective of the overall population conditions in those systems. If continued declines are linked to UPD development, appropriate corrective actions should be implemented.

Wetland Vegetation

Wetland vegetation surveys only occurred in wetland BBC 52 and were intended to monitor if the UPD development has had an adverse impact on the botanical community of this bog wetland system. The study design for this component of the overall natural resources monitoring program seems to be adequate to detect changes at this scale. However, some of the specific questions about wetland botanical community integrity relative to development progress will require more time in order to determine quantitatively if there has been an adverse impact. Given that this basin is still largely undeveloped, the prescribed monitoring timeline (sampling every other year, 4 times following 75% buildout) may provide enough information to establish a trend, assuming a new ecological trajectory is established due to some chronic impact to the system. At that time it may be necessary to re-evaluate the monitoring plan, which could include extending the monitoring period, in order to make that determination quantitatively.

Wetlands

Headwaters wetlands in the Big Bear Creek and Snoqualmie River systems remain within the seasonal fluctuations that were observed during the baseline data collection period. The Evans Creek wetlands are very affected by beaver activity. However, other than some construction related events that were corrected, the wetlands do not show unexpected changes in hydrologic behavior.

Streams

The two Colin Creek tributaries are behaving as expected. The Unnamed Creek bypass is functioning and data from the two Unnamed Creek gauges are consistent with expectations with respect to summer low flows and storm peak magnitudes. The Adair Creek bypass is functioning, but does not seem to carry much flow. Adair Creek is exhibiting some affects consistent with an urbanizing watershed, i.e., increased “flashiness.” To mitigate for these changes in hydrograph, an adjustment of the bypass level setting may be necessary.

Facilities

Monitoring of drainage facilities is slated to begin under the conditions set forth in the monitoring plans and will continue until 5 years after 75 % buildout has occurred in respective basins. Fifty-three stormwater facilities have been or will be constructed for this development. Of those, seven facilities have been identified as representative and will be included in this monitoring program.

Stream Cross-section stability

Most of the differences in stream channel cross-sections that were measured during this study do not represent geomorphically significant changes in stream stability. However, changes in Adair and Rutherford Creeks may be a reflection of some upstream change in watershed hydrology that is manifest as a chronic impact, or they could represent the channel response to a single event (e.g., a piece of large wood falling in the channel). In either case, continuing to monitor these locations in subsequent years should be required and if further geomorphic changes are observed, corrective actions should be implemented.

Water Quality

Due to some inadequacies with the sampling design, we lack a detailed understanding of the natural variability in conductivity and pH between years. In addition, inconsistent sampling protocols and field personnel may introduce additional sources of variability. Nevertheless, there are apparent increases in conductivity, dissolved oxygen, and pH across wetland and stream locations that may warrant an assessment of stormwater facility function to determine whether any additional best management practices (BMPs) can be implemented to further mitigate the impacts of UPD development.

Welcome Lake

Water Quality monitoring by the Water and Land Resources Division (WLRD) of King County, is slated to begin following 75% buildout in the basins that drain to the lake, which occurred during spring 2006. Therefore, much of the data that have been collected on Welcome Lake have been by volunteers, consultants, or the Lakes group in the King County Department of Natural Resources and Parks. The data collected to date indicate that there is no consistent long-term trend toward decreased water quality in Welcome Lake. In fact, there is some evidence that total phosphorus has decreased in the water column over time.

Sediment Metals

Prior to 2006, there were only four instances where metal concentrations exceeded State guidelines. In each case, the degree to which the concentrations exceeded their respective limits was relatively small. Therefore, the potential for adverse effects to aquatic organisms is considered to be slight. No corrective actions are recommended at this time; however, it may be necessary to add additional sediment sampling sites. Sampling was reduced to Rutherford and Colin South beginning in 2005 since these were the only two sites mandated by the monitoring plans (King County 1999, King County 2001). However, Colin South goes dry during most summers making base flow sampling inconsistent from year-to-year. Therefore, it may be appropriate to re-establish sampling at Unnamed Creek, which has year-round flows and sediment data from 1991, 1999, 2000, and 2001 with which to draw comparisons.

Groundwater Quantity and Quality

There appears to be no evidence that the UPDs have negatively impacted groundwater. There are some indications that nitrate concentrations in the shallow groundwater have increased. However, this conclusion is only qualitative due to a lack of adequate data coverage and great variability in the data. It appears that these elevated nitrate concentrations are not an effect of the UPD development. Changes in the monitoring program are not called for at this point.

2.0 Introduction

Two urban planned developments (UPD), Trilogy and Redmond Ridge, are currently under construction in King County, Washington. This report marks the mid-point of development, signaling the point where permits for 2500 dwelling units have been issued. In some basins, construction is largely completed. In others, it is just beginning. Overall, the construction phase is approximately 75% completed. The purpose of this report is to assess the effectiveness of the protective measures for natural resources that were implemented as a condition of these developments. The mid-point is seen as a moment during the construction phase of the project where information gained from this monitoring program can be used to make adjustments to existing structures and infrastructure as well as inform future construction practices. The Water and Land Resources Division (WLRD) of King County has been performing the ecological monitoring of affected resources related to these developments.

Although these are two independent developments, many of the monitoring elements are identical and are therefore being conducted simultaneously. The two projects are treated as one within this report for simplicity. This report describes the progress and results of the UPD ecological response monitoring program conducted beginning in 1998 pre-construction conditions through a continuum of changing landscape conditions currently. Monitoring programs were developed to coincide with water years because the data collection elements of this program are driven mostly by seasonal weather patterns that dictate hydrologic conditions.

The project area encompasses approximately 1035 hectares and contains numerous high value natural resources including bogs and other wetlands and headwaters streams for two major drainages (i.e., the Cedar/ Lake Washington and Snoqualmie River drainages). The site is located on a broad upland plateau with generally low-relief (Figure 2.0). Slopes are generally less than 15%, locally exceeding 40% to the east where the plateau abuts the Snoqualmie River valley. Prior to development, the property was largely second growth evergreen and coniferous forest with native understory vegetation made up mostly of species normally found in western Washington. The area has a history of glacial activity, most recently the Vashon glaciation 13,000 to 15,000 years ago. Glacial geological deposits include outwash, till, and recessional drift. Soils on the site are largely comprised of glacially derived material and peat/ organic soils (GeoEngineers, 1995).

The monitoring plans for the developments contain elements for aquatic macroinvertebrates, amphibians, fish, wetland vegetation, hydrology and flow analysis, stream cross-section stability, water quality, sediment quality, and groundwater. In addition, special attention was required for Welcome Lake water quality despite the fact that it lies wholly outside of the UPD boundaries, but downstream of affected wetland and stream resources (King County 1999). Some additional baseline data for the UPD projects were collected in 1989-93. The data being collected currently are compared in this report with this early baseline wherever possible. However, most of the measurements collected under this program were collected after the first permits were issued. During 1998-99 construction was just beginning and development impacts were still negligible. Therefore, in some instances data collected in the early phases of construction are useful for both confirming the consistency of the early baseline data and establishing pre-construction conditions.

This report presents analyses focused on natural resources impacts relative to land cover and land use changes within the UPDs. Impacts to the following resources were addressed as part of this report.

- Macroinvertebrates
- Amphibians
- Fish
- Wetland Vegetation
- Hydrology/Flow Analysis
- Stream Cross-section stability
- Water Quality
- Sediment Metals
- Groundwater Quantity and Quality

Unintended impacts to other ecosystem components are not anticipated, nor expected to receive further study.

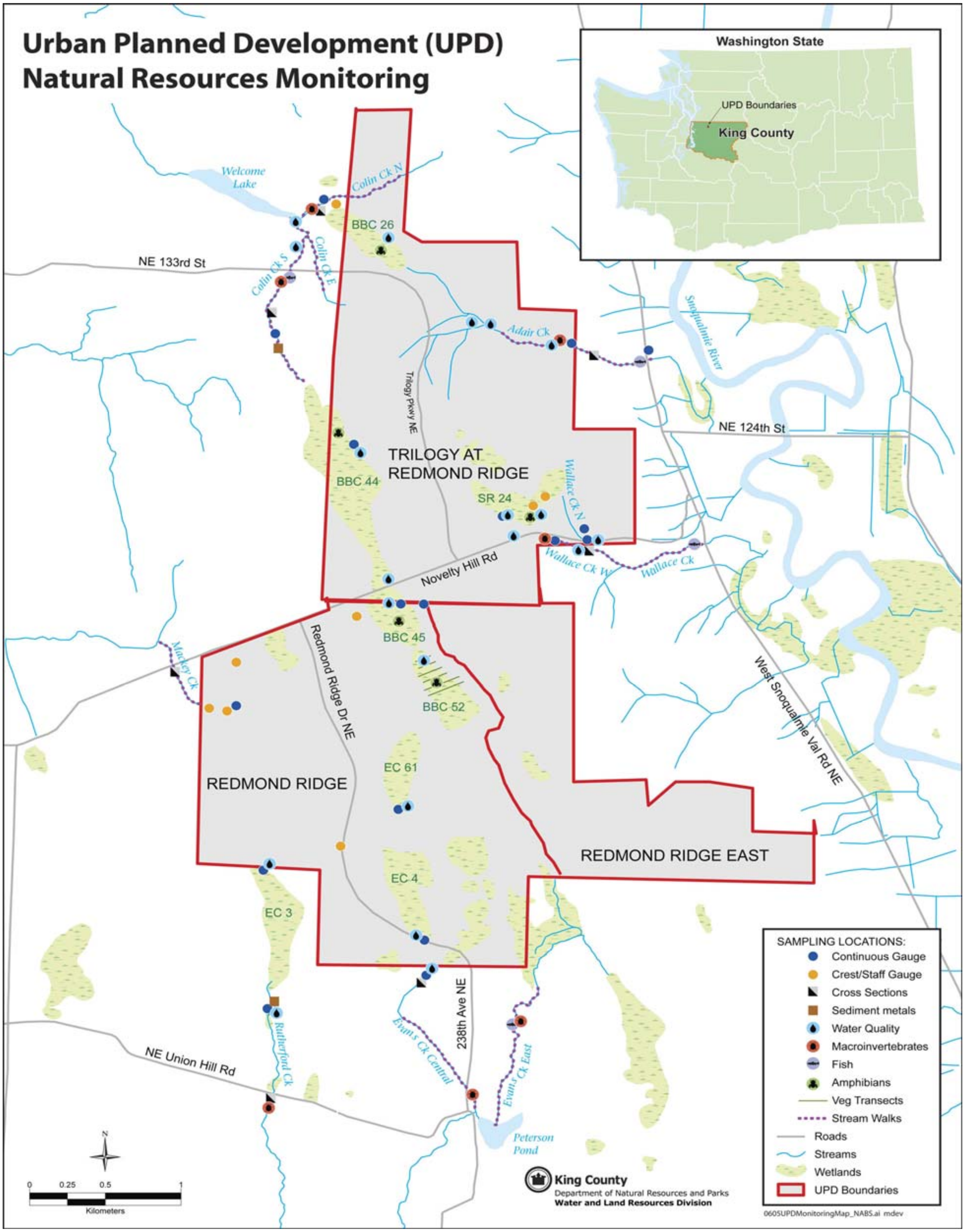

Figure 2.0. Location Map of UPD projects including locations of natural resources monitoring parameters.

2.1 King County Goals

King County had two main goals for this mid-point review: first, to determine if chronic impacts related to the UPDs are consistent with those in the EIS; and second, if the impacts are not consistent with the EIS, then can conditions be imposed prior to the issuance of future permits that can mitigate for those impacts. Specifically, natural resource goals of the mid-term review are to establish whether provisions to protect wetlands, plants and animals have been met through (a) the designation of natural resource protection areas encompassing wetlands and buffers, (b) limitations on encroachments into buffers for necessary road, utility, storm water, and recreational facilities, and (c) minimizing hydrological impacts to wetlands via storm water management, providing for mitigation of wetland impacts, retaining native vegetation, and providing educational materials (Raedeke Associates Inc. 2006).

To asses whether these goals are being achieved at least two monitoring types should initially be implemented and ongoing. These include *compliance* monitoring to determine if regulations have been followed and no illegal encroachment or damage has occurred, and *effectiveness* monitoring to assess if SEPA natural resource protection/restoration goals have been met. Compliance monitoring has been carried out as expected with the projects implemented. When altered, the resulting changes have been reviewed by staff and all changes mutually agreed upon by King County, consultants, and developers. Effectiveness monitoring determines the extent to which environmental protection goals have been met. It is generally preferable to measure effectiveness by directly monitoring biological parameters (Karr 1998, Karr and Chu 1999). Accordingly, we listed and quantified the impacts to natural resources (wetlands, plants, and animals) and reviewed the project documentation to determine whether identified impacts were, or are being mitigated in accordance with agreed to requirements (but also in accordance to broader SEPA Goals).

The listing of Cumulative Impacts identifies expected outcomes and would be the underpinnings for hypothesis testing through the development of a robust study design of data collection and analysis. Impacts that cannot be mitigated are identified as are those that can. Although some impacts are unavoidable and cannot be corrected through mitigation, it is more important to ensure that proper mitigations are in place for those impacts that can be corrected.

2.2 Mitigatable

2.2.1 Plants and Animals

- Retention of more than 500 ha of native open space on both properties which would maintain linkages among retained habitat on and off the UPD Site.

2.2.2 Wetlands

- Direct alteration of wetlands for Trilogy UPD totals less than one acre
- Loss of wetlands due to road fill
- Direct alteration of wetlands for Redmond Ridge UPD/FCC totals less than 1 ha (less than 1/2 ha fill and less than 1/2 ha for overstory removal and hydrologic changes)

2.3 Unmitigatable

2.3.1 Plants and Animals

- Conversion of 405 ha of native forest habitat to other uses and fragmentation of the remaining forest.
- Loss of native forests attributable to both projects would cause greater reduction than of either project alone.
- Some species, such as wide-ranging mammals, such as black bear, may be eliminated from area.
- Use of area by wide-ranging birds, such as pileated woodpeckers and raptors would likely diminish.
- Other species more adapted to urban areas and edges would likely increase in numbers.

2.3.2 Wetlands

- Development of the UPDs increases the potential for hydrologic and water quality impacts to Wetlands 44, BBC 44, BBC 45, and Welcome Lake. But the change in BBC 44 and BBC 45 was expected to be minor, primarily resulting in slightly higher water levels during fall and winter; no substantial impact expected.
- No adverse impacts likely to wetland Plant communities around the margins of Peterson pond (Wetland EC38).

3.0 Construction Overview

3.1 Trilogy

The general development phasing takes approximately 2 years to complete once the plats are recorded and the building permits are issued. The construction begins with land clearing, road construction, soil management, and is followed by building construction. Construction of the Trilogy project began during summer 1998 and affects parts of three creek drainages. Portions of Colin Creek (tributary to Lake Washington), Adair Creek, and Unnamed Creek (both tributary to the Snoqualmie River) are affected by the development. In addition, 29 stormwater facilities were constructed as part of that development. These facilities either discharge directly to wetlands or a bypass pipe system, or allow the stormwater runoff to infiltrate directly into the ground. Of particular concern are the facilities that discharge detained stormwater into the bog Wetland BBC 44. Currently, the percent developed ranges from fully built (100%) to no construction activities. Average percent built in the affected creek watersheds ranges from 0% to approximately 65% (Figure 4.1.1.1).

In addition to construction associated with the Trilogy development, construction across the property of the Tolt 2 water supply line for the City of Seattle commenced during summer of 1998. The pipeline crosses the Trilogy property from the eastern border, turning south along 236th Avenue NE, and continuing west along Novelty Hill Road. Pipeline construction on Trilogy was completed in the fall of 1998.

3.2 Redmond Ridge

Construction of the Redmond Ridge project began during spring 1998 and has been largely completed with the exception of the Business Park. As part of the construction activities, 43 stormwater facilities were constructed and are now fully operational. These facilities either discharge directly to wetlands or a bypass pipe system, or allow the stormwater runoff to infiltrate directly into the ground. In 1998 the initial stages of construction occurred almost entirely within the Mackey Creek basin. In 1999 construction extended southward into the upper portion of the Rutherford Creek watershed. Construction in the Evans Creek drainage is nearing completion currently (Figure 4.1.1.2).

4.0 Monitoring Activity

Monitoring activities were established to determine if development actions caused changes to natural resource processes, creating conditions outside of those predicted in the SEPA documents as reflected in the UPD permits (King County 1996). Monitoring for the project was categorized into five types: Baseline Monitoring; Construction Monitoring; Implementation Monitoring; Facility Performance Monitoring; and, Resource Monitoring (King County 1999). This report highlights resource monitoring efforts and presents results in terms of biological, physical, and chemical assessments of the responses of natural resources to this development.

4.1 Biological Assessment

The primary goal of the comprehensive monitoring plan is to assess impacts to the aquatic systems within and downstream of the Trilogy and Redmond Ridge UPD sites. The monitoring plan includes assessment of chemical and physical changes that may impact aquatic habitat, although these monitoring elements can not indicate the response from living organisms in the streams. Aquatic biota will respond to the conditions in which they live, and therefore can be indicative of cumulative impacts to an aquatic environment (Karr and Chu 1999). Changes to the presence, abundance, and diversity of species residing in a stream can serve as an indicator of how aquatic organisms are impacted by changes in land use in the watershed. Fish and aquatic invertebrates are two commonly used indicators in biological assessments. In addition, wetland-breeding amphibian populations were included in this monitoring program because changes in their populations can have profound effects on wetland ecosystem dynamics (Richter and Ostergaard 1999). Finally, vegetation in the bog wetland system BBC 52 was monitored to assess if changes in hydrology were having adverse effects on the vegetation communities (King County 2001).

4.1.1 Benthic Macroinvertebrates

Overview

Benthic macroinvertebrates are key components of lotic ecosystems providing a functional link between organic matter and fish in aquatic food webs. Analyses of the benthos can provide information reflective of habitat quality, overlying water quality, and potential food resources present. As such, benthic macroinvertebrates are excellent indicators of general stream conditions. They are routinely used in biomonitoring programs due to their high abundance and diversity, limited migration patterns, response to environmental disturbances, and natural population structure unaltered by stocking or harvesting (Rosenberg and Resh 1993, Fore et al. 1996).

No specific projections regarding changes in benthic macroinvertebrate communities were made prior to construction (Herrera Environmental Consultants Inc. 1992, Beak Consultants Incorporated 1995). However, given the predictions for little change in the amount of sediment exiting the sites, no significant stream channel impacts resulting from moderate increases in stream discharge (GeoEngineers 1992, 1995), and only minor overall impacts to water quality (Herrera Environmental Consultants Inc. 1992, Beak Consultants Incorporated 1995) it follows that changes to benthic macroinvertebrate community structure and composition should be minimal if these predictions hold true.

In order to monitor and assess post-development impacts on biotic integrity, benthic macroinvertebrate sampling and analysis was conducted on seven headwater streams draining Redmond Ridge and Trilogy. The benthic index of biological integrity (B-IBI) was chosen to evaluate changes in macroinvertebrate community structure. B-IBI declines of more than 20% violate the thresholds established in the UPD monitoring plans (King County 1999, 2001) and could trigger corrective actions.

Within the Pacific Northwest, the benthic index of biological integrity (PNW B-IBI) has been used extensively since the mid 1990’s to evaluate the biological condition of regional streams (Kleindl 1995, Fore et al. 1996, Karr and Chu 1999, Morley and Karr 2002). The PNW B-IBI is composed of 10 metrics representing taxa richness, tolerance, feeding habits and ecology, and population attributes. These metrics were selected for inclusion in the PNW B-IBI due to their predictable response to anthropogenic disturbance (Kerans and Karr 1994, Kleindl 1995, Fore et al. 1996, Patterson 1996, Karr and Chu 1999).

The streams originating on the two UPDs are zero or first order headwater streams. However, the PNW B-IBI used widely across the Puget Lowland region was developed from data collected from larger, 2nd order and larger streams. Differences in flow regime, energy inputs, and riparian interactions between headwater streams and larger higher order stream systems (Hynes 1970, Vannote et al. 1980, Gomi et al. 2002) shape community diversity and composition and influence B-IBI scores. As a result, the currently accepted PNW B-IBI may not be appropriate for assessing lower order headwater streams.

(Wachter 2003) evaluated 18 headwater streams within the Puget Lowland region and proposed that samples be collected in spring rather than late summer/early fall because of biological and sampling complications related to seasonal drying at several sites. In addition, (Wachter 2003) suggested alternative scoring thresholds to the 10 metrics in the PNW B-IBI (Appendix 4.1.1.1). The resulting headwater B-IBI (HW B-IBI) needs further testing across the Puget Lowland region and across years, however, it was used for comparison purposes in addition to the PNW B-IBI, to assess the potential impacts of human disturbance.

Procedures

Benthic macroinvertebrate sampling and analysis was conducted in spring 1991[1], 1999, 2000, 2001, and 2006 primarily at locations on Adair, Unnamed, Colin North, Colin South, Evans Middle, Evans East, and Rutherford Creeks (Figure 2.0). In addition, fall sampling was carried out in 1997[2] and 2005[3] at a subset of these locations.

Macroinvertebrates were collected following the recommended sampling protocols outlined by (Karr and Chu 1999). At each location, a Surber sampler (500 µm mesh, 0.3 m^2 frame) was used to collect three replicate samples along the midline of a single riffle starting first with the downstream end, then the middle, and finally near the upstream end. Each sample was processed and identified separately without compositing. All large material (e.g., large gravel and woody debris) within the sampling area were scrubbed by hand and examined before being placed downstream. A "weed tool" was used to vigorously agitate the substrate within the perimeter of the frame to a depth of approximately 10 cm, for 60 seconds. Each sample collected was condensed and transferred to the sample container and preserved in the field with 95-100% ethanol (EtOH).

Exceptions to the above procedures took place in 1991 and 2005. In 1991 three replicate samples were collected with a Surber sampler with a larger mesh size (0.9 mm); these samples were combined into one composite sample per station. The samples were initially preserved in 50% isopropanol or methanol (rather than ethanol) and refrigerated for a week. Samples were then drained and preserved with 95% ethanol. In addition, many of the 1991 samples had excessive amounts of fines possibly due to sampling locations in depositional areas rather than true riffles (King County 1993).

In 2005, sampling took place in September coinciding with the larger scale benthic macroinvertebrate sampling conducted by King County. Since these data are not directly comparable to the 1998-2000 UPD data due to the difference in sampling season (i.e., June versus September), the methods were adjusted in order to facilitate comparisons with regional data. As a result, replicate samples were collected from three separate riffles and combined into one composite sample per station following the methods described by (King County 1995). 2005 was the only year that sample collection shifted to one composite per location and sampling in subsequent years resumed collecting and analyzing 3 individual replicates, which is a more appropriate method for evaluating site-specific changes.

After field collection each year, all samples were sent to a contract lab[4] for taxonomic identification to at least genus level whenever possible. In 1991, due to an excessive amount of

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[1] See (Comings et al. 2000), Appendix C for 1991 sample locations and results. These samples were often collected from depositional areas and organisms were only identified to family. Therefore, these data are not comparable to data from subsequent years.
[2] In 1997, Colin South and Rutherford Creeks were not sampled.
[3] In 2005, only Rutherford and Unnamed Creeks were sampled.
[4] In 1991 Goodpasture Aquatic Consulting was the contract laboratory. From 1997-2000 the samples were processed by University of Washington graduate students including Jeff Adams (1997), Sarah Morley (1999), and Heidi Wachter (2000). From 2001 onward the samples were processed by ABR, Inc. – Environmental Research & Services.

fines, the analyst developed and tested a sub-sampling technique that sorted and analyzed 25 to 100% of each sample (King County 1993). From 1997 to 2000 whole samples were sorted and identified. From 2001 onward, 525-550 organisms were subsampled from each replicate[5]. Samples collected in 1991 were identified to family. In all subsequent years, insect taxa were identified to genus when possible except to sub-family for Ceratopogoniae (biting midges) and to family level for Chironomidae (midges), Dolichopodidae (aquatic long legged flies), Dytiscidae (predaceous diving beetles), and Sciomyzidae (marsh flies). Capnidae (slender winter stoneflies), Leuctridae (rolled winged stoneflies), and Odonata (dragonflies and damselflies), were identified to family or order through 2000, but have been identified to genus in subsequent years. Non-insect invertebrates were identified to family, order, or class.

Macroinvertebrates were classified by habit or functional-feeding groups (e.g., predator and clinger) (Merritt and Cummins 1996), and by pollution tolerance and long-lived status (Wisseman 1995). Ten metrics were calculated for each replicate at every sample location. These metric values were averaged and given scores of one, three, or five based on the previously established scoring criteria for the PNW B-IBI and the HW B-IBI. These ten metric scores were summed to provide an overall PNW or HW B-IBI score ranging from 10 to 50.

Results & Discussion

Due to the seasonal sampling differences in 1997 and 2005 (i.e., sampled in the fall as opposed to spring) and the limited taxonomic resolution and variation in sampling procedures for 1991 (i.e., individuals identified to family level rather than genus, and sampling in areas with large amounts of fine sediment), the analysis of benthic macroinvertebrate composition and biotic integrity will be limited to data available from spring sampling in 1999, 2000, 2001, and 2006 for the purposes of this report. Numerous factors including the inherent variability of macroinvertebrate data, limited post-development relative to pre-development data, and sources of sampling and analysis errors magnified by a lack of consistency in field and lab personnel and a small sample size, may make it difficult to identify differences (especially statistically significant differences) in benthic communities or to attribute changes to UPD development. However, biological significance should be emphasized and early trends, which may not be statistically significant, could provide hints of future changes.

Wachter calculated percent landcover for the sub-basin upstream of each sample location (Wachter 2003) (Figure 4.1.1.1). The sub-basin urban landcover for each macroinvertebrate sampling location ranged from 3.5 % in Evans East to 50.7 % in Evans Middle in 1998 prior to UPD clearing and construction. Prior to 2001, no clearing or development took place on Trilogy, portions of which drain to Colin North, Colin South, Adair, and Unnamed Creeks. Construction at Redmond Ridge, which drains to Rutherford, Evans Middle, Evans East, Colin South, and Mackey Creeks, was primarily within the Mackey drainage prior to 2001, and this creek was not monitored as part of the macroinvertebrate sampling effort. Therefore, the 1999, 2000, and 2001 benthic macroinvertebrate data were used to define conditions prior to UPD build-out, hereafter referred to as "pre-development." Post-development data are represented by only one year of data (2006).

[5] ABR, Inc. uses a sub-sampling approach, however if fewer than 525-550 organisms are present in the sample, the entire sample is processed. This was frequently the case for these headwater systems.

B-IBI scores for pre-development biological conditions ranged from 20 (poor) at Evans East in 1999 to 44 (good) at Adair in 2001 for the PNW B-IBI and from 24 (poor) at Colin South and Evans East in 1999 to 46 (excellent) at Adair in 2000 for the HW B-IBI (Table 4.1.1.1). The HW B-IBI scoring thresholds routinely increased scores relative to the PNW B-IBI scores by 2 to 6 points, although in four cases the HW scores were two points lower and there were ten instances where PNW and HW scores were the same. Differences in the two B-IBIs can be attributed to the lower scoring thresholds for percent tolerant, percent predator, and clinger taxa richness, in addition to the higher scoring thresholds for long-lived taxa richness for the HW B-IBI (Appendix 4.1.1.1). For additional background on the differences between the PNW and headwater B-IBIs see (Wachter 2003).

Averaged across the three pre-development years, the HW and PNW B-IBI scores ranged from poor to good (See Appendix 4.1.1.2 for description of scoring criteria). Adair was the only site classified as good, Colin North and Evans East were classified as poor, and the remaining sites were classified as fair (Table 4.1.1.2). The post-development HW and PNW B-IBI scores ranged from 18 (poor) at Colin North to 36 (fair) at Adair.

Macroinvertebrate assemblages have frequently been used to discern the impacts of urbanization (Kleindl 1995, Rossano 1995, Morley and Karr 2002, Wachter 2003). However, despite the large variation in urbanization (i.e., 3-50%) (Figure 4.1.1.1) between UPD sub-basins prior to development, there were no correlations between sub-basin percent urbanization and PNW or HW B-IBI scores (Figure 4.1.1.2). Similarly, there were no correlations between percent development within each UPD sub-basin and 2006 PNW or HW B-IBI scores. Various legacy effects (e.g., many of these sites were previously logged) could be contributing to the poor relationship between urbanization and biotic integrity (Harding et al. 1998). Or perhaps defining urbanization by a single number (i.e., percent urban) is an over simplification and other landscape scale measures that better capture the extent and pattern of urbanization including the intensity and location of development relative to the stream (e.g., percent urban within a buffer, percent impervious surface, frequency of buffer gaps, etc.) may be more appropriate. However, additional analyses of landscape scale influences on local biotic conditions were beyond the scope of this project. While this result may be unexpected, it is a reminder that direct biological measures may provide valuable information about catchment condition and biotic integrity that cannot necessarily be discerned with use of satellite image surrogates that do not capture cumulative effects and multiple influences (Fore et al. 2001).

Looking at each of the individual metrics that comprise the B-IBIs can provide additional information regarding community changes that cannot be discerned from the total scores alone (Figure 4.1.1.3). Pre-development B-IBI values were compared to 2006 values and instances where the 2006 mean was outside the variability[6] of the pre-development data were flagged. Clinger richness was the only individual metric that showed declines in every creek. This was also the only metric where 2006 values dropped below pre-development variability at Evans East where no UPD development has taken place to date. However, clinger richness for Evans East

[6] For each metric or total B-IBI score, the pre-development variability range was defined as the 1999-2001 mean of the parameter in question minus the standard deviation (SD) to the mean plus the SD, unless otherwise specified.

dropped only 28% compared to declines of between 35 and 63% for the other sites. Invertebrates classified as clingers typically prefer clean, stable substrate (Merritt and Cummins 1996). Therefore, the decline in clinger taxa richness across locations could be in response to changes in substrate stability and increased sedimentation that limit access to microhabitats vital to the survival of many clinger species.

Four of the six streams within UPD sub-basins showed declines in both overall and Plecoptera (stonefly) taxa richness, while three of the six locations had declines in Trichoptera (caddis fly) taxa richness (Figure 4.1.1.3, Table 4.1.1.3). Ephemeroptera (mayfly) taxa richness decreased in Colin South and Unnamed. Percent tolerant individuals and percent dominance by the three most abundant taxa were the only two measures expected to increase in response to growing urbanization (Karr and Chu 1999). Yet, only Adair and Colin North showed increases in percent tolerant individuals and Colin South showed an increase in percent dominance. The three remaining metrics (long lived and intolerant taxa richness and relative abundance of predators), did not show declines between pre- and post-development periods. However, Adair and Evans Middle did have increases in percent predators and Colin South had an increase in intolerant taxa richness, which is the opposite of what is expected in response to increasing urban pressures (Figure 4.1.1.3).

In 2006, Colin North showed declines beyond pre-development variability in five individual metrics while Unnamed, Adair, and Evans Middle showed declines in four metrics each (Figure 4.1.1.3, Table 4.1.1.3). 2006 total PNW and HW B-IBI scores dropped beyond the variability of pre-development scores for Colin North and Evans Middle, and scores also dropped for the HW B-IBI for Unnamed Creek (Figure 4.1.1.3).

Comparing the mean pre- and post-development macroinvertebrate metrics and B-IBI scores across all seven locations suggests that there have been substantial changes in community composition and structure over time. Taxa richness metrics including overall, Ephemeroptera, Plecoptera, Trichoptera, clinger and long-lived richness all show significant ($p<0.05$, ANOVA for all except long-lived which used a Kruskal-Wallis) declines from pre- to post-development (Figure 4.1.1.4). While statistically significant, some of these differences only represent a decline of about one taxa from 4 to 3 or 5 to 4 (i.e., Ephemeroptera, Plecoptera, and Trichoptera richness) or of less than one taxa (i.e., long-lived richness), which may or may not be ecologically meaningful. However, B-IBI scores shift between scoring categories based on a change in richness of only 2 to 4 taxa (Appendix 4.1.1.1), so the loss of one species or genus, especially when considered cumulatively, could be an important indication of declining conditions.

In contrast to the small declines observed in individual EPT richness categories, clinger richness and overall taxa richness showed larger declines. Clinger richness at all sites dropped from over 10 to under 6 while total richness decreased from more than 26 taxa to about 20 (Figure 4.1.1.4). As previously indicated, clingers rely on interstitial spaces between substrate and are sensitive to fine sediment that reduce the complexity and availability of these habitats (Fore 2002). As human activities impact the watershed, native taxa tend to decline (Karr and Chu 1999). Declines in total taxa richness are often associated with increases in dominance of a few species or higher relative abundance of tolerant individuals.

Contrary to expectations in urbanizing basins, intolerant richness and relative abundance of predators both increased significantly ($p<0.01$, Kruskal-Wallis). Neither overall B-IBI score displayed statistically significant changes between development periods.

In summary, overall declines between development periods in several richness metrics across sites combined with declines of individual metrics at specific sites suggest that biotic condition may be declining in several streams affected by UPD development. Declines in clinger taxa richness may be the biggest red flag warning of continued deterioration. Colin North, Evans Middle, and Unnamed Creek have shown the greatest overall decrease in biotic condition and should be monitored closely. However, for all of these initial results, one year of post-development comparison is likely inadequate to establish trends and further post-development monitoring is necessary. See Appendix 4.1.1.3 for B-IBI scores for all sampling locations.

Threshold Exceedance Criteria

Benthic macroinvertebrate threshold exceedance criteria are violated if there was a 20% or more decline in B-IBI scores (King County 1999). B-IBI scores were chosen as a threshold rather than total abundance because abundance is generally not considered an indicator of stream health for macroinvertebrates. Furthermore, abundance may reflect differences in sampling and analysis procedures rather than differences in biotic condition[7]. The monitoring plans did not specify whether to use the PNW or the HW B-IBI, therefore both were assessed.

The threshold exceedance criteria were exceeded for the HW B-IBI at Unnamed (21.0%) and for both the PNW and HW B-IBIs at Colin North (27.0% PNW, 35.7% HW) (Table 4.1.1.2). Evans Middle exhibited an 18.8% change in the HW B-IBI; however, this decrease did not exceed the threshold criteria.

A statistical power analysis by (Fore et al. 2001) demonstrated that a change in B-IBI of seven or more points (out of 10-50 total possible) represents a statistically significant and biologically meaningful change in biotic condition. Using this criterion and the HW B-IBIs, Colin North and Unnamed Creek had a biologically significant decrease in stream health with declines of 10.0 and 8.0 points, respectively. Evans Middle had a decrease of 6.0 for the HW B-IBI, just below the criterion. For the PNW B-IBI, no locations had a decrease of more than 7 points, although Colin North and Unnamed Creek just missed the threshold with drops of 6.7 and 6.0, respectively (Table 4.1.1.2).

Corrective Actions

B-IBI scores appear to be in decline at several creeks draining from the UPDs. Some of these declines are beyond the range of variability established during three years of pre-development data from 1999 to 2001. Declines in the B-IBI on Unnamed and Colin North exceed established threshold criteria for HW B-IBI scores. However, one year of post-development data is

[7] The Redmond Ridge Monitoring Plan (King County 1999) uses the B-IBI threshold exceedance criteria, whereas the Trilogy Monitoring Plan (King County 2001) refers to a 20% decline in total abundance. However, due to the justification presented in the report text, B-IBI was chosen for threshold exceedance criteria for all macroinvertebrate data regardless of which UPD they drain.

insufficient to define a trend. These streams should continue to be monitored closely, but no corrective actions are warranted at this time. The monitoring plans (King County 1999, 2001) mandate that sampling duration continue until at least three years following 75% build-out in each basin. For most basins this entails sampling through at least 2008 or 2009 and to ensure at least three years of post-development data to compare with pre-development data monitoring must continue through at least 2008 at all locations.

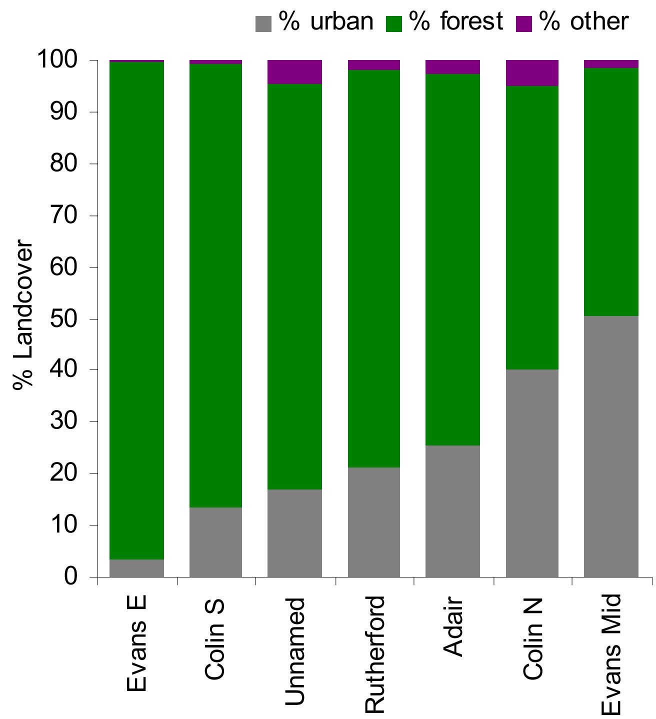

Figure 4.1.1.1 Percent urbanization in each sub-basin (taken from Wachter 2003).
"Urban" includes intense, grassy, and forested urban categories. "Other" includes grass, water, and bare land categories. Landcover classification is based on 1998 LandSat images and represents pre-development conditions.

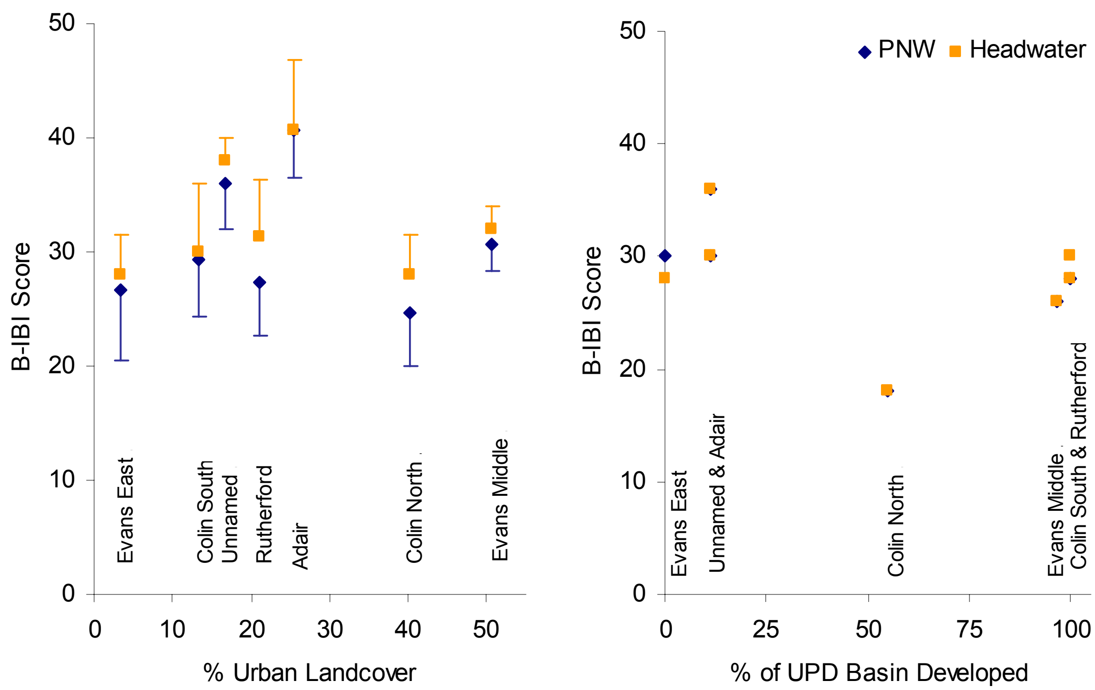

Figure 4.1.1.2 Mean pre-development PNW and HW B-IBI scores 1999, 2000, and 2001 (left) compared to upstream sub-basin urban landcover (1998) and 2006 post-development.
B-IBI scores (right) compared to percent UPD build-out. Error bars represent one standard deviation. Percent UPD build-out estimated from development phasing maps (Kpff Consulting Engineers 2004, Otak Incorporated 2006).

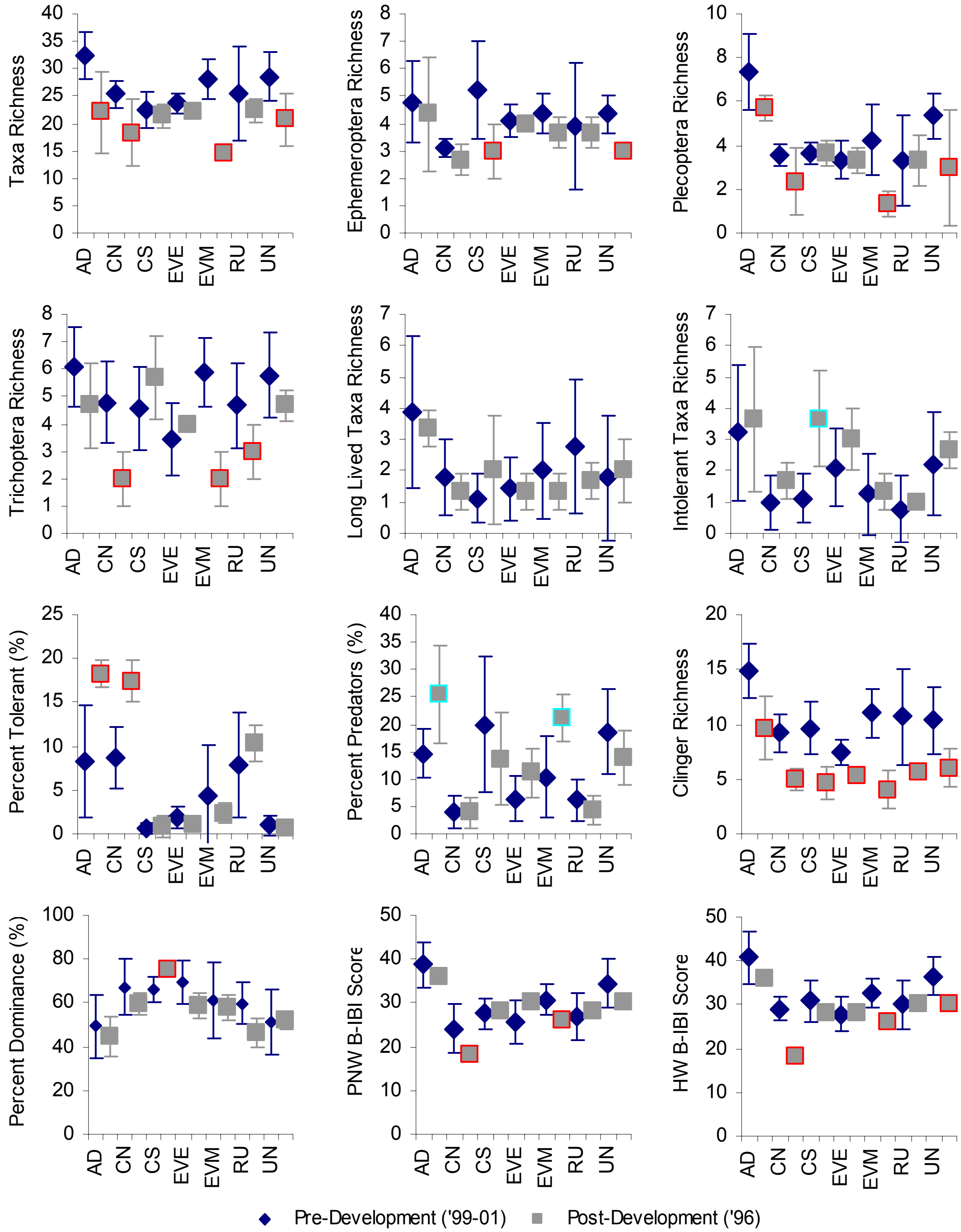

Figure 4.1.1.3 Mean values for the 10 B-IBI metrics and the two B-IBI scoring systems for pre- and post-development conditions from seven streams.

Pre-development values were averaged from 1999, 2000, and 2001. Post-development data represent the average of three 2006 replicates. Error bars represent one standard deviation. Post-development points outlined in red or turquoise have mean values outside the pre-development standard deviation variability range. Red outlines indicate a change in the direction of anticipated response to increasing urbanization, whereas turquoise represents an unanticipated response to urbanization. For example, all metrics except percent tolerant and percent dominant were expected to decrease in response to urbanization (Karr and Chu 1999).

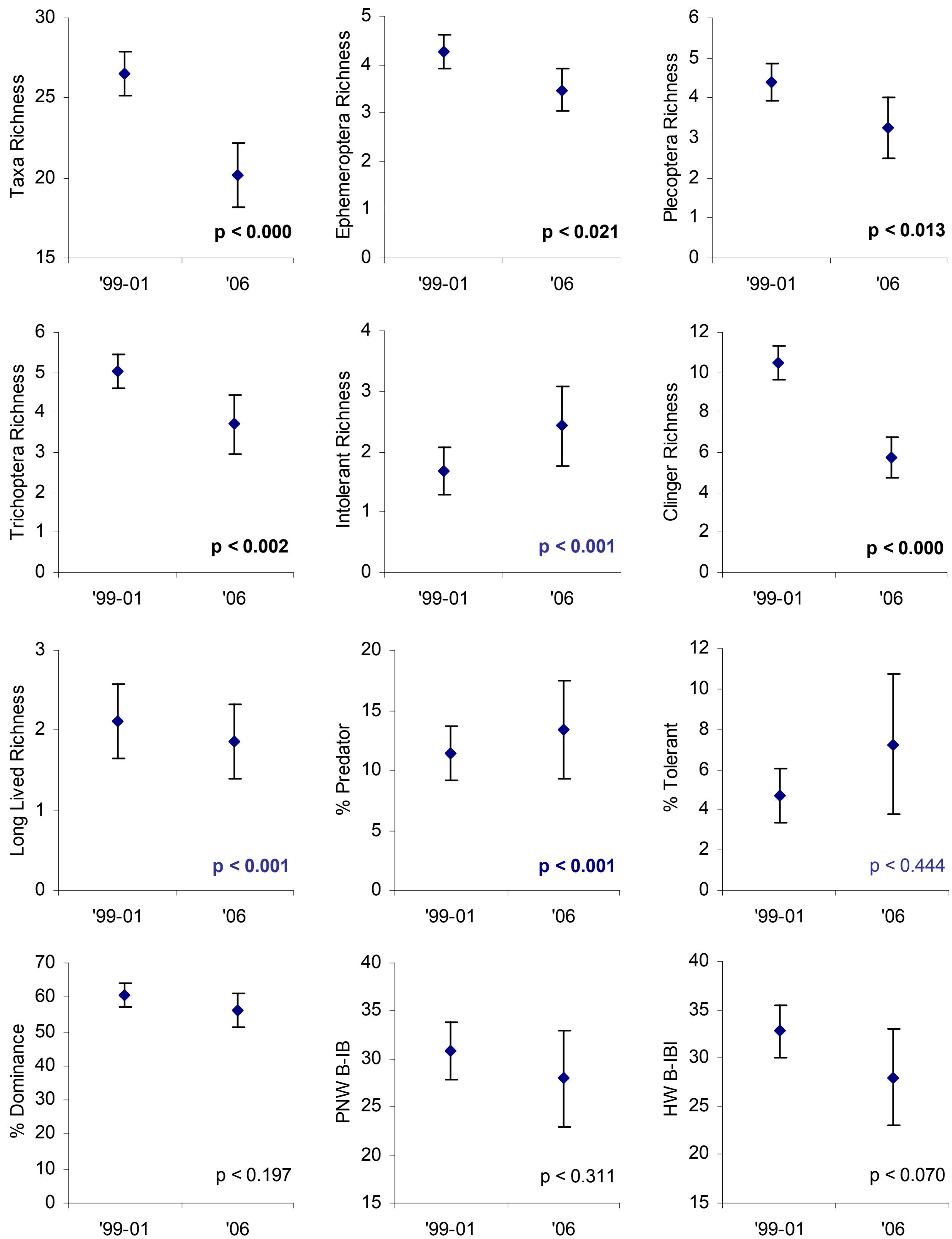

Figure 4.1.1.4 Comparison of pre- and post-development means of individual metric scores and total B-IBI scores for all sample sites combined.

Error bars represent 95% confidence intervals. P values are based on one-way ANOVA tests for variables meeting normality assumptions (p value in black). For variables not normally distributed, means were compared with the non-parametric Kruskal-Wallis test (p values in blue). P values in bold print represent significant differences at the $p < 0.05$ level.

Table 4.1.1.1 Total scores for the PNW and HW B-IBIs.

	1999		2000		2001		2006	
	PNW	HW	PNW	HW	PNW	HW	PNW	HW
Adair	36	34	42	46	44	42	36	36
Colin North	22	26	30	32	22	26	18	18
Colin South	24	24	30	30	34	36	28	28
Evans East	20	24	32	30	28	30	30	28
Evans Middle	28	30	32	34	32	32	26	26
Rutherford	30	36	22	26	30	32	28	30
Unnamed	36	36	40	40	32	38	30	30

Table 4.1.1.2 Pre- and post-development B-IBI scores for the PNW and HW B-IBIs.
n = 3 for pre-development data (1999, 2000, 2001) and n = 1 for post-development data (2006). Negative percent change values indicate a decline in B-IBI score between pre- and post-development periods and bold values surpassed threshold exceedance criteria (i.e., a decline in B-IBI scores of 20% or more).

	Pre-development			Post-development			Percent change	
	PNW B-IBI	HW B-IBI	Biological Condition	PNW B-IBI	HW B-IBI	Biological Condition	PNW B-IBI	HW B-IBI
Adair	40.7	40.7	Good	36	36	Fair	-11.5	-11.5
Colin North	24.7	28	Poor/Fair	18	18	Poor	**-27.0**	**-35.7**
Colin South	29.3	30	Fair	28	28	Fair	-4.5	-6.7
Evans East	26.7	28	Poor/Fair	30	28	Fair	12.5	0.0
Evans Middle	30.7	32	Fair	26	26	Poor	-15.2	-18.8
Rutherford	27.3	31.3	Fair	28	30	Fair	2.4	-4.3
Unnamed	36	38	Fair/Good	30	30	Fair	-16.7	**-21.1**

Table 4.1.1.3 Change matrix for individual metrics and total B-IBI scores between pre- and post-development periods.
A bold "Y" indicates that in 2006 the metric changed beyond the variability (the mean + or – the standard deviation) of the pre-development data in the direction expected in response to urbanization (Karr and Chu 1999).

Individual Metric	Expected response to urbanization	Adair	Colin North	Colin South	Evans East	Evans Middle	Rutherford	Unnamed
Taxa Richness	Decrease	**Y**	**Y**	N	N	**Y**	N	**Y**
E Richness	Decrease	N	N	**Y**	N	N	N	**Y**
P Richness	Decrease	**Y**	**Y**	N	N	**Y**	N	**Y**
T Richness	Decrease	N	**Y**	N	N	**Y**	**Y**	N
Long-lived Richness	Decrease	N	N	N	N	N	N	N
Intolerant Richness	Decrease	N	N	N	N	N	N	N
% Tolerant	Increase	**Y**	**Y**	N	N	N	N	N
% Predators	Decrease	N	N	N	N	N	N	N
Clinger Richness	Decrease	**Y**	**Y**	**Y**	**Y**	**Y**	**Y**	**Y**
% Dominance	Increase	N	N	**Y**	N	N	N	N
PNW B-IBI	Decrease	N	**Y**	N	N	**Y**	N	N
HW B-IBI	Decrease	N	**Y**	N	N	**Y**	N	**Y**

4.1.2 Amphibians

Overview

The critical importance of wetlands for flood control, wildlife habitat, water quality improvement, and as part of the general hydrologic cycle is well established (National Research Council 1992, 2001, 2002). However, urbanization has been demonstrated to impact wetlands in numerous direct and indirect ways causing habitat loss, hydrologic changes (Reinelt and Taylor 2001), and altered water quality (Horner et al. 2001b). In King County, a wide array of amphibians utilize wetlands during some life stage with eight native species breeding in lentic habitats (Richter and Azous 2001).

UPD development was predicted to reduce the amount and diversity of native animals, but few species were expected to be eliminated from the area (Raedeke Associates Inc. 1993, 1995). Direct alteration of wetlands on the Trilogy and Redmond Ridge UPDs was almost entirely avoided by retaining 60 m average buffering. However, forest lands adjacent and up to 1000 m from wetlands have been demonstrated to provide essential non-breeding habitat, food, cover, and migration corridors for native amphibian species (Richter and Azous 2001). While not specifically predicted in the EIS process, native amphibian populations may be at risk due to the loss of upland habitat outside of the 60 m buffers.

Indirect alterations to wetlands were predicted to include increased surface water runoff via stormwater management systems supplementing the dominant pre-development groundwater and precipitation sources. Despite potential increased surface runoff, changes in wetland water levels were not predicted to be of sufficient magnitude to have substantial impacts on wetland plant communities and related amphibian breeding habitat[8] (Raedeke Associates Inc. 1993, 1995).

Monitoring breeding amphibian populations may provide early warning signs regarding wetland habitat, hydrologic changes, and quality deterioration since they are considered sensitive indicators of changes in water regimes, sedimentation, and water quality (Reinelt et al. 1998, Richter et al. 1998, Richter and Azous 2001). To determine whether amphibian populations and the wetlands which are required for breeding are adequately protected by the Redmond Ridge and Trilogy UPDs, amphibian egg mass surveys were conducted twice each spring in six locations on five wetlands. Adult and larval use of these wetlands were also identified. Declines in amphibian richness or abundance in consecutive years or egg-mass mortality increases of more than 20% violate the thresholds established in the UPD monitoring plans (King County 1999, 2001) and could trigger corrective actions if linked to UPD activities.

Procedures

Amphibian breeding surveys

Amphibian breeding surveys were conducted twice each spring since 2000 at SR 24c, BBC 52, and two locations on BBC 45, since 2001 at BBC 44, and since 2002 at BBC 26 (Figure 2.0).

[8] The exception to this statement was SR 24b where November to June water levels were predicted to be approximately 0.1 m higher after development compared to pre-development conditions (Raedeke Associates Inc. 1993).

The wetlands were located on the Redmond Ridge and Trilogy UPDs. These surveys generally followed methods outlined by (Thoms et al. 1997) and modified for use for the King County volunteer wetland-breeding amphibian monitoring program (Richter and Ostergaard 1999). Two annual egg mass surveys were typically conducted by two biologists in the first three weeks of March and the second two weeks of April, varying slightly from year-to-year based on the timing of oviposition. Rain, high winds, and overcast conditions were avoided whenever possible to maximize visibility through the water column. Surveys were conducted by wading through the wetlands or using a small float tube for water deeper than 1 m. Egg masses were identified by species and percent mortality per clutch estimated within eight categories (0%, 1-5%, 6-25%, 26-50%, 51-75%, 76-95%, 96-100%, or partially hatched). Larvae, metamorphs, juveniles, and adults were also identified to species if possible and calling frogs were noted when heard.

As part of King County's Volunteer Amphibian Breeding Monitoring Program, surveys were conducted from 1995 to 2000 on BBC 26, from 1994 to 1998 on BBC 52, and in 1998 on BBC 45. However, the surveys were conducted within different areas of the same wetlands compared to the current UPD related monitoring surveys. Previous studies under controlled field conditions determined that the northwest quadrant was the preferred ovisposition habitat for Northwestern salamanders (Richter and Roughgarden 2005) (Figure 4.1.2.1). Therefore, abundance data from different sampling areas on the same wetland are not comparable. However, these data may be useful for evaluating changes in species richness and composition.

Aquatic funnel trapping

During the summer of 2006 a pilot aquatic funnel trapping study was conducted to augment egg mass surveys. Two wire minnow traps and two collapsible aquatic nylon mesh traps were set at the six annual amphibian monitoring locations (Figure 2.0) (Adams et al. 1997). At SR 24c and BBC 26 one wire and one mesh trap were submerged about 1 m beneath the water surface without resting on the substrate and one wire and one mesh trap were placed at the surface of the water so that the upper parts of the traps were always exposed to the air enabling pulmonary in addition to cuticular respiration. In the remaining wetlands all the traps were placed at the surface. The traps were each baited with 10 pieces of Purina Whisker Lickens ® salmon flavor cat food and were left overnight before being checked and collected the next day. Animals found in the traps were identified to species and life stage. Animals were measured when possible, and released immediately on-site.

Supplemental data

Water quality parameters were measured at most amphibian monitoring locations including metals in 2006. These data are primarily reported in the water quality portion of this report, except in this section in the context of wetlands in which high egg mass mortalities were identified (i.e., in BBC 45N).

Results & Discussion

Richness

A total of six amphibian species were identified at five Redmond Ridge surveyed wetlands. Egg masses of four native species were commonly found during the monitoring periods. These included Northwestern salamanders (*Ambystoma gracile*), Northern red-legged frogs (*Rana* *aurora*), Pacific tree frogs (*Pseudacris regilla*), and long-toed salamanders (*Ambystoma macrodactylum*). In addition, rough-skinned newts (*Taricha granulosa*) and non-native adult American bullfrogs (*Rana catesbeiana*) were occasionally heard or seen (Figure 4.1.2.1). No western toads (*Bufo boreas*) or Oregon spotted frogs (*Rana pretiosa*), a Washington State endangered species and Federal candidate species whose range formerly encompassed the Puget Sound region, were observed. Survey results can be used to determine species presence, but failure to see a species does not ensure its absence especially for long-toed salamanders, rough-skinned newts, and bullfrogs, which have eggs that are difficult to find (long-toed salamanders and rough-skinned newts) or active periods that are before (long-toed salamanders) or after (bullfrogs) our survey period.

Species richness and composition at both BBC 45 locations was remarkably consistent from year-to-year. Pacific treefrogs and Northwestern salamanders were found in BBC 45N in all years sampled and Northwestern salamanders, Northern red-legged frogs, and Pacific treefrogs were found in BBC 45S from 2001 to 2006 (Figure 4.1.2.1). Richness was variable in BBC 26 and SR 24c ranging from 2-4 species and only Northwestern salamanders and Pacific treefrogs were found every year. Richness decreased in BBC 44 from five species in 2001 to only two species (Northwestern salamander and Pacific treefrog) in 2005 and 2006. Richness may have increased in BBC 52 where Northwestern salamanders, Pacific treefrogs, and Northern red-legged frogs were found nearly every year. In 2006, long-toed salamanders, rough-skinned newt, and American bullfrog were also observed at this location.

Implications of low egg mass abundance

American bullfrogs, long-toed salamanders, and Northern red-legged frogs have never been observed in high numbers based on egg mass surveys in most of these wetlands (i.e., BBC 44, BBC 45, BBC 26, SR24c). In part, this is a result of the sampling protocols, which were not timed to capture American bullfrogs and long-toed salamanders. However, the protocols were designed to capture complete censuses of Northern-red legged frogs and the low abundance numbers likely reflect small population sizes.

A maximum of two adult and one larval bullfrog have been observed in any of the wetlands within the same season. Greater than five long-toed salamanders have been observed only three times (15 and 8 in BBC 26 in 1995 and 1999, respectively and 15 in BBC 52 in 2004) and only those observed in BBC 52 in 2004 were found within the same wetland area observed from 2001 to 2006 (Table 4.1.2.1). Other than in BBC 52, more than five Northern red-legged frog egg masses have only been observed once (10 in BBC 45S in 2000). Due to these low breeding numbers, it is difficult to draw any conclusions from the disappearance of these species aside from BBC 52 where Northern red-legged frog egg mass abundance ranged from 13 to 89 since 2000.

Fluctuations from zero to five or fewer egg masses for American bullfrogs, long-toed salamanders, and Northern red-legged frogs likely accounts for much of the richness variability within the monitored wetlands. However, without a longer pre-development record it is difficult to determine whether we are monitoring non-viable populations under the threat of local extinction, dispersal and range expansion, or simply natural variation for low-abundance populations. For example, in BBC 44 where we observed an apparent decline in native species richness (Figure 4.1.2.1, Table 4.1.2.2), no more than two long-toed salamander egg masses were ever observed in a given year and no Northern red-legged frog egg masses have ever been recorded (3 adults were observed in 2001). Therefore, the high species richness in 2001 (four native species and the American bullfrog) may not actually represent substantially different population dynamics compared to recent years when only Northwestern salamanders and Pacific treefrogs were observed. In fact, the data suggest that Northwestern salamanders and Pacific treefrogs have always been the primary breeding species at BBC 44.

Total egg abundance
Within each wetland, total abundance of egg masses for all species combined fluctuated considerably between years with no steady increasing or decreasing trends (Figure 4.1.2.2). At BBC 26 total abundance ranged from 243 in 2004 to 377 in 2006 with more than 300 in both 2002 and 2005. SR 24c had fewer than 150 egg masses in 2000 and 2006, but more than 270 in all interim years. BBC44 abundances peaked at 214 in 2006 with a low of 65 egg masses in 2005 and a range from 149 to 202 between 2001 and 2004. Egg mass abundance at BBC 52 crashed in 2003 to a low of 66 compared to more than 217 the year before and 248 the year after. Abundance dipped again in 2005 to 112 only to rebound in 2006 to a peak of 265. Compared to all wetlands, abundance at both BBC 45 locations were low in all years. BBC 45N abundance ranged from 19 egg masses in 2000 to 87 in 2006 whereas only one egg mass was found in BBC 45S in 2004 compared to 36 in 2003. In all other years between 11 and 21 egg masses were recorded at BBC 45S.

Fluctuations also were not consistent between wetlands within the same year. For example, egg mass abundance of all species combined was high in 2006 relative to other years in BBC 26, BBC 44 and BBC 45N, however abundance was relatively low in SR 24c and BBC 45S in 2006 (Figure 4.1.2.2). Similarly, 2005 abundance was low in BBC 45S, BBC 52, and BBC 44, but relatively high in BBC 26 and SR 24c. Natural variability and patterns of amphibian breeding location may explain some of these variations. However, precipitation and temperature, which we expect to be similar at each of these sampling sites due to their close proximity within 3 km of each other, cannot explain these within-year fluctuations.

Species-specific egg abundance & mortality: Northwestern salamanders, Northern red-legged frogs, and Pacific treefrogs
Looking at species-specific declines, Northwestern salamanders have declined for two subsequent years at three wetland locations (Figure 4.1.2.3). In BBC 26, Northwestern salamander egg masses declined from 329 to 233 to 214 between 2002 and 2004. Similar declines were observed in both BBC 45 locations (BBC 45N declined from 80 to 46 to 39 and BBC 45S declined from 20 to 3 to 0). However, in each of these basins UPD clearing did not begin until 2005. Therefore, these declines cannot be attributed to UPD development.

The 49% decline in SR 24c egg masses from a peak of 297 in 2005 to 146 in 2006, the lowest number of egg masses since 2000, should be monitored closely to determine whether this is the beginning of a declining trend. Clearing began in 2004 within this basin with large-scale construction and building throughout 2005. Therefore, the large drop in 2006 could be in response to construction impacts.

Northwestern salamander egg mortality at BBC 45N was above 30% in 2001 and above 50% in 2002 and 2006. Mortality above 5% may be cause for concern, therefore the levels at BBC 45N are troubling. This is the most disturbed of any of the UPD wetland locations with a power line easement crossing the sampling area and Novelty Hill Road crossing just downstream, however UPD clearing only began within the basin in April 2005. Concentrations for thirty metals were measured for samples collected in January, February, and April 2006 at all stream and wetland water quality locations (Figure 2.0). None of the BBC 45N samples exceeded state acute or chronic standards that could help explain these high mortalities. Additional water quality testing may be warranted in future years to help identify possible sources for increased egg mortality.

Northern red-legged frog egg clutches were only sizeable for abundance and mortality analysis in BBC 52. Abundance never declined in two consecutive years increasing steadily from a low of 20 in 2000 to a peak of 89 in 2004 and then leveling off at 42 in 2005 and 2006 (Figure 4.1.2.4). Egg mass mortality was above 14% in 2000, 2001, and 2005. Clearing within the BBC 52 basin did not begin until after 2005 sampling hence all data were pre-development except for 2006 construction data. Therefore, the elevated egg mass mortalities in 2000, 2001, and 2005 cannot be attributed to UPD development.

Pacific treefrog abundance was low in BBC 44 and BBC 45N, never exceeding 10 egg masses (Figure 4.1.2.5). Numbers within BBC 45S have been declining since 2000, but the low levels throughout (i.e., never more than 16 egg masses) may suggest that this was never optimal Pacific treefrog breeding habitat. In SR 24c, BBC 52, and BBC 26 Pacific treefrog abundance was much higher in one year compared to all other years. In SR24c, Pacific treefrog abundance from 2002 to 2006 was less than half what it was in 2000 (41) and less than a quarter of 2001 levels (82). In BBC 52, the treefrog population spiked in 2003 with 559 egg masses, well over double the number observed in 2001 and quadruple the number of egg masses recorded in all other years. In BBC 26, 2005 was the peak year with 43 clutches and no other year exceeded 22.

Pacific treefrog egg mass mortality was generally low and below 10% with only a few exceptions (Figure Amphib-5). The high mean mortality in BBC 45S in 2005 is due to 100% mortality in one of the four egg masses. Similarly, the 12% mortality in SR24c can be explained by one egg mass had 95-100% mortality while the other seven had 0%. The proportion of egg masses with high mortality has increased since 2003, the peak abundance year resulting in a trend of increasing estimates for mean mortality from 2004-2006. This trend should be observed closely in future years and if it continues corrective measures could be required.

Funnel trapping observations

During the July 2006 funnel trapping study, at least one Northwestern salamander paedomorph was caught in each sampling location with a high of 17 found in BBC 45N (Table 4.1.2.1). Pacific treefrog larvae were captured in BBC 26 and BBC 52 and one bullfrog larva was caught in BBC 52. The only fish observed during this study was one three-spined stickleback in BBC 44. This trapping was conducted to try and identify what predators are present in each wetland that may be influencing amphibian population dynamics. However, a more intensive study with greater sampling from all habitats present is required to better assess predator presence or absence.

Threshold Exceedance Criteria

Amphibian threshold exceedance criteria are violated if:

1. Any one species that formerly bred in the wetland disappears
    - Amphibians were only unrecorded in one wetland where they previously occurred, namely BBC26. Specifically, the disappearance of red-legged frogs in BBC 26 should be monitored, however no more than four egg masses (2005) were ever recorded in one year.
    - Northern red-legged frogs were never observed breeding in BBC 44. Their presence as noted in Figure 4.1.2.1 was based on adult observations. A single Northern red-legged frog egg mass was observed in 2001 in SR 24c.
    - Long-toed salamanders appeared and disappeared in BBC 44, but always in low numbers (i.e., fewer than 2 egg masses) and they were only observed once in 2004 in SR 24c. As previously mentioned, the peak breeding season for long-toed salamanders is generally mid January to early February before our surveys take place.

2. Egg mortality increases by 20% or more for Northwestern salamanders or red-legged frogs[9].
    - BBC 45N had extraordinarily high mortality levels (>30% on average) in 2001, 2002, and 2006 for Northwestern salamanders. The 2001 & 2002 mortality spikes occurred before any major UPD-related clearing and development within the basin. Even prior to UPD development, this was the most disturbed onsite wetlands with a power line easement crossing through the sampling area and Novelty Hill Road crosses just downstream. Nevertheless, mortality rates and associated water quality parameters should be monitored closely in future years.
    - Pacific treefrog egg mortality is not specifically called out as a threshold exceedance criteria[10], but steady increases in egg mortality from <5% in 2004 to more than 20% in 2006 have been observed in BBC 52.

3. Significant reduction in abundance of egg masses of Northwestern salamanders or Northern red-legged frogs for two consecutive years.
    - In SR 24c, Northwestern salamander egg mass abundance dropped by almost half in 2006. This location should be monitored closely to ensure that recent habitat conversion due to clearing and development within this basin is not causing a continued decline.

Corrective Actions

Amphibian populations at the UPD wetlands fluctuated greatly from year to year. Factors contributing to amphibian declines could include hydrologic changes leading to large water level fluctuations (Richter and Azous 2001), changes from herbaceous habitat (i.e., grasses, herbs, rushes, and sedges) to cattail and scrub-shrub habitat types within the emergent zone (Alford and

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[9] Pacific treefrogs were erroneously called out in the Trilogy Monitoring Plan (King County 2001). Northern red-legged frogs and Northwestern salamanders represent two different life strategies of pond-breeding amphibians with eggs that are easy to identify and detect.

[10] See previous footnote.

Richards 1999), introduction of predators, epidemics or parasites (Daszak et al. 2003), meteorological conditions including precipitation and climate change (Carey and Alexander 2003), and /or water quality problems (Marco et al. 1999). Some of these factors could be attributed to the increasing urbanization of the basin (e.g. hydrological changes that influence plant community structure and available habitat and water quality problems), but many cannot. The 2006 disappearance of red-legged frogs in BBC 26 and other trends exceedances noted above should be monitored closely, but at this time no corrective actions are warranted.

During construction in March 2006, there was a surface connection to BBC 26 from turbid waters pumped from a stormwater pond through a spreader device. The level spreader is a recommended erosion control (King County 2005, Washington State Department of Ecology 2005). However, in the future we recommend that:

1) The water is pumped at a rate and volume slow and small enough to allow complete infiltration. This will ensure that there is no direct connection with the wetland surface waters as was observed during spring 2006 (Appendix 4.1.2.1).
2) A float is attached to the hose to ensure that surface water is being pumped rather than bottom water, which may be more likely to draw out the very sediments the pond was designed to settle out.

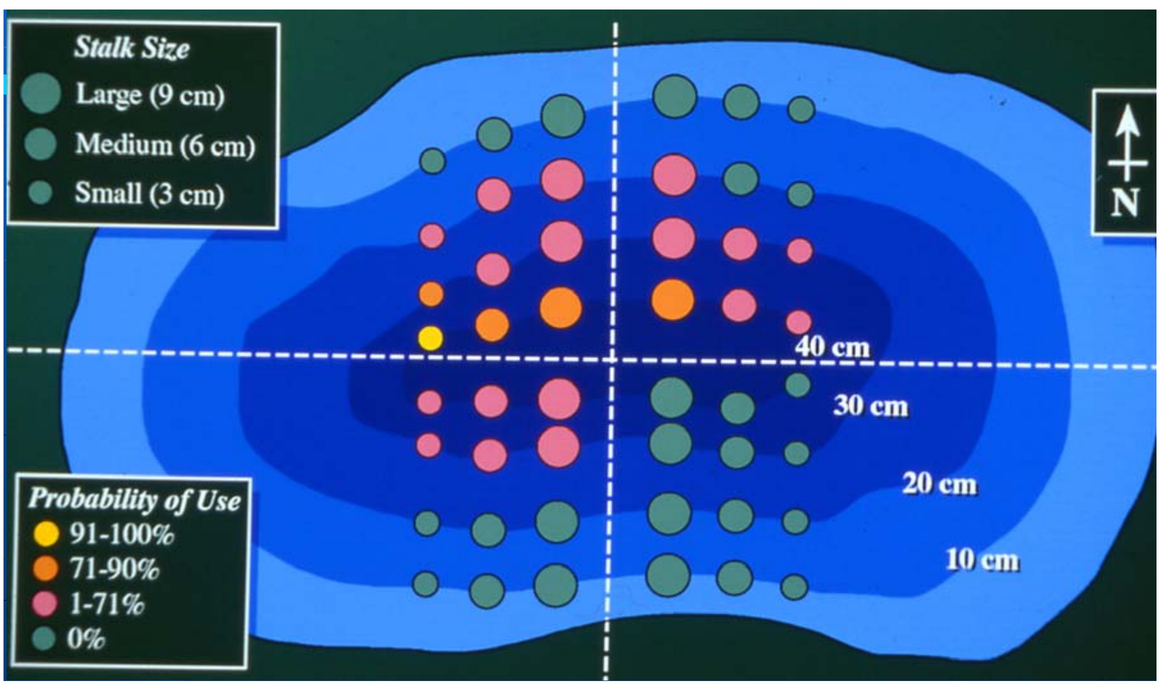

Figure 4.1.2.1 Probability of finding Northwestern salamanders spawning within different quadrants of a wetland based on depth and stalk size (Richter and Roughgarden 2005). The stalk size is in mm, not cm as the legend erroneously reports.

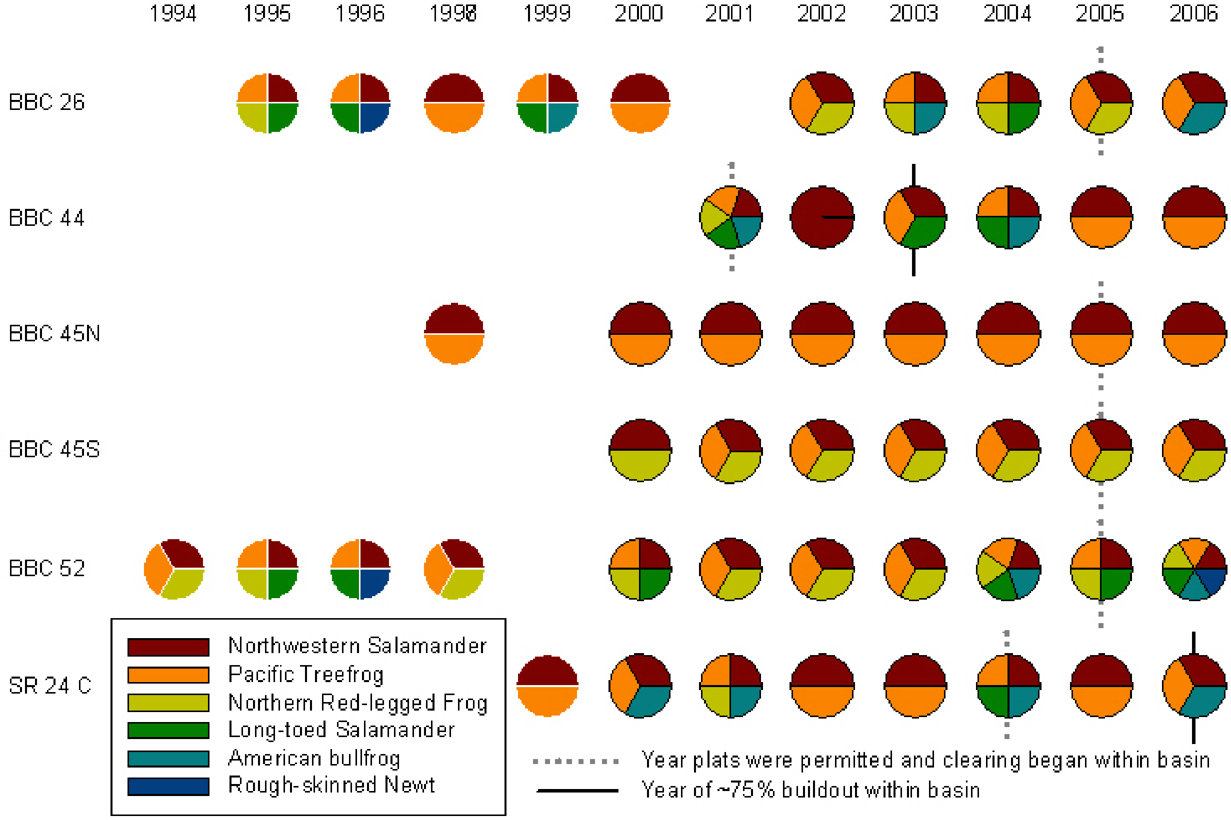

Figure 4.1.2.2 Species composition of amphibians at various UPD wetland sites.
No American bullfrog or rough-skinned newt egg masses were ever observed during these surveys, but adults were noted on several occasions. The proportion of the pie assigned to each species is based on the species richness and is not representative of abundance. The pies sub-divided by white lines (1994-1999 & BBC 26 2000) represent data collected by volunteers and in all cases except for BBC 45N these data were collected at a different location from subsequent surveys.

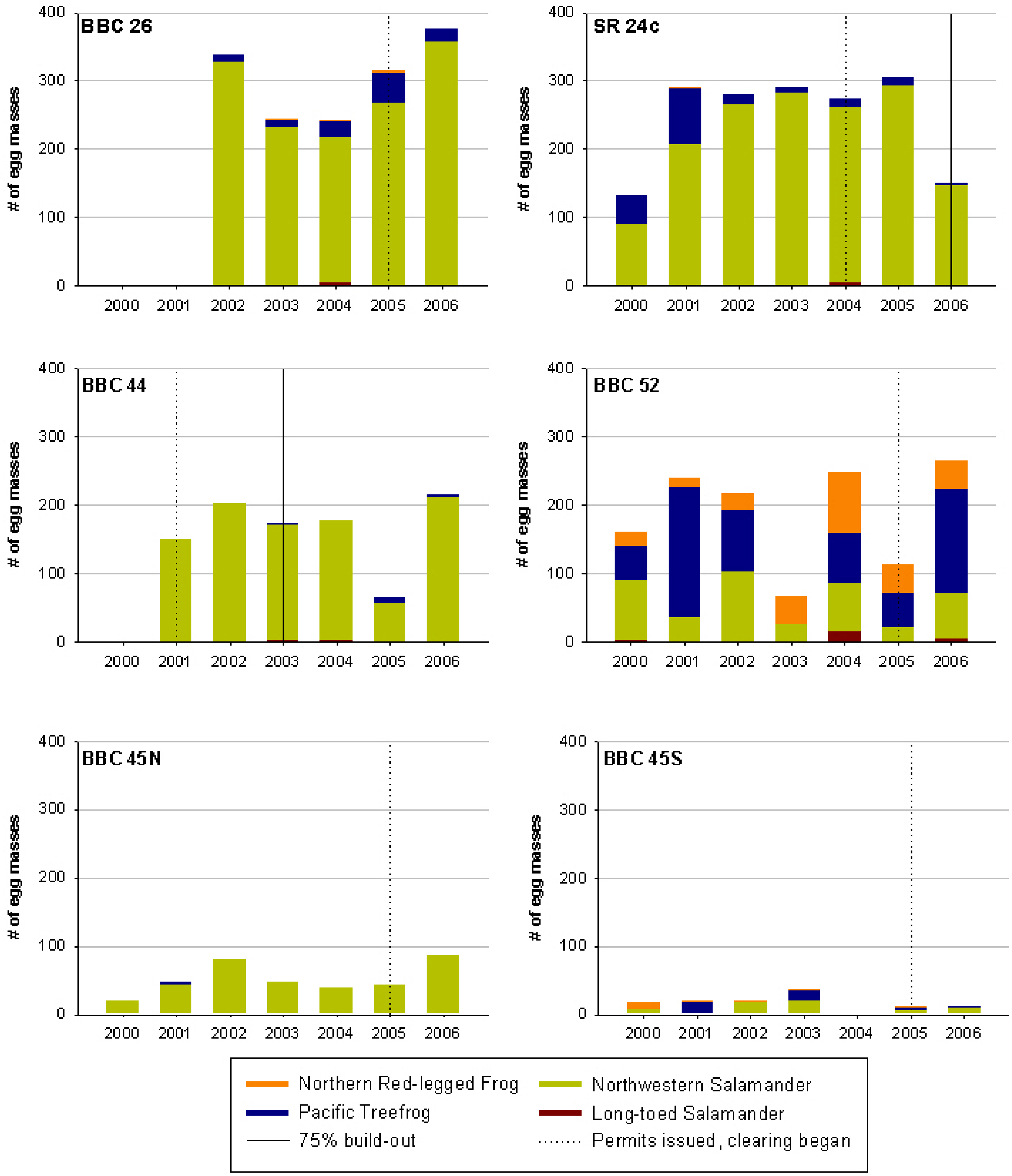

Figure 4.1.2.3 Amphibian egg mass abundance.
To ensure that masses were not double counted, the highest number of eggs from the two annual surveys is reported here. Generally, the Northern red-legged frog, long toed salamander, and Northwest salamander numbers came from the first survey and the Pacific treefrog numbers from the second survey. 75% build-out is estimated to occur two years after permits are issued and clearing begins.

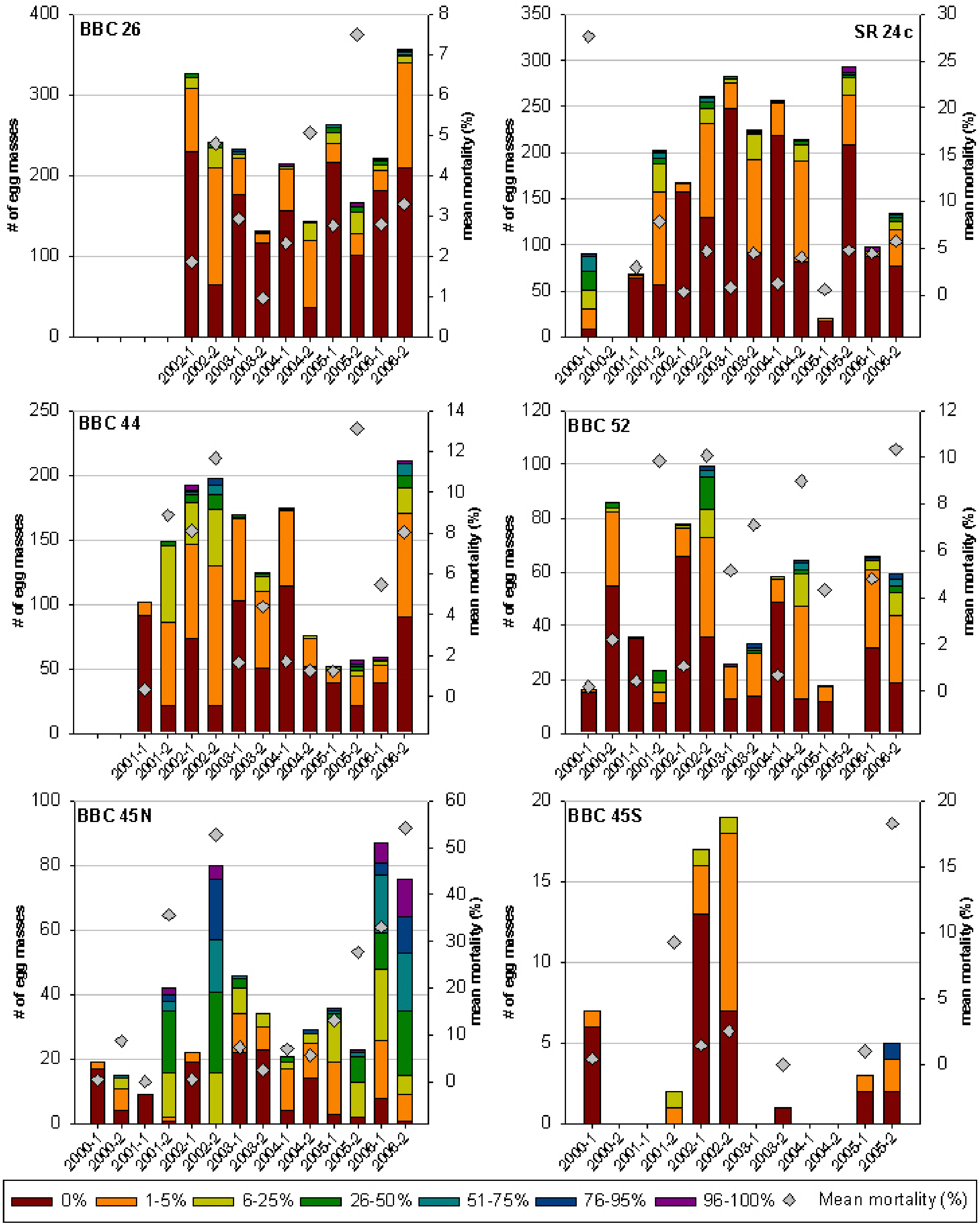

Figure 4.1.2.4 Northwestern salamander egg mass mortality and abundance.
Each year is divided into the two survey periods (e.g., 2000-1 and 2000-2). The data within the same year are not cumulative since some egg masses could be present during both surveys. Therefore, the abundance for a given year is assumed to be the maximum between the two surveys, which may be an underestimation. Partially hatched clutch data is not presented here. Mean mortality was calculated by averaging the midpoint of each category. The resulting number is only an approximation of mortality since these data were categorical.

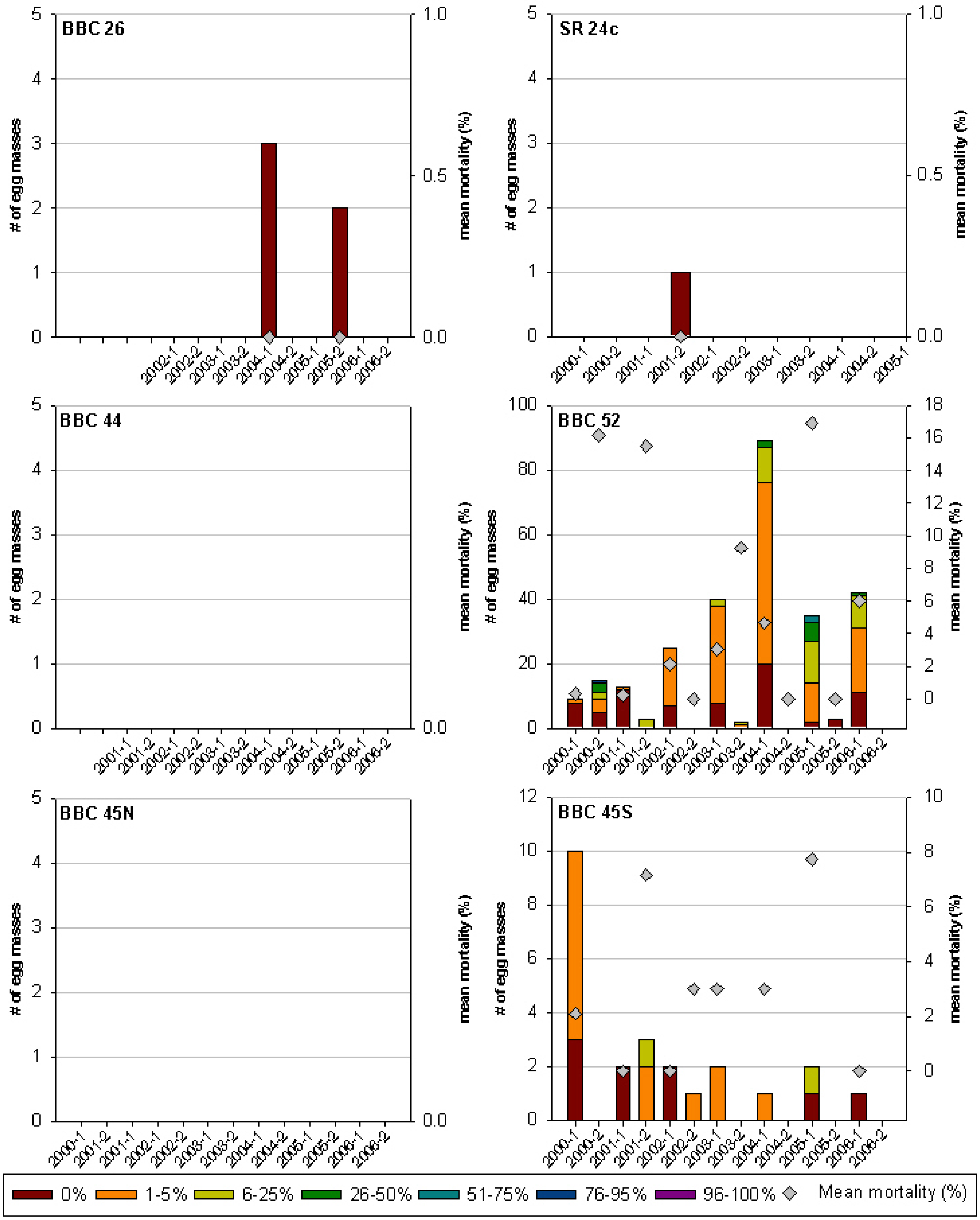

Figure 4.1.2.5 Northern red-legged frog egg mass mortality and abundance.
Each year is divided into the two survey periods, (e.g., 2000-1 and 2000-2). The data within the same year are not cumulative since some egg masses could be present during both surveys. Therefore, the abundance for a given year is assumed to be the maximum between the two surveys, which may be an underestimation. Partially hatched clutch data is not presented here. Mean mortality was calculated by averaging the midpoint of each category. The resulting number is only an approximation of mortality since these data were categorical.

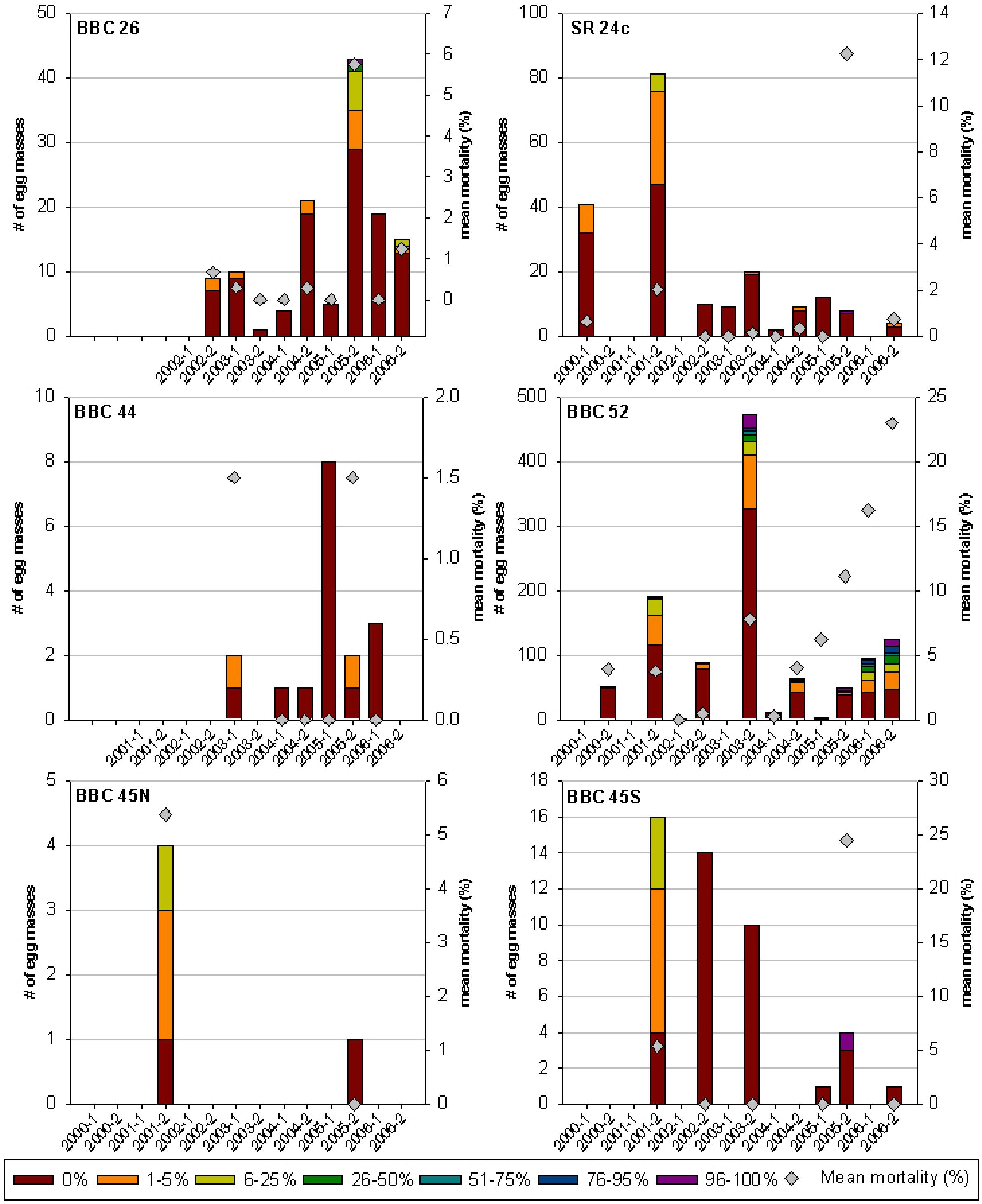

Figure 4.1.2.6 Pacific treefrog egg mass mortality and abundance.
Each year is divided into the two survey periods, (e.g., 2000-1 and 2000-2). The data within the same year are not cumulative since some egg masses could be present during both surveys. Therefore, the abundance for a given year is assumed to be the maximum between the two surveys, which may be an underestimation. Partially hatched clutch data is not presented here. Mean mortality was calculated by averaging the midpoint of each category. The resulting number is only an approximation of mortality since these data were categorical.

Table 4.1.2.1 Egg mass abundance for four species of wetland-breeding amphibians.
Egg mass abundance for the four most commonly observed species of amphibians from 2000 to 2006 (0 = 2000, 1 = 2001, etc.). Two surveys were conducted each year and these numbers represent the maximum number of eggs observed between the two surveys.

	Long-toed Salam.							**Pacific Treefrog**							**N. Red-legged Frog**							**Northwestern Salamander**						
	0	**1**	**2**	**3**	**4**	**2**	**6**	**0**	**1**	**2**	**3**	**4**	**2**	**6**	**0**	**1**	**2**	**3**	**4**	**2**	**6**	**0**	**1**	**2**	**3**	**4**	**2**	**6**
BBC 26			0	0	4	0	0			9	10	22	43	19			0	2	3	4	0			329	233	214	268	358
BBC 44		0	0	2	2	0	0		0	0	2	1	8	3		0	0	0	0	0	0		149	202	169	175	57	211
BBC 45N	0	0	0	0	0	0	0	0	4	0	0	0	1	0	0	0	0	0	0	0	0	19	42	80	46	39	42	87
BBC 45S	0	0	0	0	0	0	0	0	16	14	11	0	4	1	10	3	2	2	1	2	1	7	2	20	3	0	5	10
BBC 52	3	0	0	0	15	1	5	51	190	89	559	74	50	152	21	13	25	40	89	42	42	86	36	103	33	70	19	66
SR 24c	0	0	0	0	5	0	0	41	82	15	20	11	12	4	0	1	0	0	0	0	0	91	207	265	282	257	293	146

Table 4.1.2.2 Results of vertebrates captured during July 2006 funnel trapping.
Four traps were set for ~ 24 hours in each wetland. Invertebrates caught: >20 water boatmen & 1 water scorpion, 1 Dytiscid beetle, 3 dragonflies and 2 damselflies in BBC 26; 1 giant water beetle in BBC 52;

Wetland	Species	Adult	Larval	Paedomorph
BBC 26	Northwestern Salamander		4	4
	Pacific Treefrog		1	
BBC 44	Three-Spined stickleback	1		
	Northwestern Salamander		1	1
BBC 45N	Northwestern Salamander			17
BBC 45S	Northwestern Salamander			8
BBC 52	American Bullfrog		1	
	Northwestern Salamander			1
	Pacific Treefrog		1	
SR 24c	Northwestern Salamander			5

4.1.3 Fish Survey Monitoring

Overview

Changes in chemical and physical properties associated with increased urbanization and impermeable surface area have direct effects on stream habitat and ultimately stream ecology (Booth and Jackson 1997). The impacts of anthropogenic change upon rivers are particularly evident in the growing numbers of imperiled river-associated species, including fish (Allan and Flecker 1993). According to the categorization of (Williams et al. 1989) regarding the endangerment of North American freshwater fishes, roughly one out of three species and subspecies is endangered, threatened, or deserving of special concern (Allan 1995).

(Wang et al. 2001) suggested that urban development minimizing connected impervious surfaces and preserving undeveloped buffer areas should have less impact on stream habitat and fish than conventional types of development. The 60 m (200 ft.) average buffering employed on all streams and wetlands on the UPDs models this approach. Prior to development, it was predicted that development of the UPDs would not significantly alter the ability of fish to live, grow, and reproduce due to the proposed mitigation techniques (The Watershed Company 1995).

Therefore, periodic fish sampling was conducted on Adair, Colin South, Evans East, and Unnamed Creeks on reaches downstream of the UPDs to assess pre-development status (1991 & 2000) and compare potential post-development impacts (2006) on fish community structure and abundance. Fish abundance declines of more than 20% violate the thresholds established in the Trilogy and Redmond Ridge monitoring plans and could trigger appropriate corrective actions (King County 1999, 2001).

Procedures

Fish sampling was conducted on Adair, Colin South, and Evans East Creeks once annually in 1991, 2000, and 2006 in addition to Unnamed Creek in 1991 and 2000[11] (Figure 2.0). Sampling took place in September or December 1991[12] and in June 2000 and 2006 using three- or four-pass electrofishing depletion methods. We used four pass electrofishing removal/depletion methods in 1991 and 2000 and three pass methods in 2006 to survey fish within 60-100 m long reaches[13] (Zippin 1956, 1958, Platts et al. 1983). At each site, we used a Smith-Root type VII electrofisher with dip nets to collect fish that were transferred to buckets when captured and held until each pass was completed. Between each pass, every fish was identified to species and fork length measured[14] prior to being released downstream of the sample reach. In 1991, fish were sedated with MS-222 (tricaine methanesulfonate) to facilitate handling. Sedatives were not used in subsequent surveys.

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[11] Unnamed was not sampled in 2006 because access was denied by the property owner.
[12] Evans East was sampled in December 1991 due to channel drying in September 1991.
[13] Sample reaches were ~ 60 m in length in 1991 and 92-100 m in length in 2000 and 2006.
[14] In 2006, fish length was estimated into four size classes: 0-50, 50-100, 100-150, and >150 mm.

Block nets were used to prevent migration in or out of the reach at the downstream ends in 2000 and 2006 and at the upstream ends in 2000. In 2006, the upstream stopping point was selected at a habitat break (e.g., a riffle or gradient break) based on its ability to maintain a closed population.

Population estimates and 95% confidence intervals were calculated from depletion sampling using the methods and formulas outlined in (Lockwood and Schneider 2000). However, such estimates are inherently variable and for comparison population estimates were also calculated using methods described in (Seber and Cren 1967) and (Zippin 1958).

Results & Discussion

The baseline results from the 1991 surveys are not comparable to the June 2000 and 2006 data because they were collected during a different time of year (i.e., September and December compared to June). However, the 1991 results can be used as a reference for the species present in each survey reach.

From 1991 to 2006, no fish species have disappeared from any of the streams sampled. In 1991, only cutthroat trout (*Oncorhynchus clarki*) were found at any of the sample locations, although no fish were found in Evans East (Table 4.1.3.1). In 2000, cutthroat trout were found in Evans East and coho salmon (*Oncorhynchus kisutch*) were identified in Adair for the first time. In 2006, coho were also found in Colin South, although these may have been planted in the creek as part of the Washington Department of Fish and Wildlife (WDFW) Salmon in the Classroom program by Wilder Elementary School students[15]. In 2006, several riffle sculpin (*Cottus gulosus*) were found for the first time in Adair and a single juvenile pumpkinseed sunfish (*Lepomis gibbosus*) was found in Evans East.

While it is promising that we have not observed the disappearance of cutthroat trout in any of the streams since the onset of development, it is important to note that high quality coldwater streams naturally have relatively few species. Whereas environmental degradation is frequently associated with decreases in species richness in warm waters, it is typically linked to an increase in species richness in coldwater streams (Lyons et al. 1996). However, in the case of the sampled reaches, the single juvenile pumpkinseed was the only fish that seemed out of place in a coldwater system and presumably it ventured upstream from the warm water habitat present in Peterson Pond. Sculpin and salmonids are commonly dominant in coldwater systems. However, living in the stream benthos, sculpin are frequently overlooked and are notoriously difficult to sample. Benthic fish swim in short bursts and sink when stunned, making them more difficult to capture with electrofishing methods (Cowx and Lamarque 1990, B.C. Ministry of Environment 1997). Therefore, it is not beyond the realm of possibility that sculpin have always been present in the sample reach of Adair, particularly if early field personnel concentrated fishing efforts more at the top or middle of the water column than the bottom relative to 2006 crews.

Population estimates for the 2000 and 2006 sample periods indicated that cutthroat trout abundance declined approximately 92%, 66%, and 61% within the sampled reaches of Adair,

[15] According to the Wilder Elementary web site (http://schools.lwsd.org/wilder/watershed.htm), the school began participating in the Salmon in the Classroom program in 1990 (Wilder Elementary School 2006).

Colin South, and Evans East, respectively (Figure 4.1.3.1). In contrast, the abundance estimate for coho in Adair increased 109% between 2000 and 2006. Fish populations fluctuate from year-to-year and without comprehensive baseline data establishing the range of natural pre-development variability, it is difficult to determine whether the population changes observed were due to natural fluctuations, increased urban development, or even differences in sampling efficiency that could be attributed to the discontinuity in sampling personnel throughout the monitoring period. Furthermore, there is extensive variability in population estimates based on the method used to calculate them (Figure 4.1.3.2). For example, the estimates of population abundance for Adair cutthroat trout in 2000 ranged from 275 to 404.

The ratio of juvenile coho to cutthroat abundance has been proposed as an index of the condition of salmonid habitat based on past studies suggesting that cutthroat trout may replace coho as urbanization increases within a watershed (Scott et al. 1986, May et al. 1997). However, none of the sample locations had coho in 1991 and only one had coho in 2000. Therefore, we do not have sufficient pre-development coho to cutthroat ratios to use fish community composition as an indicator of degrading conditions.

In addition to the fish captured, northwestern (*Ambystoma gracile*) and Pacific giant salamanders (*Dicamptodon tenebrosus*) were found in Adair, Colin South, and Evans East during 2000 and 2006 stream sampling. The electrofishing surveys were not designed to estimate amphibian populations, therefore only presence or absence was noted.

Threshold Exceedance Criteria

Fish threshold exceedance criteria are violated if fish abundance estimates declined by greater than 20% compared to baseline and other monitoring data through time (King County 1999, 2001). Fish abundance declined by over 60% for cutthroat trout in Adair, Colin South, and Evans East between 2000 and 2006.

Corrective Actions

Changes in scour rates or decreases in water quality associated with the build-out of the UPDs were predicted to not significantly alter the ability of fish to live, grow, and reproduce in the streams originating onsite (The Watershed Company 1995). In fact, The Watershed Company concluded that if the period of minimal flow or drying was reduced as anticipated, stream flow conditions could be advantageous to cutthroat trout or other fish species present. However, reductions in abundance of cutthroat trout have been observed in Adair, Colin South, and Evans East.

As mentioned previously, fish populations fluctuate naturally and personnel changes between years likely reduced sampling consistency. Additionally, population estimates are inherently variable relying on numerous assumptions[16] that may or may not hold true and reflect just a snap shot in time of the specific reach sampled. Furthermore, no development from the UPDs has

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[16] Assumptions include: 1) emigration and immigration by fish during the sampling period are negligible; 2) all fish within a specified sample group must be equally vulnerable to capture during a pass; 3) vulnerability to capture of fish in a specified sample group must remain constant for each pass; and 4) collection effort and conditions which affect collection efficiency mush remain constant (Lockwood and Schneider 2000).

taken place within the Evans East basin to date. Therefore, the 61% decline in cutthroat trout in this creek cannot be attributed to UPD clearing, construction, and build-out and may be part of the natural variability of populations or other confounding factors (e.g., climate, ocean conditions, predation, development elsewhere in the basin, downstream conditions affecting migration, etc.).

Clearing and development of the Adair basin has largely taken place since 2004, and the basin is anticipated to hit 75% build-out in fall 2006 (Lowe 2005). The Colin South basin has had intensive development in some areas, and little development in others and the entire basin is probably close to the 75% build-out threshold that defines "post-development" as of fall 2006. Therefore, the summer 2006 data may reflect construction monitoring, rather than post-development monitoring as was intended. In addition, the three electrofishing passes in Adair in 2006 were not depleting the numbers of cutthroat trout, making this population estimate highly variable.

Due to the aforementioned reasons, no corrective actions are recommended at this time. Nevertheless, the declines in Colin South, Evans East, and Adair cutthroat trout populations should be monitored closely for continued changes over time. If continued declines are linked to UPD development, appropriate corrective actions should be implemented in the future.

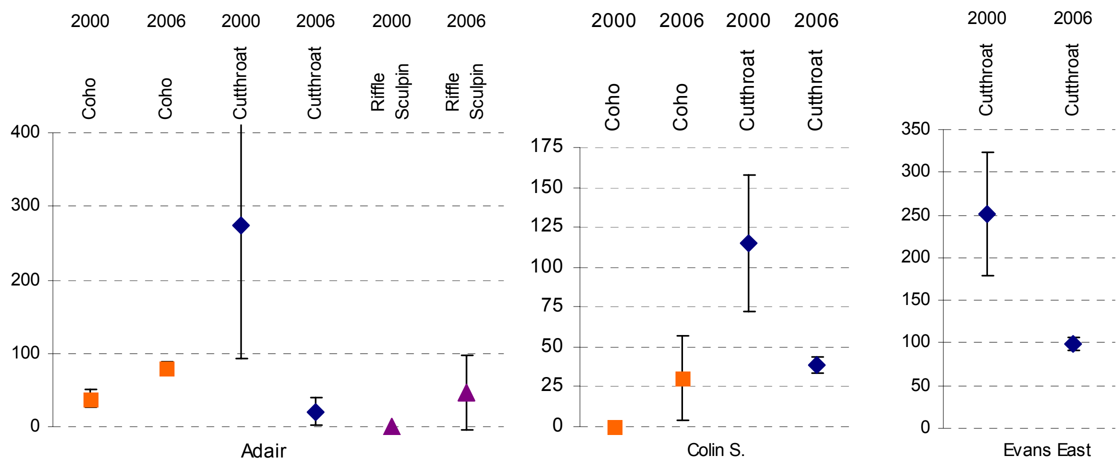


Figure 4.1.3.1 Abundance estimates within each sample reach

Abundance estimates within each sample reach were calculated from three electrofishing passes using the methods outlined in (Lockwood and Schneider 2000). Error bars represent 95% confidence intervals. One pumpkinseed sunfish was observed in Evans East in 2006, but data were not sufficient to make population estimates. Unnamed Creek was not sampled in 2006 due to property access denial.

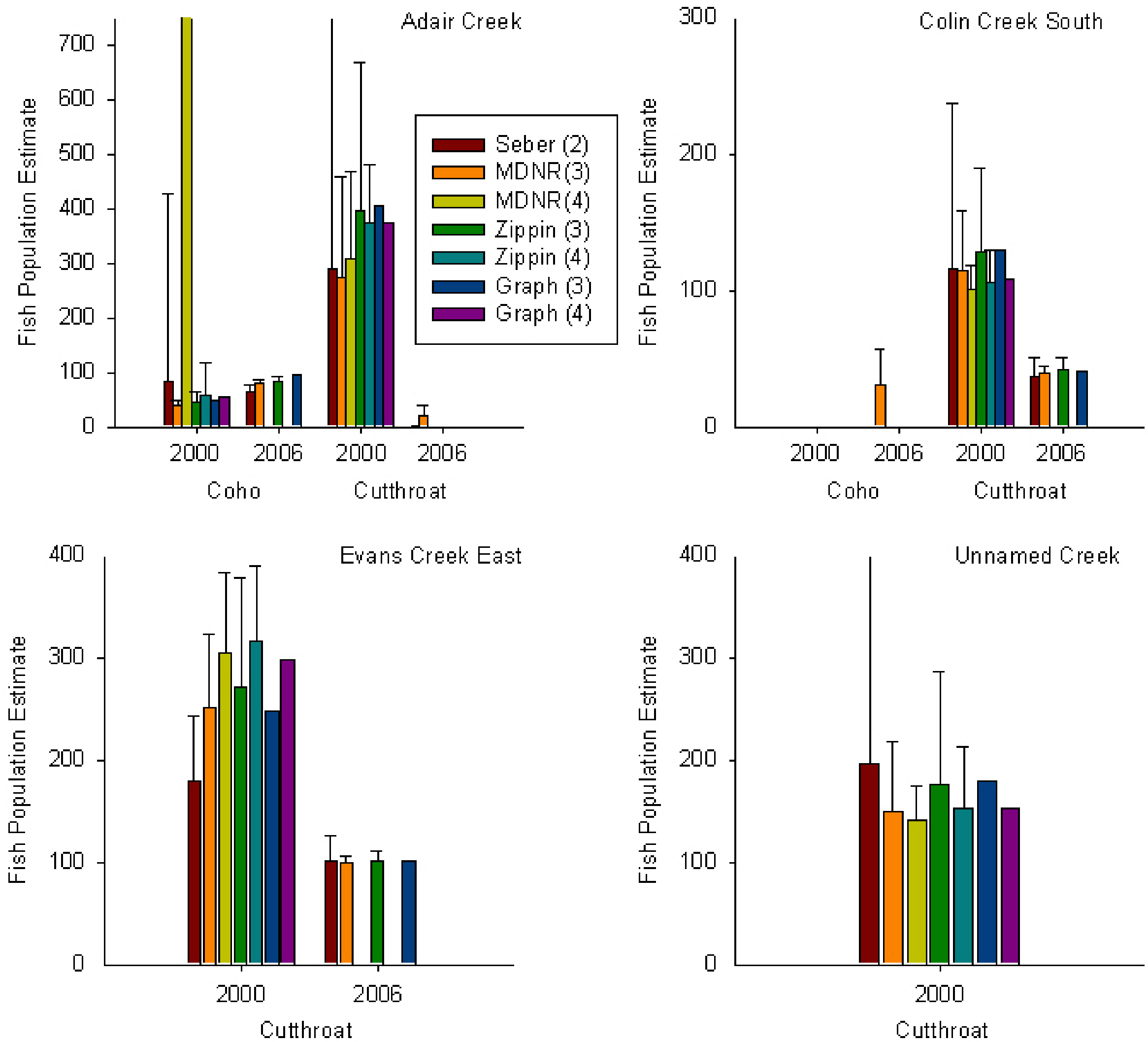


Figure 4.1.3.2 Comparison of fish population estimates calculated by different methods.
Error bars represent the upper 95% confidence interval. The number in parentheses in the legend represents the number of passes the estimate is calculated from. In 2006, only three passes were conducted so no four-pass calculations could be made. If the fish population was not being depleted, no estimate could be made using most methods (e.g., Colin South Coho in 2006) or highly skewed estimates resulted (e.g., Adair Creek MDNR (4) in 2000). Estimation formulas are described in (Seber and Cren 1967), (Lockwood and Schneider 2000), and (Zippin 1958).

Table 4.1.3.1 Species present during three electrofishing sampling dates on four creeks draining the UPDs. 1991 data were collected in December for Evans East, and in August for the other three creeks. 2000 and 2006 sampling was conducted in June.

Stream	1991	2000	2006
Adair	Cutthroat	Cutthroat, Coho	Cutthroat, Coho, Riffle Sculpin
Colin South	Cutthroat	Cutthroat	Cutthroat, Coho
Evans East	No fish	Cutthroat	Cutthroat, Pumpkinseed
Unnamed	Cutthroat	Cutthroat	Not Sampled

4.1.4 Vegetation Monitoring

Introduction and Methods

Vegetation monitoring surveys were performed on wetland BBC-52 during 2000, 2002, 2004, and 2006 following (Elzinga 1998). The surveys were performed along six transects across the wetland that ranged in length from 105 m to 240 m and varied in the number of quadrats from five to eight. In addition, the sampling was set up in three strata (forested, shrub, and herb). These three strata were sampled in 10 m radius quadrats, 5 m radius quadrats, and 1 m radius quadrats, respectively. Data collected during the surveys included identification to the most specific taxonomic level possible, and categorical assignments of percent cover. The ends of each transect were marked with flagged rebar and each quadrat was marked with a piece of PVC pipe that marked its center. This ensured that sampling occurred at the same location in each year's survey. All strata occurred on each transect. In addition, continuous water level (stage) monitoring occurred during the study period. Wetland vegetation diversity metrics were compared to water level to test whether water level influences the vegetation in the wetland system. Correlation values are presented where a relation was observed (Appendix 4.1.4.1).

Six categories of vegetation cover classes were used to define the percent cover of specimens that were enumerated during the sampling. Coverage classes indicated 0 (trace), 1 (0.5% to 5%), 2 (6% to 25%), 3 (26% to 50%), 4 (51% to 95%), 5 (76% to 95%), and 6 (96% to 100%). Samples were identified to species when possible. However, some genera were lumped into general categories (e.g., mosses and lichens).

Given that no real development, with the exception of two athletic fields, has occurred in the basin in proximity to wetland BBC-52 and that clearing did not begin until 2005 for most of the drainage, the first three sampling events were used to define the "pre-development" conditions with respect to the data analysis. In this way, mean conditions with variability bounds were established that could account for natural variability among years, and sampling effectiveness among staff including differences in observer bias and technical capability. Analyses were limited to comparisons between 2006 observations and the mean conditions for the first 3 sampling epochs.

Results by Transect

In all, 164 unique species were identified in the vegetation surveys (Appendix 4.1.4.1). The number of species identified within each plot on each transect varied year to year. However, mean values among all transects ranged between approximately 15 and 20 species enumerated (Figure 4.1.4.1). The lowest diversity occurred at transect 6, plot E with a mean diversity of 3.5 species. There were four locations that had more than 30 species noted (Transect 1, Plot B; Transect 2, Plot B; Transect 3, Plot B; and, Transect 4, Plot B). All of these locations on Transects 1, 2, and 4 were forest plots, and Transect 3 was a shrub plot.

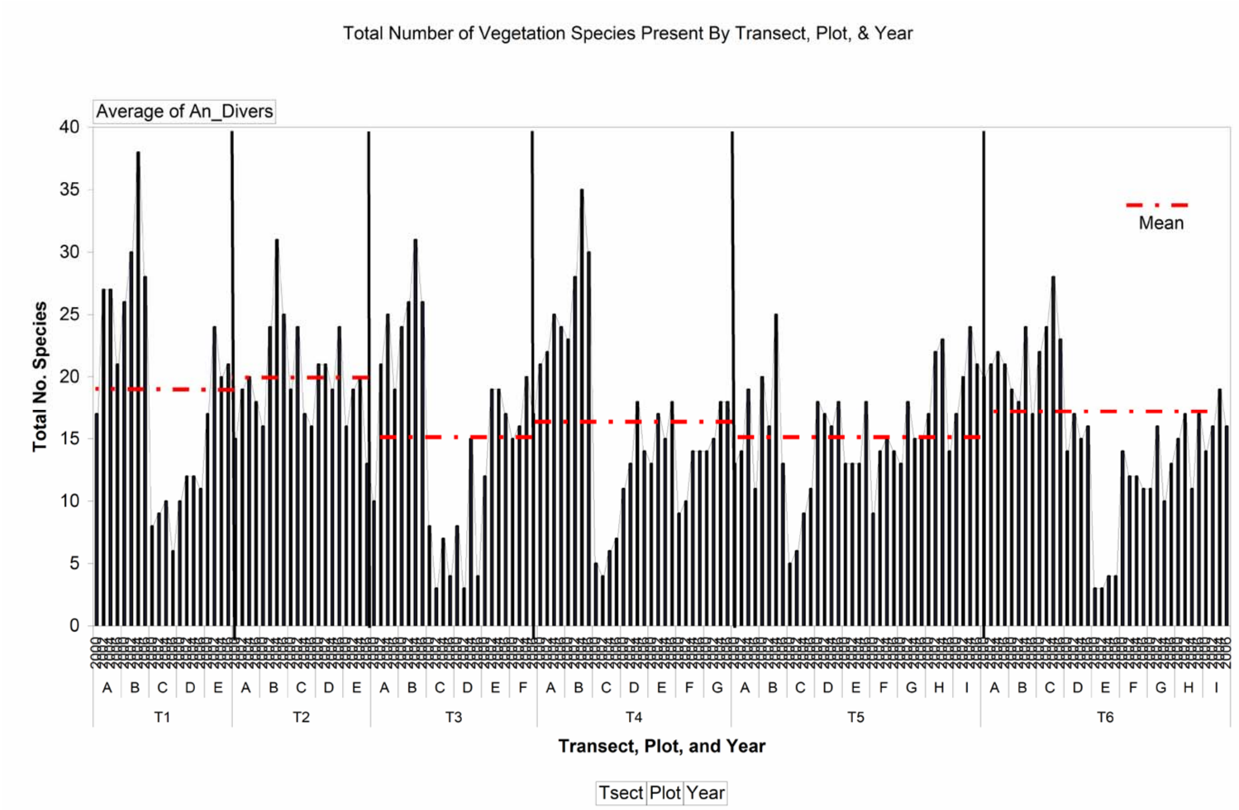


Figure 4.1.4.1 Total number of plant species observed at each transect, and plot during all years (2000 to 2006).
Bold vertical lines indicate breaks between transects. Dashed red lines indicate mean number of species for each transect among all years

Transect 1:

This transect had five plots (A through E, respectively, Figure Vegetation 2, Panel 1). Plot A was a forested site with a wide variance in terms of the number of individual plants measured across years (range 289 to 729). The wide range of values could be attributed to different competencies among field technicians, slight differences in the locations measured, or a combination. Plot B was also a forested site with the fewest number of individual plants recorded during the 2000 field visit (676 individuals), and the most recorded during 2004 (1444 individuals). Plot C was categorized as an herb/ emergent plot and had a low value of 36 individual plants during 2006, and a high value of 100 individuals during 2004. Plot C also showed a negative correlation with the wetland water stage (R = -0.69). Plot D was a shrub plot with between 100 and 144 individuals enumerated at this location. Plot E was forested with between 289 and 576 individual plants at this location. This plot showed a positive correlation with wetland stage (R = 0.32, Appendix 4.1.4.1). Again, because the forested plots were 10 m in diameter, it is expected that there would be many more individuals at these locations than in the 1 m herb plots.

Transect 2:
This transect also had 5 plots (Figure 4.1.4.1, Panel 2). Plot A was forested with a range of individual plants numbers from 225 during 2000 to 400 during 2004. Plot B was a shrub plot (quadrat size 5 m in diameter), with 256 individuals during 2000 and 961 individuals during 2006. Plot C was classified as a shrub transect part of the time and an herb transect the remainder of the time. The transitional nature of this plot can explain the large variation in the number of individuals because the shrub plots are 25 times the area as the herb plots (i.e., 5 m in diameter as opposed to 1 m diameter). Plot C showed a negative correlation to water stage in this wetland (R = -0.40, Appendix 4.1.4.1). Plot D was a shrub transect with a low of 361 individual plants enumerated during 2004 and 576 individual plants during 2006. In addition, reed canary grass and evergreen blackberry were noted at this site and are both considered to be noxious weeds in King County. Plot D also showed a positive relation to water stage (R = 0.83, Appendix 4.1.4.1). Plot E was classified as a shrub quadrat and had a low of 169 individual plants noted during 2006 and a high of 400 during 2004. This plot also showed a negative relation with water stage (R = -0.64, Appendix 4.1.4.1).

Transect 3:
This transect consisted of 6 plots (A through F) ranging from forest to shrub to herb (Figure 4.1.4.1, Panel 3). Plot A was a forested site with between 100 and 625 individual plants enumerated (2000 and 2004 respectively). Plot B was also a forested plot with individuals enumerated ranging from 576 during 2002 to 961 during 2004. In addition, reed canary grass was found at this location as well. Plot C was an herb site with between 9 individuals during 2002 to 64 during 2000. Plant diversity at this location exhibited a negative correlation with water stage (R = -0.65, Appendix 4.1.4.1). Plot D was also an herb site with plant numbers ranging from a low of 9 during 2002 to a high of 225 during 2004. This location was also negatively correlated with water stage (R = -0.38, Appendix 4.1.4.1). Plot E was a shrub site with diversity ranging from 144 to 361 (2000 and 2004 respectively). Reed canary grass was observed at this site as well. Plot F was a forest quadrat with diversity numbers ranging from 225 to 400 (2000 and 2004 respectively). In addition, English ivy and Holly were noted at this site. Both plants are considered noxious by King County.

Transect 4:
Transect 4 consisted of 7 plots (A through G, Figure 4.1.4.1, Panel 4). The strata observed on this transect consisted of forest, shrub, and herb. Plot A was a forested site with individual plant numbers ranging from 441 during 2000 to 625 during 2004. The noxious weed Holly was observed at this location. In addition, the total number of individual plants enumerated at this site was positively correlated with water stage (R = 0.48, Appendix 4.1.4.1). Plot B was also a forested site with plant numbers ranging from 529 to 1225 (2000 and 2004 respectively). Holly, Reed Canary Grass, and Evergreen Blackberry (noxious weeds in King County) were also observed at this location. In addition, European Mountain Ash which is a non-native but not considered weedy, was observed at this location. Plot C was an herb plot with between 16 and 49 individuals enumerated at this location (2002 and 2006 respectively). This plot showed a positive correlation with wetland water stage (R = 0.77, Appendix 4.1.4.1). This is the first herb plot that has exhibited an increase in diversity relative to an increase in water stage. Plot D was classified as shrub most of the time but was also classified as an herb plot during approximately

1/3 of the surveys. The number of individual plants enumerated at this location ranged from 121 during 2000 to 324 during 2004. In addition, the noxious weed Reed Canary Grass was noted at this location. Plot E was a shrub plot that showed a positive correlation with wetland water stage (R = 0.86, Appendix 4.1.4.1). The plant diversity at this location ranged from 169 during 2000 to 324 during 2006. Plot F was also a shrub plot with diversity ranging from 81 to 196 during 2000 and 2004 respectively. Vegetation numbers in this plot was also positively correlated with wetland water stage (R = 0.65, Appendix 4.1.4.1). Plot G was a forested plot with plant numbers ranging from 196 during 2000 to 324 during 2004 and 2006 and was positively correlated with wetland water stage (R = 0.67, Appendix 4.1.4.1). In addition, the noxious weed Holly was observed at this location.

General findings indicate that at most locations there have not been alarming changes in the vegetation communities in this wetland. However, when samples for 2006 are compared against the mean pre-development conditions, there is a general decrease in total diversity. Still, in most cases the change is within the expected variability of the mean conditions. In cases where the magnitude of difference is outside the normal range of variability, the absolute change is usually very small. Nonetheless, a decrease in species diversity can signal impacts that affect the most sensitive species. With that said, one year of comparison is insufficient to establish a trend.

Transect 5:

Transect 5 was 225 m long and consisted of 9 vegetation sampling plots 4 of which were forested, 4 were shrub, and 1 was an herb plot (Figure 4.1.4.1, Panel 5). Plot A was a forested plot with plant numbers ranging from 121 during 2006 to 361 during 2004. This site exhibited a negative correlation with wetland water stage (R = -0.46, Appendix 4.1.4.1). Plot B was also forest with plant numbers ranging between 169 and 400 (2006 and 2000 respectively). This site showed a relatively strong negative correlation with wetland water stage (R = -0.63, Appendix 4.1.4.1). Plot C was an herb quadrat with total plant numbers ranging from 25 to 121 (2000 and 2006 respectively). This site showed a strong positive correlation with wetland water stage (R = 0.88, Appendix 4.1.4.1). In addition, the noxious weed Reed Canary Grass was observed at this location. Plot D was a shrub site with total plant numbers ranging between 256 during 2004 to 324 during 2000 and 2006. This site also showed a positive correlation with water stage, albeit a weaker one (R = 0.32, Appendix 4.1.4.1). Plot E was a shrub site with total plant numbers ranging from 169 during 2000, 2002, and 2004 to 324 during 2006. This site had a very strong positive correlation with wetland water stage (R = 0.96, Appendix 4.1.4.1). Plot F was also a shrub site with total plant numbers ranging from 81 during 2000 to 225 during the 2004 survey. Total plant numbers were positively correlated with water stage at this location as well (R = 0.47). Plot G was also a shrub quadrat. Total plant numbers at this location ranged from 169 during 2000 to 324 during 2002. Plot H was a forested site with total plant numbers 196 to 529 during 2006 and 2004 respectively. Plant numbers exhibited a negative correlation with wetland water stage at this location (R = -0.55, Appendix 4.1.4.1). Plot I was also a forested quadrat with plant numbers ranging from 289 during 2000 to 576 during 2004.

Transect 6:

Transect 6 was the longest transect with a total length of 240 m. Like the other transects in this vegetation study, Transect 6 was comprised of forested, shrub, and herb quadrats (Figure 4.1.4.1, Panel 6). There were 9 survey plots located on this transect (A through I). Plot A was a forested quadrat with a relatively low diversity. Plant numbers observed at this location ranged from 400 during 2000 to 484 during 2004. Additionally, this plot exhibited a negative correlation between total numbers of botanical individuals and wetland water stage (R = -0.46, Appendix 4.1.4.1). Plot B was forested, had total plant numbers ranging from 289 during 2006 to 576 during 2004, and was strongly negatively correlated with wetland water stage (R = -0.64, Appendix 4.1.4.1). Plot C was a shrub quadrat with plant numbers ranging from 484 to 784 during 2000 and 2004 respectively. In addition, the noxious weed Holly, and the non-native European Mountain Ash were observed at this location. Plot D was also a shrub site with plant numbers ranging from 196 to 289 individuals. The low and high years were 2000 and 2002 respectively. Plot D showed a positive correlation between total numbers of plants and wetland water stage (R = 0.46, Appendix 4.1.4.1). Plot E was an herb quadrat with plant numbers ranging from 9 during the 2000 and 2002 surveys to 16 during the 2004 and 2006 surveys. This location also exhibited a positive correlation between total plant numbers and wetland water stage (R = 0.63, Appendix 4.1.4.1). Plot F was also an herb quadrat with total plant numbers ranging from a low of 121 during 2006 to a high of 196 during 2000. This location exhibited a strong negative correlation with wetland water stage (R = -0.82, Appendix 4.1.4.1). Plot G was classified as herb approximately 20% of the time and shrub approximately 80% of the time. Total plant numbers at this location ranged from 121 during the 2000 survey to 256 during the 2002 survey. Plot H was classified as shrub slightly less than 1/3 of the time and forest the remainder of the surveys. This location exhibited a range of total plant numbers that ranged from 121 during 2004 to 289 during 2002 and 2006. There was also a positive correlation between total plant numbers and wetland water stage at this location (R = 0.48, Appendix 4.1.4.1). Plot I was a forested quadrat with total plat numbers ranging from 196 during 2000 to 361 during the 2004 survey.

Conclusions and Recommendations

The vegetation surveys on wetland BBC 52 were intended to monitor if the UPD development has had an adverse impact on the botanical community of this bog wetland system. The study design for this component of the overall natural resources monitoring program seems to be adequate to detect changes at this scale. However, some of the specific questions about wetland botanical community integrity relative to development progress will require more time in order to determine quantitatively if there has been an adverse impact. In addition, some of the questions that have arisen as a result of our analyses may be explained by sampling error. For example, it was not expected that there should be radical changes in total plant numbers in forested or shrub quadrats among years and yet, there were large changes observed during some years. Additionally, systematic changes in community composition would likely show consistent trends. The only consistency in the error among plots and transects is that total numbers of individual plants is generally higher during 2004 than during other years. This suggests a systematic increase in sampling effort or efficiency during 2004. Moreover, the positive correlation between plant numbers and water levels in these strata was a surprise. A reasonable hypothesis would be that plant numbers and diversity should decrease in herb quadrats if there were adverse hydrological changes in the wetland. Also, the presence of invasive species and changes in local dominance could show a demonstrable effect in herb quadrats during the time period studied but would be unlikely to cause community shifts in shrub, and less likely to cause shifts in forest communities. The quadrats where non-native, noxious, and otherwise undesirable plants have been observed should continue to be monitored into the future to ensure that these plants do not affect the ecology of the system. In addition, efforts to eradicate weeds throughout the County should incorporate these locations into their programs.

To varying degrees, more analyses can be performed with the existing dataset in a comparison of 2006 and subsequent years to pre-development conditions. More data will be needed in order to make a quantitative determination of the nature and extent of chronic impacts to vegetation communities relative to the UPD. Given that this basin is still largely undeveloped and that the rate of development is relatively slow for this particular location within the UPDs, the prescribed monitoring timeline (sampling every other year, 4 times following 75% buildout) may provide enough information to establish a trend, assuming a new ecological trajectory is established due to some chronic impact to the system. However, at that time it may be necessary to re-evaluate the monitoring plan, which could include extending the monitoring period, in order to make that determination quantitatively.

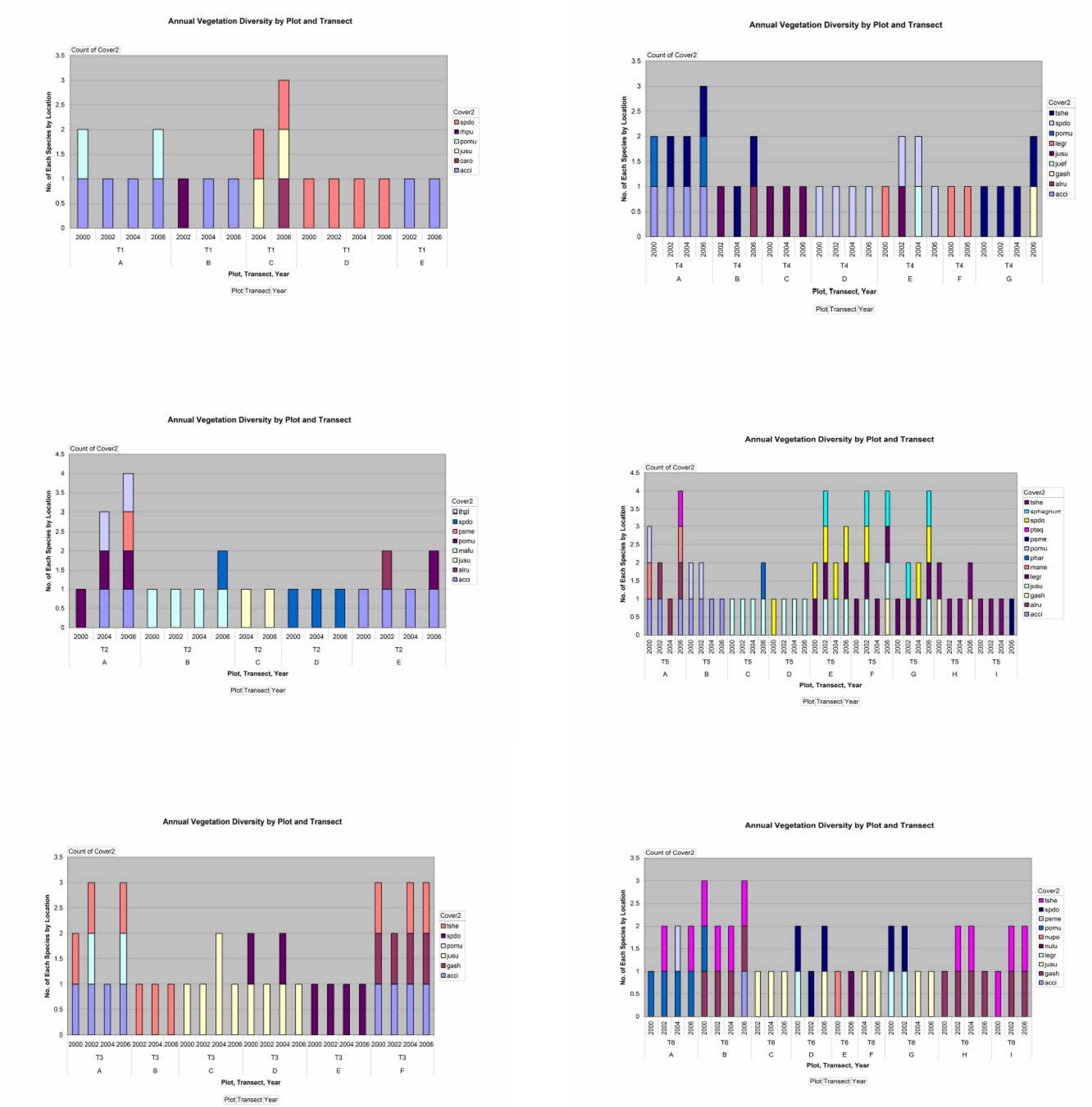


Figure 4.1.4.1 Wetland BBC 52 annual vegetation diversity by transect, plot, and year for transects 1 through 6, plots A through I, and years 2000 through 2006.

Upper left-most panel represents Transect 1, and lower right-most panel represents Transect 6. Panels move left to right and top to bottom representing Transects 1 through 6 respectively. For all transects, Plots begin with A on the left, and move right through the farthest-most plot. The legend on the right of each panel indicates the species of plants that were observed during the surveys for respective locations. Count values on the Y axes, indicate the number of times each species was observed.

4.2 Physical Assessment

There is a rich literature documenting the effects that changes in land cover have on watershed hydrology, and stream condition sensu (Horton 1939, Wolman 1967a, Karr 1991, Allan 2004). This section focuses on the hydrologic and geomorphic effects of the watershed changes related to the UPDs. We extensively monitored stream discharge, wetland water stages, and stream channel cross-sections as responses to changes in watershed hydrology. In addition, early investigations documented conditions prior to the development for qualitative stream habitat surveys, riparian canopy cover, stream bank stability, and other stream characteristics (Comings et al. 2000). All of the creeks affected by this development are small headwater tributaries that flow into larger salmonid-bearing streams (Bear Creek, Evans Creek, and the Snoqualmie River). In addition, Adair Creek, Colin Creek South, Evans Creek East, and Unnamed Creek all supported salmonid populations during the study period (See

section). Nine stream and wetland systems drain the two UPD sites (Figure 2.0).

- Adair Creek
- Colin Creek North Fork
- Colin Creek East Fork
- Colin Creek South Fork
- Evans Creek East Fork
- Evans Creek Middle Fork
- Mackey Creek
- Rutherford Creek
- Unnamed Creek

Adair Creek

Adair Creek begins at a beaver dam and drains the northeast corner of the Trilogy property. The surrounding riparian community is mixed conifer and deciduous forest with considerable undergrowth of Salmonberry (*Rubus spectabilis*) bushes along the entire length of the stream. The bed substrate is mostly small cobble with an overlying silt layer that is easily disturbed. At 25 m downstream from the beaver dam is the 1997 benthic invertebrate sampling site. Approximately 190 m downstream is the King County flow gauge (53A) and about 25 m beyond that are the monitoring cross-sections (Figure 2.0). The stream gradient increases as the stream flows eastward toward the Snoqualmie River Valley with a concomitant substrate coarsening due to increased stream power (Julien 2002). The gradient remains high until it contacts the Snoqualmie Valley floor. The stream is influenced by large volumes of large woody debris (LWD) that exerts some control on sediment movement within the channel. There is some evidence of recent bank erosion in small patches but overall the channel appears to be relatively stable. As the stream contacts the Snoqualmie Valley floor, gradient decreases and the sediment predictably becomes finer. Just upstream of the West Snoqualmie Valley Road (about 790 m downstream of its origin) blackberry (*Rubus discolor*) vines along the bank become increasingly prevalent.

Colin Creek North

The headwaters of Colin North originates northeast of the Trilogy property near the Tolt pipeline easement and flows southward in a low-gradient streambed that is largely silt with a few gravel riffles (Figure 2.0). A layer of detritus covers the streambed and is easily disturbed. The watershed is generally low gradient and vegetated with a forest comprised primarily of young Red Alder (*Alnus rubra*). At approximately 200 meters from its origin, the stream enters the north side of wetland BBC 26 and emerges a short distance downstream at the western end of the wetland. The streambed consists of small gravel with a substantial amount of fine material, both organic and mineral. Within this section there are four cross sections established, and very little apparent erosion along this reach. Approximately 125 meters downstream of the wetland is the King County flow gauge 02C. The stream enters Welcome Lake approximately 630 m from its origin.

Colin Creek East

Colin Creek East drains a small section on the western edge of Trilogy and begins in a very muddy, wetland south of NE 133rd Street just inside the UPD property boundary (Figure 2.0). It crosses NE 133rd Street in a small culvert and flows northward into the "Lake of the Woods" neighborhood. The creek broadens in some places to become small wetland areas but is mostly very stream-like. Though narrow, the riparian vegetation remains intact for the entire length of the stream. There is abundant wood in the stream, and the substrate consists of small cobble and gravel overlain with considerable organic material in some locations. This tributary joins the main stem of Colin Creek South just prior to the culvert under NE 137th Street, approximately 360 m from its origin.

Colin Creek South

This stream originates in wetland BBC 44 approximately 300 meters downstream of the western edge of the Trilogy UPD (Figure 2.0). The creek flows north through a sandy/silty channel with small cobbles and several small LWD-forced pools. Approximately 500 meters downstream from the origin is King County flow gauge (02D) and the cross-section measurement stations. The stream channel gradually widens and deposits sediment sand and gravel forming bars with a few large cobbles. Both banks exhibit visible erosion in places despite being well vegetated (primarily with Salmonberry, *Rubus spectabilis*; Vine Maples, *Acer circinatum*; Devils Club, *Oplopanax horridus*; and Sword Fern, *Polystichum munitum*). The gradient is approximately 4 percent. Approximately 1000 meters from the wetland boundary, the stream crosses 133rd Avenue NE and subsequently merges with Colin Creek East (08-0134). Approximately 1600 meters from the headwater wetlands near the confluence with Welcome Lake there are a series of beaver dams that impound the system.

Evans Creek

The middle fork of Evans Creek originates near the southern property boundary of Redmond Ridge where King County flow gauging weir, 18B, serves as the outlet of wetland EC-4 (Figure 2.0). Downstream of the UPD boundary, the stream flows through a development known as Harrington North, where it passes under several small roads. Coniferous riparian communities on the stream vary from 15 meters to greater than 30 meters in this area. The stream is low gradient and is reflected in the substantial silt deposition. There are few if any riffles; in places the stream could actually be considered a wetland. The riffles that were encountered were usually associated with hydraulic jumps at culverts where higher energy scoured away the fine sediment. Because of the intact riparian areas, there is still significant wood influencing the direction and flow of the stream, creating a few large pools.

Mackey Creek

The headwaters of Mackey Creek come from subsurface flow draining from the northwest corner of the Redmond Ridge property (Figure 2.0). The reach of channel under investigation has not had any apparent surface flow during the monitoring period. The stream channel runs southeasterly about 30 meters before coming to a culvert under NE 104th Place. On the downstream side of this culvert a discharge point from the roadways of the current development. This runoff infiltrates into the ground within 10 to 20 meters so that there is only a short reach of surface flow before the channel becomes dry again. In approximately 50 meters of dry channel there is a culvert under Novelty Hill Road. Downstream of the Novelty Hill culvert only slight evidence of surface water flow is apparent until the ground becomes very muddy and wetland-like. Even without surface water, the thalweg can be followed because the small valley has a very distinct V-shape to it. The stream enters wetland BBC-46 about 100 meters below the culvert and 200 meters beyond the western boundary of the UPD property.

Rutherford Creek

Rutherford Creek originates in a wetland system that begins on the Redmond Ridge property and continues south beyond the property boundary about 300 to 400 meters (Figure 2.0). The stream thus begins downstream of the UPD boundary. The streambed is comprised of fine, silty mud that is loosely packed to a depth that exceeds 0.5 m in some places. The banks are primarily vegetated with Red Alder and Salmonberry. Approximately 350 meters downstream is the King County flow gauge 18F. Downstream of the gauge the steam flows about 10 to 20 more meters before entering a pond in a horse pasture with no riparian vegetation and substantial bank erosion due to the horses. Downstream of the pond, the stream continues for approximately 150 meters through a healthy and wide coniferous riparian zone. Beyond this the stream flows onto a property with another large pond, the outlet of which flows into the culvert under NE Union Hill Road. Immediately downstream of the Union Hill culvert are the Rutherford Creek monitoring cross sections.

Unnamed Creek

Unnamed Creek begins at a culvert under Novelty Hill Road just beyond the southeastern edge of the Trilogy UPD (Figure 2.0). For the first 100 meters or so the stream is low gradient with a muddy bottom and sluggish flow. But substrate size gradually increases in the downstream direction. Approximately 200 meters downstream of the origin is the King County flow gauge, 53B. The valley containing the creek is quite narrow with steep sidewalls, incising in the downstream direction. Locally, erosion has exposed the till along the hillslopes colluvially depositing clumps of hard, crumbly clay in the riffles. There are sections of the stream that have considerable bank erosion, particularly near the cross-section site, which is about 470 m from the origin of the stream. The riparian forest consists primarily of coniferous trees on both banks with Salmonberry and Devil’s Club growing close to the stream. As a result, there is a prodigious amount of wood in the stream. In some places the wood is so prevalent that the stream can not be seen beneath it. In other places, very large logs lie suspended overhead between the narrow valley walls. The culvert entrance at the 243rd AV NE crossing is rocked over. This causes the stream to impound at high flows. Gauge data and high water marks indicate impoundment depths of up to two meters. The confluence with the south fork of Unnamed Creek is approximately 700 meters downstream of the origin. The south fork is relatively large, increasing the flow substantially. Downstream of the confluence, the stream gradient decreases and the riparian community is increasingly dominated by deciduous trees.

4.2.1 Hydrology/ Analysis of UPD Flow Data

Hydrology is regarded as a dominant organizing factor controlling ecosystem conditions. Hydrologic processes are governed by complex interactions among a host of variables including precipitation, geologic template, and land cover (Ward 2004). Because hydrology is such a dominant force in ecosystems, and because changes in hydrology due to development can have such profound effects on ecosystem function, it is a key element in this monitoring program. The movement of water through streams is a dominant force controlling the physical condition of any stream (Julien 2002). Changes in the rates and timing of delivery of water to stream channels due to urbanization can cause changes to the physical channel conditions that are more extensive than would be predicted under unaltered conditions (Konrad 2005).

Baseline hydrologic conditions were established by measuring flow through installed hydraulic flumes on Adair, Colin North, Colin South, and Unnamed creeks. Thin crested weirs were used on Rutherford and Evans during 1989 through 1996 (Figure 2.0). In addition, crest – stage gauges were established to measure water level fluctuation in several wetland systems on the project. In 2001 the most important wetland monitoring locations had continuous water level recorders installed to better measure water level fluctuation. The gauging data were collected in fifteen-minute increments. Four of the stream gauges (Adair, Colin South, Rutherford, and Unnamed) were reactivated in 1998 prior to the start of construction and continue currently. Evans, Colin North and the South Fork of Unnamed were reactivated in 2000 and continue currently.

The stage data collected during 1998 to 2006 was translated to flow data by King County staff and put in a standard format for use with a model for hydrologic analysis and prediction. The formatted data were compared to the baseline conditions to look for changes in flow characteristics relative to changes in watershed hydrology (i.e., degree built, using permit phasing as a proxy for actual percent land cover change).

Hydrologic data collected in and around the UPDs include precipitation and flow data. Precipitation was collected at several sites around the UPDs – combining these sites provides a daily record of rainfall from 10/1/1988 through 8/7/2006 (Table 4.2.1.1, Figure 4.2.1.1). Individual rain gauges and their periods of record are: 02v, 10/1/1995 – 8/7/2006; 18u, 10/1/1988 – 9/30/2000; 18v, 10/1/1998 – 11/18/2004; and RRUPD, 7/23/2002 – 10/17/2005. Averages were used for periods of overlap in the record.

Table 4.2.1.1 Mean annual precipitation in inches for all rain gauges in the vicinity of the UPDs for 1989 to 2006.

Year	Total Precipitation (in)
1989	42.9
1990	47.6
1991	52.4
1992	39.3
1993	42.9
1994	35.0
1995	50.4
1996	57.5
1997	67.6
1998	40.0
1999	54.9
2000	50.4
2001	35.2
2002	47.0
2003	34.0
2004	43.7
2005	40.7
2006	37.8*

* Year to date

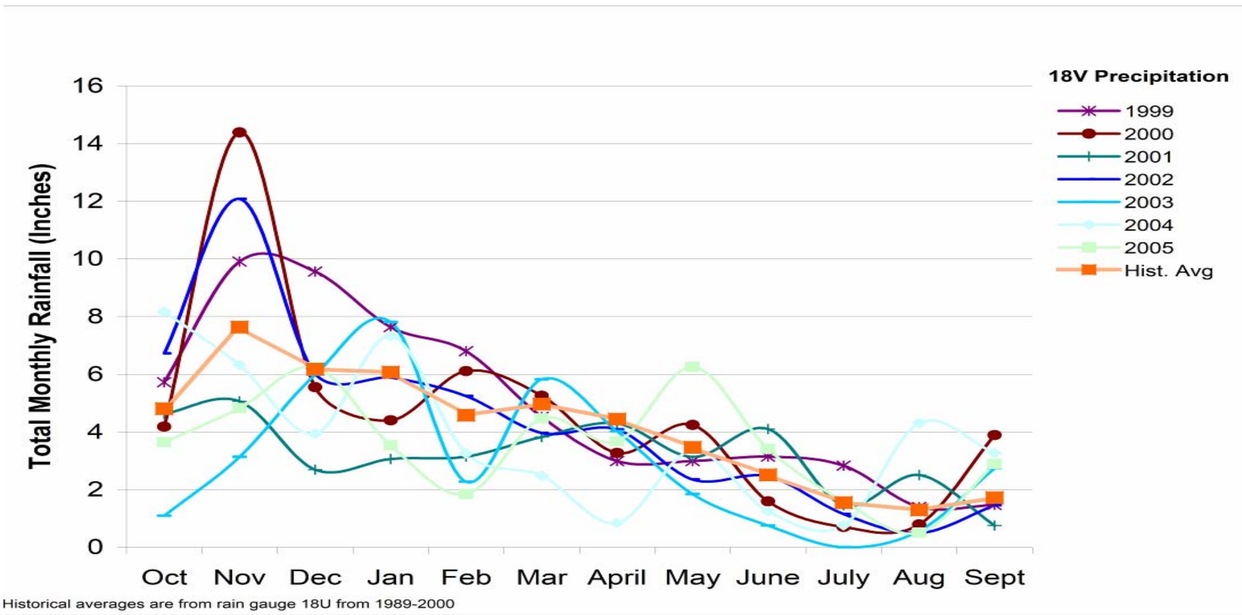


Figure 4.2.1.1 Total monthly precipitation at King County gauging station 18V during 1999 to 2005. The Historical average represents the mean precipitation at King County gauging station 18U during 1989 to 2000.

4.2.2 Streams, Wetlands, and Facilities

Beginning in 1998, King County Water and Land Resources Division (WLRD) established the UPD hydrologic monitoring program to measure stream flows and wetland water levels in the Trilogy and Redmond Ridge UPD's to prepare for post-development monitoring. The primary goal of the hydrologic monitoring was to determine if there were unexpected changes in the hydrology of the streams and wetlands due to the construction of the UPDs. Changes in land cover related to the UPDs has created unexpected hydrologic effects that can be resolved in the streams and wetlands that drain the area. This section of the report reviews the monitoring data from the stream and wetland gauges.

Hydrologic data were collected at seven locations in and around the UPDs (Table 4.2.2.1). Stations 18f and 18b are within the area influenced by the Redmond Ridge developed areas, while 02c, 02d, 53a, 53b, and 53c monitor the systems affected by the Trilogy at Redmond Ridge development. For all but one gauge the period of records are split in two – one pre-development, and one concurrent with development.

Table 4.2.2.1 Summary of hydrologic gauging stations and period of record for each respective station. See Figure 2.0 for location of these gauges.

Site ID	Name / Location	Period of record
02C	Colin Creek, north fork/flume at Lake of the Woods Development	4/10/1991 - 6/29/2004 and 9/11/2000 - 7/6/2006
02D	Upper Colin Creek, south fork/flume, Tributary 0132	4/11/1990 - 9/30/1993 and 9/20/2000 - 7/21/2006
53B	Snoqualmie River Tributary, Unnamed Creek, west fork, flume	4/10/1990 - 6/29/1994 and 10/1/2000 - 6/8/2005
53C	Snoqualmie River Tributary, Unnamed creek, north fork/flume	6/7/1990 - 6/29/1994 and 3/25/2002 - 5/31/2005
53A	Adair Creek	4/11/1990 - 6/28/1994 and 3/5/1998 - 8/9/2006
18B	Northridge Evans Creek #4 (Redmond Block South), south weir, hobo level logger	10/1/1988 - 10/1/1993 and 9/13/1999 - 11/27/2004
18F	Rutherford Creek at NE 76th	10/4/2000 - 8/10/2006

All sites have predevelopment data except for Rutherford Creek (Site 18F, Figure 4.2.2.1). Development in both UPDs was assumed to take two years for each division built – year one for clearing and grading, year two for construction. Division construction periods are staggered in time for all gauge locations. For most gauges development occurs over the latter part of the record. As such no post-development data are available for a comparison in a before/after scenario. This analysis uses the software package, Indicators of Hydrologic Alteration, version 7 (The Nature Conservancy 2005), developed to examine trends over the record and potential differences in hydrology between "before" and "during" UPD construction.

Streams

The two Colin Creek tributaries are behaving as expected. The Unnamed Creek bypass is functioning and data from the two Unnamed Creek gauges are consistent with expectations with respect to summer low flows and storm peak magnitudes. The Adair Creek bypass is functioning, but does not seem to carry much flow. Technical problems with the configuration of the bypass make our gauge data somewhat less reliable than we need for accurate assessment of the bypass performance. Adair Creek is exhibiting some affects consistent with an urbanizing watershed, i.e., increased "flashiness." A close look at the setting of the bypass level is indicated.

Wetlands

Wetlands BBC45 and SR24B remain within the seasonal fluctuations that were observed during the baseline data collection period. We observed unexpected water level fluctuations in wetland BBC44. The wetland fluctuated less than 1 foot during the baseline data collection period of 1990–1994. The fluctuation in water year 2003 was 1.5 feet, which was a 50% increase in variability in that system. During summer 2005, the wetland water level rose 1.5 feet from June to September. Normally, stream and wetland stage would decrease through the dry part of the summer. This abnormal hydrologic behavior could indicate the presence of beavers working at the outlet of the system that caused the summer increase in stage. During the summer of 2006, the wetland stage behaved normally (i.e., the stage dropped during the dry months), possibly indicating that the beaver influence on the system was removed. In any case, large increases in annual wetland level fluctuation and level rises counter to the normal seasonal variation are outside of the expected results. The Evans Creek wetlands are very affected by beaver activity. Other than some construction related events, the wetlands do not show unexpected changes in hydrologic behavior.

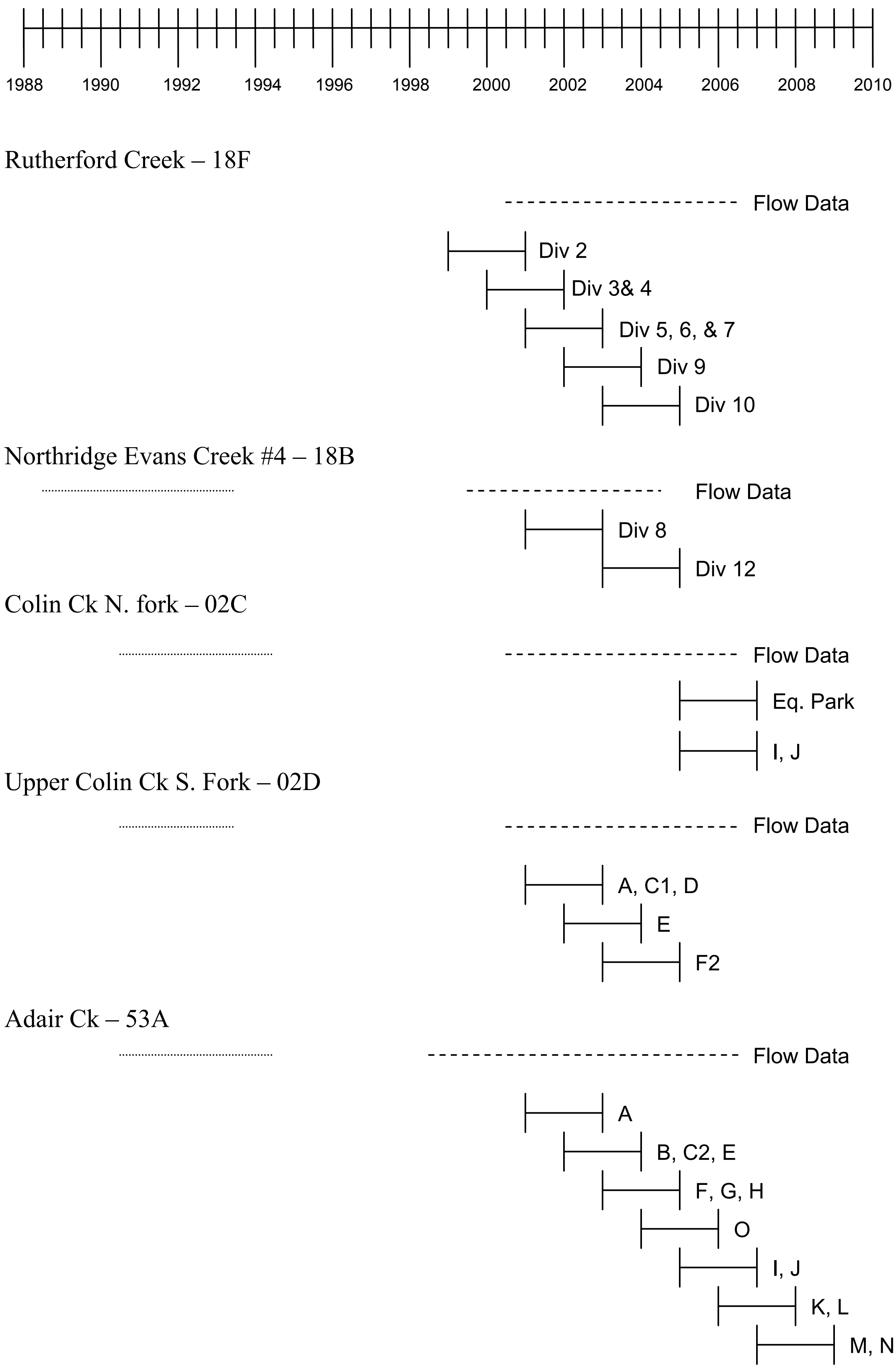
Facilities

Monitoring of drainage facilities is slated to begin currently and will continue until 5 years after 75 percent buildout has occurred in respective basins. Fifty three stormwater facilities have been or will be constructed for this development. Of those, seven facilities have been identified as representative and will be included in this monitoring program (Table 4.2.2.2). Some of the stormwater facilities were constructed early in the development, and some have been constructed more recently. However, as of yet, none of the facilities have exceeded their capacity relative to storm events since the facilities came online. Additionally, monitoring activities range from water quality to continuous water stage monitoring.

Table 4.2.2.2. Stormwater retention and detention (R&D) facilities to be monitored as required under the UPD permit.

Parameters include water level (stage) and discharge from the facility.

Site Name	Gauge Number	Parameters	Data Recorder
Bear Cr. 2 Pond #1	RR-BC2_1	Stage	Manual
Evans Cr. 3 Pond #3	RR-ECW3_3	Stage	Manual
Mackey Cr. 1 Pond #1	RR-MC1_1	Stage	Manual
Mackey Cr. 2 Pond #1	RR-MC2_1	Stage	Manual
Mackey Cr. 4 Pond #1	RR-MC4_1	Stage	Manual
BBC Unnamed #1	BBC-UN1	Discharge	Continuous Electronic
Evans Cr. 1 Pond #1	EC1B1	Discharge	Continuous Electronic


1988
1990
1992
1994
1996
1998
2000
2002
2004
2006
2008
2010
Rutherford Creek – 18F
Flow Data
Div 2
Div 3& 4
Div 5, 6, & 7
Div 9
Div 10
Northridge Evans Creek #4 – 18B
Flow Data
Div 8
Div 12
Colin Ck N. fork – 02C
Flow Data
Eq. Park
I, J
Upper Colin Ck S. Fork – 02D
Flow Data
A, C1, D
E
F2
Adair Ck – 53A
Flow Data
A
B, C2, E
F, G, H
O
I, J
K, L
M, N

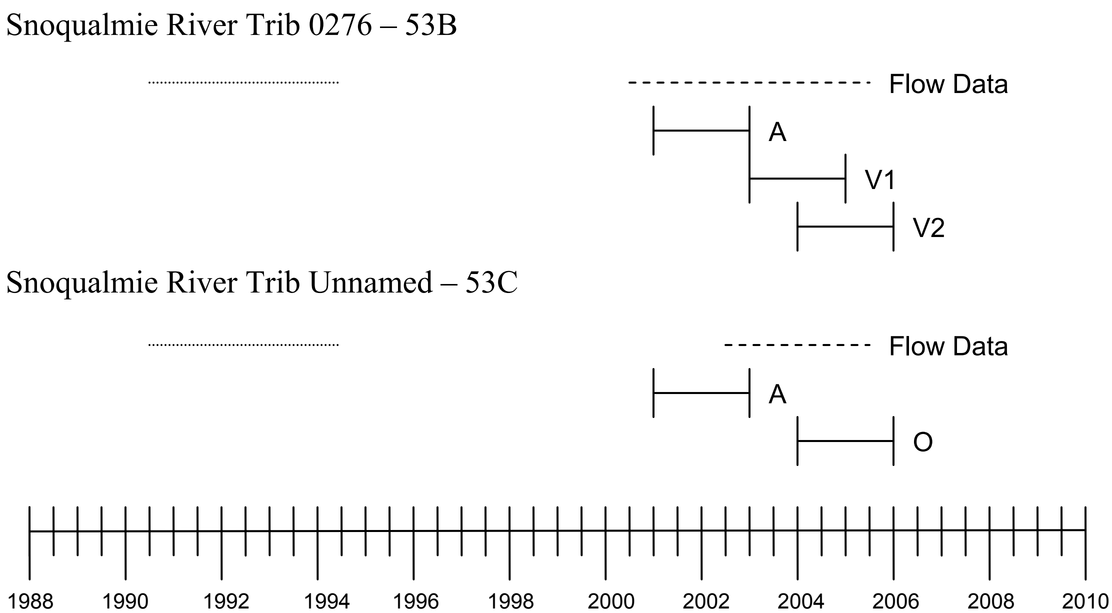


Figure 4.2.2.1 Available flow data for each gauge along with the phasing of construction for that gauge. Stream names and gauge identifiers are located along the left side of the figure. Date axes are at the top and bottom of the figure. Development phasing is shown to the nearest half year increment below, the dotted line marked "Flow Data" which corresponds to the availability of the mean daily flow record at that location..

Hydrologic data for Adair Creek, King County gauging station 53A, was analyzed and presented as a representative example of changes in hydrology in a before and after construction comparison (Figure 4.2.2.1). Peak flow estimates for existing and developed land use in the UPDs were created using HSPF (Bicknell 1997) and a historical rainfall record from 1950-1991. These estimates were made for monitored basins within the UPDs. However, actual return frequencies cannot be estimated with certainty from a short period of record, as is the case for all monitored sites within the UPD area. To assess the impact of the development on hydrology, the hydrologic analysis package Indicators of Hydrologic Alteration (IHA, (Conservancy 2005) was used to investigate high flows by analyzing the gauged record at this station (Figure 4.2.2.2). Because it is impossible to accurately extrapolate out to a 25-year or longer flow probability from a five year data set, within year peaks (high pulses and small floods in IHA) were examined to better assess any trends in flows that might be correlated with changes in ecosystem function (Timm in prep). In this sample dataset, small floods did not occur with sufficient frequency to effect changes in stream condition.

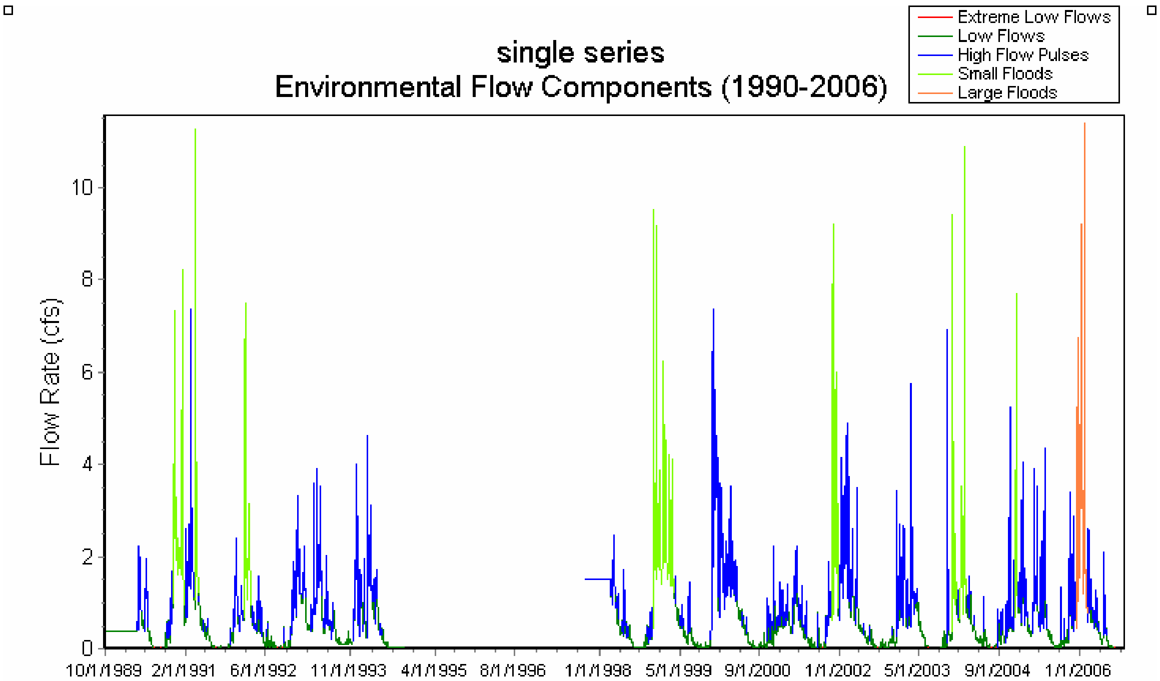


Figure 4.2.2.2 Indicators of Hydrologic Alteration (IHA) modeled output of flow metrics computed for the pre- (1990 – 1993) and post-construction (1999 – 2006) periods of the Trilogy UPD.
The different colors represent IHA metrics as calculated from the flow record.

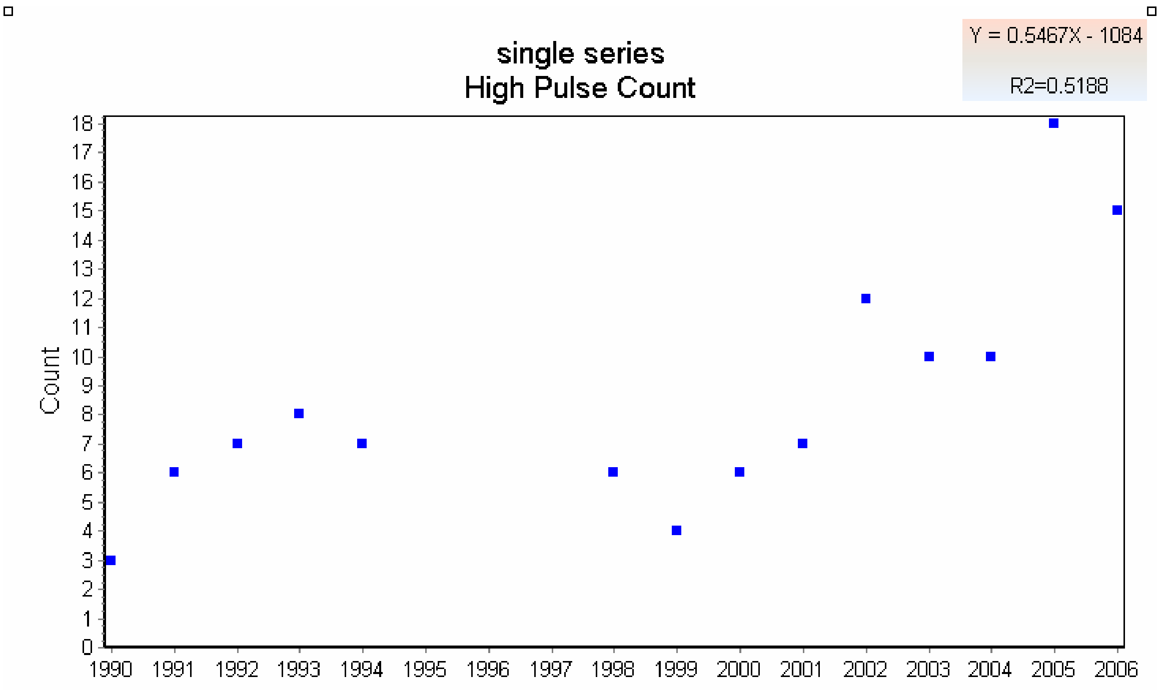


Figure 4.2.2.3 Graph of the number of times flows were calculated at "High Pulses."
Note the positive slope of the curve indicating an increase in high pulses. Also, the pre-construction period (1990-1993) is roughly on track with the early construction phases of the project (Figure 4.2.2.1).

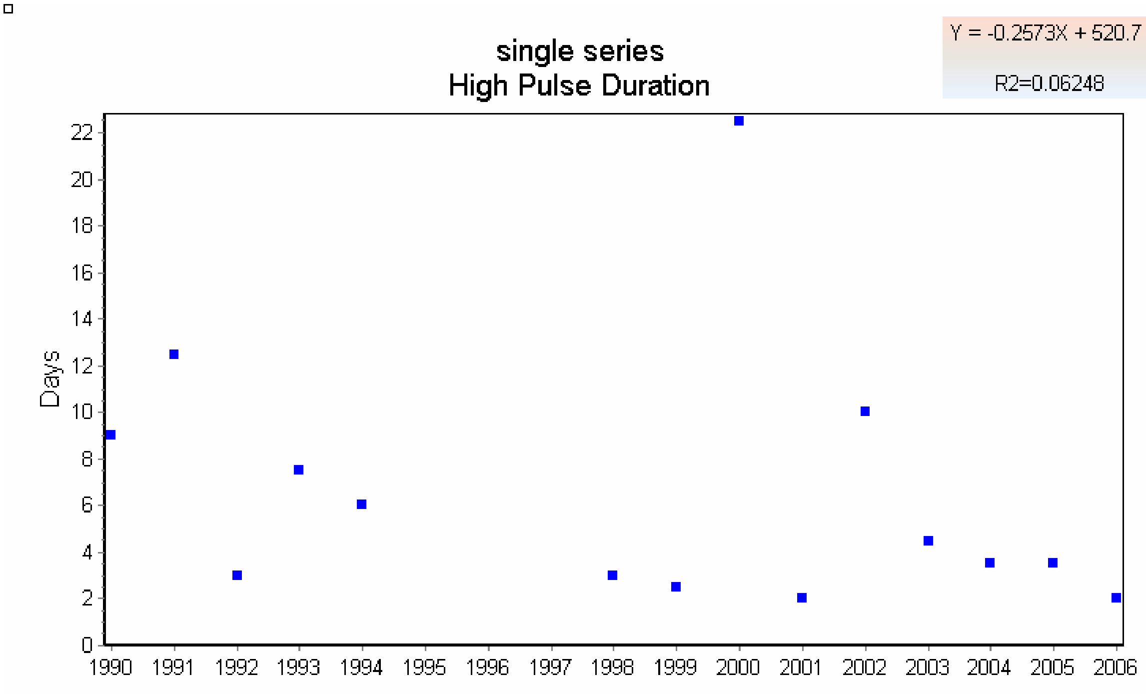


Figure 4.2.2.4 Graph of the duration of "High Pulses" in days as calculated using IHA software.
Note that 2000 was the highest year in terms of the length of time streams were exhibiting high flows.

Adair Creek (53A) showed an increase in high pulses (Figure 4.2.2.4) with the increase in development. This system appears to be getting "flashier" over time as measured by within-year pulse metrics. For Adair the trend was for increases in high pulse events (Figure 4.2.2.3) with little change to the duration of flows. Development in the Adair Ck drainage began in 2001, and new phases began every year after until 2007 resulting in a complete "build out" projected in 2009. For each year after 2001, the flow record reflects changes to the hydrologic cycle from removal of soil water storage and the addition of impervious surface. Although insufficient data exist for a statistical test, the most recent years – reflecting the greatest alteration of landscape – also have the highest pulse count. This suggests the basin has already become a "flashier" system.

4.2.3 Stream Cross-Section Stability Analyses

There were seven small streams that were monitored in terms of their cross-section stability as part of this study. Channel geometry as an indicator of stream channel stability was monitored at two to four cross-sections in each stream during the period of the UPD construction with at least one year of baseline conditions collected before construction commenced. Cross-sections were measured from 1 to 12 times depending on the stream (Table 4.2.3.1).

The objective was to measure whether changes in watershed hydrology (i.e., land clearing and conversions to impervious surfaces) interfered with watershed processes to the extent that there was a measurable effect on stream channel stability. Under normal watershed conditions, stream channels will respond to the prevailing hydrologic conditions by eroding, routing, and depositing materials (i.e., water, sediment, and LWD) relative to their inputs in a condition that has been widely considered to be a quasi-equilibrium (Wolman 1967b). Systems with hydrologically intact watersheds are characteristically in this type of balance. However, when disturbances alter the amount and timing of the delivery of materials to the channel, a stream system's ability to process these changes is reflected in the channel stability (Lancaster 2001). In particular, the hydrologic effects of urbanization can cause profound shifts in disturbance regime that are manifest as changes in a streams competence to rout materials and ultimately its channel form (Wolman 1967b, Konrad 2005).

Table 4.2.3.1 Year and location of stream cross-section surveys in the seven UPD streams.
"Y" indicates that the surveys were performed at that location during the respective year.

	Adair					Colin Cr. N.				Colin Cr. S.				Evans Cr.				Evans Cr. E.				Mackey Cr.				Rutherford Cr.				Unnamed Cr.			
Year Surveyed	*1*	*2*	*2b*	*3*	*4*	*1*	*2*	*3*	*4*	*1*	*2*	*3*	*4*	*1*	*2*	*3*	*4*	*1*	*2*	*3*	*4*	*1*	*2*	*3*	*4*	*0*	*1*	*2*	*3*	*1*	*2*	*3*	*4*
1991			Y	Y		Y																											
1992				Y		Y								Y*	Y*	Y*							Y			Y*				Y*	Y*		
														Y†	Y†	Y†										Y†				Y†	Y†		
1993														Y*	Y*	Y*										Y*							
														Y†	Y†	Y†										Y†							
1998	Y	Y		Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y						Y				Y	Y	Y	Y	Y	Y		
1999					Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y						Y	Y			Y	Y	Y	Y	Y	Y		
2000	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y						Y	Y			Y	Y	Y	Y	Y	Y		
2001	Y	Y	Y	Y	Y	Y	Y	Y	Y																	Y	Y	Y	Y				
2002	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y		Y	Y	Y		Y	Y			Y	Y	Y	Y	Y	Y		
2005	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y						Y	Y			Y	Y	Y	Y	Y	Y		
2006	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y						Y	Y			Y	Y	Y	Y	Y	Y		

* Streams measured during spring.
†Streams measured during fall.

Methods

Stream channel stability in the seven streams that occur within or are affected by the UPDs was monitored by measuring streambed elevations at prescribed cross-section locations within the channel network following the monitoring requirements set forth under the permit conditions for the development. Spot elevations along cross-section measurements were analyzed to determine if changes in watershed hydrology due to land development, and storm water management could be detected. Streambed elevation was measured using a surveyor's auto level telescope and stadia rod following standard protocols (Harrelson et al. 1994). All cross-section spot elevations were measured relative to an iron pin monument on the top of the left bank and in some years, longitudinally connected using the left bank iron pin in cross-section 1 as the common point of beginning among all cross-sections for a given stream reach under investigation. In general, cross-sections were enumerated from downstream to upstream (i.e., the downstream-most cross-section was given the number 1).

We used an analysis of variance (ANOVA) for non-parametric data (i.e., a Kruskal-Wallis test) to test the hypothesis that there were no differences in relative stream cross-section bed elevations among years. In more than 60% of comparisons, there was a significant reason to reject that hypothesis. We used a Mann-Whitney test between pairs of years with a Bonferoni adjustment to address multiple comparison error. The "H" statistical data were reported as χ^2 values because for large sample sizes, the distribution of H is approximated by the χ^2 distribution (Zar 1999). In addition, for cross-sections with enough inter-annual geomorphic differences to reject the null hypothesis, despite the violation of the assumption of normality, one-way ANOVA was employed with a Bonferoni post-hoc test to indicate which pairwise comparison lead to the rejection. In this way, the ANOVA was used to inform the Mann-Whitney tests.

Results and Discussion

In most cases, the stream channels under investigation appear to be relatively unchanged geomorphically. That is, the normal sediment, wood, and water dynamics seem to be functioning similarly to pre-UPD construction conditions. However, 62.5% of the time there were channel changes at some cross-sections when compared to other years that were statistically significant at the 95% confidence level. In many of those cases, the absolute differences in the channel were small and could easily be attributed to various types of measurement error. These errors may have been due to differing degrees of technical competency among crews, instances where a respective cross-section monument changed, or slight differences in the path taken across the stream when measurements were made. In particular, when changes among years are not consecutive, the geomorphic significance of the statistical difference should be evaluated. Comparative figures of cross-section relative elevations are in Appendix 4.2.3.1.

In other cases, there were measurable changes in the stream cross-section geometry (i.e., Adair Creek and Rutherford Creek). In these instances, it appears that wood and sediment recruitment and storage in the channel affected the cross-sections during the study period. In the case of Rutherford Creek, there was significant sediment aggradation in cross-sections 2 and 3. Changes in channel hydraulics at these cross-sections relative to their upstream counterparts (increase in channel width downstream of a sharp bend in the channel) result in a decrease in the stream's competency to transport sediment, depositing sediments from some upstream source (Julien

2002, Abbe 2003). The fact that this deposition occurred during 2005 to 2006 and did not occur during other periods under investigation is somewhat curious. However, due to the extensive wetland complex downstream of the UPD boundary, and upstream of the monitoring location, it is highly doubtful that this sediment deposition resulted from any activity in the UPD. The channel changes in Adair Creek can be attributed to the large wood that fell in the channel, disruptting the sediment dynamics that existed previous to the LWD recruitment event (Abbe 2003). However, the hydrologic changes to the Adair Creek sub-watershed have indicated that there are changes in the volumes, rates, and timing of flows in this system that could possibly contribute to shifts in the sediment dynamics and channel form (see Hydrology Section this manuscript; (Konrad 2005).

Adair Creek

The monitoring site is located approximately 30 meters downstream of King County flow gauge 53A (Figure 2.0). Monitoring began at this site in December of 1991 with the installation of two cross-sections ("A" now designated as "3," and "B" now designated as "2") and a longitudinal profile measurement by GeoEngineers. A second set of baseline measurements was taken by GeoEngineers in October of 1992. When monitoring began in 1998, two more cross-sections were added, one upstream and one downstream of the original two. Cross-section B, established by GeoEngineers, was not found upon return in 1998. Cross-section 2 was established as close as could be determined to the original position and is probably within 2 m of the original site.

Cross-Section 1:

There was a statistically significant difference among all years of data at this cross section (Table 4.2.3.2). This cross-section was approximately 5 m wide except for in 2006, the point of beginning was moved approximately 7 m because the tree in which the iron pin was nailed fell into the channel. So, the 2006 cross-section was normalized to the pin on the right bank instead. Total annual change in cross-sectional area ranged from an increase during 2001 to 2002 of 0.107 m^2 to a decrease in area of 0.571 m^2 during 1998 to 1999. The greatest total change in cross-sectional area when all years were compared to the 1998 baseline condition was from 1998 to 2006 when the channel area decreased 0.936 m^2 (range 0.150 to 0.936).

Cross-Section 2:

This cross-section did not significantly change during the study period (Table 4.2.3.2). The channel at this location was approximately 9 m wide. The smallest interannual change in cross-sectional area was during 1999 to 2000 (+0.018m^2), and the largest change was during 2002 to 2005 (-0.315 m^2). When all years were compared to the 1998 baseline condition, 2006 exhibited the greatest change with a decrease of 0.020 m^2 (range 0.001 to 0.020 m^2).

Cross-Section 2b:

Cross-section 2b did exhibit significant change during the study period (Table 4.2.3.2). Post-hoc tests indicate that 2002 was significantly different from all years. Similarly, 2006 was significantly different from 2005, but not different from all other years. All other years are not statistically dissimilar at the 0.05 level. The largest change in channel cross-sectional area occurred during 2002 to 2005 when the cross-sectional area increased by 0.32m^2. However, there was substantial uncertainty about the exact location of the cross-section monuments in the field which resulted in the cross-section being re-monumented in 2005.

Cross-Section 3:

During 2006 this cross-section was significantly different from all other years (Table 4.2.3.2). However, when 2006 data were removed from the analysis, the hypothesis was not rejected at the 0.05 significance level. We attribute the changes to this cross-section to the recruitment, prior to the 2005 survey, of a large piece of wood to the channel that stored a substantial amount of sediment. For all inter-annual comparisons, the stream channel was very stable except for the 2006 survey which indicates a significant aggradation of sediment and wood in the channel (approximately 5.1 m^2). In addition, when all years were compared to the pre-construction time periods, there was no significant change until 2006. However, given the stability displayed in the rest of the channel (upstream and downstream), the changes exhibited in this cross-section can be attributed to the localized accumulation of large wood and sediment in the channel.

Cross-Section 4:

Cross-section 4 is not significantly different during the study period (1998 to 2006, Table 4.2.3.2). This cross-section did exhibit downcutting during 2005 to 2006 that resulted in an increase in cross-sectional area of approximately 1.5 m^2 and a total change of approximately 1.7 m^2 during the period under investigation. Some of this may be explained by the relatively large flows during the 2006 water year. A comparison of the mean annual discharge for 2006 and the period water year 2001 to 2006, indicates that the 2006 mean annual discharge was approximately 39% higher than normal. In addition, 31% of the flows during the 2006 water year exceeded the mean annual flow for that year and 43% of the flows during that year exceeded the overall mean flow for the system (see section on Hydrologic Analyses, this document).

Colin Creek North

The north fork of Colin Creek drains the north end of the Trilogy property. The monitoring site is located approximately 40 meters upstream of King County flow gauge 02C (Figure 2.0). Monitoring began at this site in December of 1991 with the installation of a single cross section ("A" now designated as "1") and a longitudinal profile measurement by GeoEngineers. GeoEngineers collected a second set of measurements during October of 1992. When construction monitoring began in 1998, three more cross sections just upstream were added.

Cross-Section 1:

The bed surface of this stream was monitored 9 times beginning in 1991 and last surveyed in 2006. The stream channel was approximately 6 m wide at this cross-section. There were no statistically significant differences among all years of data at cross section 1 (Table 4.2.3.2). Inter-annual geomorphic changes at this station were very small. The largest change in cross-sectional area was a decrease of 0.14 m^2 during 1991 to 1998. All other inter-annual changes were less than 0.1m^2. Overall changes when all years were compared to 1991 were less than 0.1m^2. This is relatively stable cross-section that has shown no geomorphic response to changes in hydrology.

Cross-Section 2:

This cross-section was surveyed annually between 1998 and 2006. The stream channel was approximately 6 m wide at this location and was stable during the study period. There were no statistically significant changes inter-annually, or among all years when compared to the 1998 baseline conditions (Table 4.2.3.2).

Cross-Section 3:

The channel at this cross-section is approximately 8 - m wide and has shown no statistical difference inter-annually or when each year's channel geometry was compared to the baseline 1998 conditions (Table 4.2.3.2). The surveys at this location were performed on an annual basis 7 times during 1998 to 2006 with the exception of 2003 and 2004. The largest inter-annual change occurred during 2005 to 2006 when the channel exhibited some slight aggradation (0.24 m^2). Similarly, the largest change when all years were compared to the 1998 baseline conditions occurred during 2006 when that comparison yielded a slight aggradation of approximately 0.21 m^2. All other changes were smaller than 0.02m^2.

Cross-Section 4:

The channel in this location was approximately 10 - m wide and like the other cross-sections in this reach of the stream, very stable. There were no statistically significant changes to this cross-section geometry either inter-annually, or when each year was compared to the 1998 baseline conditions (Table 4.2.3.2). The largest inter-annual change occurred during 2001-2002 when the channel eroded very slightly (0.06m^2) and the largest change when compared to the 1998 baseline conditions was during 2006 when the channel eroded approximately 0.3 m^2.

Colin Creek South

The south fork of Colin Creek drains the southwest portion of the Trilogy property. The monitoring site is located approximately 30 meters downstream of the King County flow gauge 02D (Figure 2.0). Monitoring began at this site in December of 1991 with the installation of two cross sections ("A" and "B") and a longitudinal profile measurement by GeoEngineers. GeoEngineers took a second set of baseline measurements in October of 1992. Neither of these two cross sections could be located upon return in 1998 at the beginning of construction monitoring. Four new cross sections were established in the same general area as the GeoEngineers sections.

Cross-Section 1:

There were no statistically significant differences among all years of data at this cross section (Table 4.2.3.2).

Cross-Section 2:

Similar to cross-section 1, this location showed no significant changes during the study period (Table 4.2.3.2).

Cross-Section 3:

This cross-section showed significant change in a statistical sense. But, in an absolute geomorphic sense, the magnitude of change was very small (Table 4.2.3.2).

Cross-Section 4:

At this cross-section, 1998, 1999, and 2000 are not significantly different from each other. However, 2006 is significantly different from 1998 and 2000 at the 0.05 level (Table 4.2.3.2). At the 0.10 level, 2006 is also significantly different from 1999. 2002 and 2005 are not significantly different from 1999 and 2000, however. On an absolute basis, 2006 is only slightly different than any other year. The largest mean difference in cross-sectional area was -0.16 m^2 (when compared with 1998) and the average mean difference among all years was -0.11 m^2. Despite the statistical significance of the differences, the overall difference was small and probably inconsequential geomorphically.

Evans Creek

The middle fork of Evans Creek drains the south-central section of the Redmond Ridge property (Figure 2.0). Evans Creek is another small tributary in the Bear Creek system. The stream in this location is very low gradient (0.012%) and very low energy due to the gradient. The monitoring site is located approximately 100 meters downstream from the King County flow-gauging weir known as the "south weir." Monitoring began at this site in April of 1992 with the installation of two cross sections ("A" now designated as "1," and "B" now designated as "2") and a longitudinal profile measurement (GeoEngineers 1992, 1995). Baseline measurements were collected in November of 1992 and in May of 1993. When construction monitoring began in 1998, one more cross section was added downstream.

Cross-Section 1:

Te stream cross-sectional geometry did change significantly at this location during 1998 to present (Table 4.2.3.2). 1998 and 2005 were significantly different from all years, 2006 was significantly different from 1999 and 2000. Given the extremely low energy of this stream, the potential for change due to increases in hydraulic energy is very low. Therefore, the differences in this cross-section are likely due to measurement discrepancies among years as opposed to changes in the physical condition of the channel.

Cross-Section 2:

Similar to cross-section 1, the channel geometry at this location is statistically different from 1998 to 2006 (Table 4.2.3.2). However, given the low gradient conditions and the stable appearance of the channel, the differences are likely attributable to measurement error.

Cross-Section 3:

Cross-sectional geometry at this location is statistically different among years during the study period (Table 4.2.3.2). However, the difference is likely due to measurement error.

Mackey Creek

Mackey Creek drains the northwest portion of the Redmond Ridge property. The monitoring site is located approximately 10 meters upstream of the Novelty Hill culvert (Figure 2.0). Monitoring began at this site in June of 1992 with the installation of one cross section ("A") (GeoEngineers 1992, 1995). This cross section was not located in 1998 at the beginning of construction monitoring, so a new cross ("1") section was established in the same general area as the GeoEngineers cross section. The GeoEngineers section was found in 1999 and designated as "2"; both cross sections will be included in future monitoring.

Cross-Section 1:
Cross-section geometry at this location is statistically different from 1998 to 2006, or among years within the study period (Table 4.2.3.2). This difference must be an artifact of measurement error because the channel has not had surface flow during the study period.

Cross-Section 2:
Cross-section geometry at this location is statistically different from 1999 to 2006, or among years within the study period (Table 4.2.3.2). However, as with cross-section 1, the differences must be a function of measurement error because there has been no water in the channel during the study period.

Rutherford Creek

Rutherford Creek drains the southwestern edge of the Redmond Ridge property. The monitoring site is located just downstream of the Union Hill Road culvert (Figure 2.0). Monitoring began at this site during March, 1992 with the installation of two cross sections (“A” now designated as “0,” and “B”) and a longitudinal profile measurement (GeoEngineers 1992, 1995). Baseline measurements were taken in October of 1992 and also in May of 1993. When construction monitoring began during 1998, three more cross sections were added downstream of the 225th Avenue NE culvert. In addition, the original cross section “B” was abandoned.

Cross-Section 0:
Cross-section geometry at this location was statistically different from 1991 to 2006. Among years within the study period, 2001 was different from 1991 at the 0.05 significance level. All other yearly comparisons were not significantly different. Also, 2001 was not different from any other years in pairwise comparisons (Table 4.2.3.2). The largest change between successive monitoring periods was between the spring and fall surveys during 1993 when the channel cross-sectional area changed by 0.17 m^2. The largest change from 1991 baseline conditions through 2006 occurred in 2001 when the channel cross-sectional area changed by -0.16m^2. Overall, the channel is very stable in this location. The statistical difference in channel measurements is not a good indicator of geomorphic change at this cross-section.

Cross-Section 1:
Cross-section geometry at this location is statistically different from 1998 to 2006 (Table 4.2.3.2). However, the channel appears relatively stable at this location despite the small statistical significance of the measurement differences.

Cross-Section 2:
This cross-section is significantly different in 2006 from all other years except 2000 when it was also nearly significantly different at the 0.05 level (Table 4.2.3.2). Further examination of the data suggests that beginning during 2005, there was sediment aggradation in this cross-section that decreased the cross-sectional area. The decrease was not statistically significant between 2002 and 2005. However, during 2005 to 2006, the change was significant. Geomorphically, the channel widens and creates a floodplain at this transect. There is a 43% increase in channel width between cross-sections 1 and 2. So, there would be a corresponding decrease in stream power making it a deposition zone. Presumably, there was some event upstream during the

period between 2005 and 2006 when fine sediments were introduced to the system and routed through the narrower stream channels to the vicinity of cross-section 2.

Cross-Section 3:
This cross-section is significantly different in 2005 and 2006 from the baseline conditions surveyed in 1998 at the 0.05 level (Table 4.2.3.2). Beginning during 2005, there was sediment aggradation in this cross-section that decreased the cross-sectional area. The decrease was not statistically significant between 2002 and 2005. However, in the 2005 data, there was a decrease in cross-sectional area at this location of greater than 2 m^2 when compared to the 1998 cross-sectional geometry. The 2006 data when compared to the 1998 cross-section, similarly reveal a greater than 2 m^2 decrease in cross-sectional area indicating a substantial sediment aggradation in this area of the channel and adjoining floodplain.

Unnamed Creek

Unnamed Creek drains the southeast corner of the Trilogy property. The monitoring site is located approximately 230 meters downstream from the NE 243rd Street culvert (Figure 2.0). Monitoring began at this site in February of 1992 with the installation of two cross sections ("A" now designated as "1," and "B" now designated as "2") and a longitudinal profile measurement (GeoEngineers 1992, 1995). Baseline measurements were collected during October 1992. When construction monitoring began in 1998, cross section "A" was easily relocated. Cross section "B" however, was not located and a new cross section was established as close to the old location as could be determined and is probably within 2 - m of the original site.

Cross-Section 1:
Cross-section geometry at this location is not statistically different from 1998 to 2006, or among years within the study period (Table 4.2.3.2).

Cross-Section 2:
Cross-section geometry at this location was statistically different from 1998 to 2006 (Table 4.2.3.2). In particular, 2005 was different from baseline conditions measured during 1992 and 1998 and when compared to 2002. Total aggradation at this location measured almost 2.4 m^2 when 2005 conditions were compared to baseline. However, year to year sediment aggradation at this location was relatively small with 2002 to 2005 being the largest change (0.501 m^2) in cross-sectional area.

Table 4.2.3.2 Results of Kruskal-Wallis comparison of means.
Relative elevation measurements for each cross-section in each stream were compared against all years of measurements to test for statistical differences among years. Probability values (p Value) are reported at the 95% confidence level. See Appendix 4.2.3.1 for graphical comparisons of each stream cross-section among all years.

STREAM	X-SEC	STATISTICALLY DIFFERENT	χ^2	p Value
Adair	1	Yes	25.784	0.000
	2	No	0.801	0.992
	2b	Yes	20.651	0.001
	3	Yes	125.92	0.000
	4	No	11.65	0.070
Colin North	1	No	0.895	0.999
	2	No	1.851	0.933
	3	No	0.735	0.994
	4	No	2.896	0.891
Colin South	1	No	0.523	0.991
	2	No	2.399	0.792
	3	Yes	4.564	0.471
	4	Yes	30.547	0.000
Evans	1	Yes	52.146	0.000
	2	Yes	7.527	0.184
	3	Yes	7.361	0.195
Mackey	1	Yes	9.597	0.143
	2	Yes	5.046	0.410
Rutherford	0	Yes	6.093	0.413
	1	Yes	5.798	0.446
	2	Yes	44.123	0.000
	3	Yes	28.129	0.000
Unnamed	1	No	3.092	0.668
	2	Yes	8.655	0.124

Conclusions

Most of the statistically significant differences in stream channel cross-sections that were measured during this study are likely attributable to measurement errors that were magnified due to sample size effects. Despite the precision of surveyor's equipment, error among years can be introduced by very slight variations in path measured as the survey crew traverses the channel. Starting and ending a cross-section at monumented points guarantees that the ends of a cross-section will remain consistent among years unless the monuments are moved or lost. But, differences in rod placement, for example, can make measurable differences even in the absence of real geomorphic changes in the channel. For example, if a surveyor measures the elevation of the top of a rock in one survey year, and the side or back of it in successive years, the difference could be significant. Further, similar deviations in measurement along an entire cross-section are magnified arithmetically. With that said, differences measured in Adair Creek and Rutherford Creek do represent geomorphically significant changes in cross-sectional geometry. The changes in these two cross-sections may be a reflection of some upstream change in watershed hydrology that is manifest as a chronic impact, or they could represent the channel response to a single event (e.g., a piece of large wood falling in the channel). In either case, continuing to monitor these locations in subsequent years should be required and if further geomorphic changes are observed, corrective actions should be implemented.

4.3 Chemical Assessment

4.3.1 Water Quality

Overview

High water quality is essential for proper functioning of aquatic ecosystems and survival of organisms within. Yet, anthropogenic changes in a watershed such as increased development and urbanization, often alter water quality through increased delivery or concentration of nutrients, pesticides, organic chemicals, and heavy metals found in urban runoff and treated wastewater (Klein 1979, Heaney and Huber 1984, May et al. 1997, Brown et al. 2005). Prior to development of the Redmond Ridge and Trilogy UPDs, the area consisted primarily of second and third growth forests most recently logged around 1936 (King County 1999). This conversion from forest to urban could degrade water quality in the headwater wetlands and ultimately impact downstream aquatic ecosystems. The majority of wetlands on the plateau have naturally low pH, alkalinity, and hardness typical of bogs[17]. Therefore, urban runoff which is typically nutrient and cation enriched, poses a concern for maintaining the integrity of these wetlands and their receiving waters (Kulzer et al. 2001).

Mitigation techniques such as stormwater detention, indirect discharge, groundwater infiltration, and wetland buffering coupled with natural characteristics such as flat topography and a lack of definitive surface channels were predicted to effectively negate any potential adverse water quality impacts resulting from UPD development (Herrera Environmental Consultants Inc. 1992, Beak Consultants Incorporated 1995). In order to assess whether these predictions held true, water quality sampling was conducted annually from 2002 to 2006 three times between January and April at twenty locations, including a subset of pre-development baseline locations. Stream or wetland water quality degradation including violations of state and federal water quality standards or changes from pre-development conditions for pH, temperature, turbidity, or other parameters constitute exceedances of the threshold criteria defined by the Trilogy and Redmond Ridge monitoring plans and could trigger corrective actions (King County 1999, 2001).

[17] According to WAC Chapter 173201a (1997): “Bog” means those wetlands that are acidic, peat forming, and whose primary water source is precipitation, with little, if any outflow.

Procedures

Twenty locations, including seven streams primarily downstream of the UPDs and thirteen onsite wetlands, were sampled annually in January, February, and April from 2002 to 2006 (Figure 2.0). *In situ* measurements were routinely monitored and in 2006 water samples were collected and analyzed for alkalinity, hardness, and total metal concentrations.

For each sampling period temperature, dissolved oxygen, specific conductivity, pH, and turbidity were collected *in situ*. Temperature, dissolved oxygen, and conductivity measures were collected using an YSI model 85 handheld system. Dissolved oxygen was always calibrated at the beginning of each day. In lotic systems, the probe was placed on the substrate and measurements were recorded whereas in lentic environments, the probe was gently bobbed through the water column below the surface because the dissolved oxygen probe is stirring dependant (YSI Incorporated 1998).

We primarily used EM-Reagents color pHast ® paper 4.0-7.0 range to measure pH, although pH 2.5-4.5 or 6.5-10 paper were used when conditions warranted it. Periodically, pH was measured using a variety of meters (e.g., Oakton pHTestr2 and Orion model 230Aplus portable pH meter), however these did not seem consistently reliable and had difficulty stabilizing on a reading in the low pH environments present, possibly because of low battery charge. Therefore, the meter data are not reported here.

Turbidity was estimated using a Hach 2100P Turbidimeter. Samples were collected before any other samples or measurements were taken to minimize disturbance that could impact measurements. Turbidity was always calibrated compared to four standard solutions at the beginning of each sample period (Hach 1999).

Beginning in 2006, water samples were collected in pre-labeled containers for hardness, alkalinity, and total metals concentrations. Two bottles were collected at each site and the hardness samples were preserved with nitric acid in the field. Samples were delivered to the King County Environmental Lab (KCEL) once all samples for that period were collected and chain of custody notes were recorded. Each parameter was analyzed by KCEL using standard operating procedures (King County Environmental Lab 1999, 2003a, 2006d).

In addition to the current sampling procedures, continuous turbidity measurements were conducted at Adair, Colin South, Rutherford, and Unnamed Creeks from 1998 until 2005. However, operational problems with the turbidity probes occurred throughout this time due to calibration, fouling, and erratic readings leading to unreliable results (see (Comings et al. 2001b) for documentation of initial problems encountered). Therefore, continuous turbidity monitoring was eliminated from the monitoring program August 2005 (Appendix 4.3.1). Due to the irregularity of these data, they will not be addressed in this report.

Results & Discussion

As part of the Environmental Impact Statement (EIS) process, stream water quality data was collected monthly for the Redmond Ridge and Trilogy developments between 1981-84 and 1990-92 (King County 1993). Additional wet season storm sampling was conducted in 1992. Surface waters on the site were found to have low values for pH, turbidity, dissolved oxygen, and nutrient levels. When compared to other streams and wetlands in the Puget Sound region, the pre-development results were generally at the high end of water quality ranges typical of areas dominated by relatively undisturbed wetlands (Herrera Environmental Consultants Inc. 1992, King County 1993, Comings et al. 2000). The exception was metals, which had several violations of chronic state water quality standards for lead, copper, and zinc. The low hardness at the UPDs produces a much lower toxic standard for various metals and it was not uncommon to find metals concentrations in undeveloped wetlands that naturally exceeded state standards (Beak Consultants Incorporated 1995). A complete list of water quality parameters for these data can be found in Appendix D of the 1998-99 University of Washington Monitoring Report (Comings et al. 2000). Due to difficulties in confirming the exact locations and methods utilized during this sampling, in addition to changes in the timing of sampling, we will primarily focus our analysis on water quality parameters collected between 2002 and 2006 unless otherwise noted.

Specific Conductivity

Conductivity is a measure of the total dissolved constituents, including nutrients and pollutants, within the water and while there are no established standards for conductivity, it is frequently used as an indicator of water quality because elevated values are associated with urbanization within a basin (Azous 1991). (Clinton and Vose 2006) found that conductivity was over three times higher at urban headwater stream locations than reference locations. Throughout this report when referencing conductivity, we will always be referring to specific conductivity because it is standardized for a temperature of 25º C.

Conductivity across all sample sites and years averaged 41.2 µS/cm (SD = 19.4, n = 293) with a mean of 50.2 µS/cm (SD = 17.1, n = 72) for the 7 stream locations and 36.6 µS/cm (SD = 19.3, n = 221) for the 13 wetland sample sites. These values are relatively low compared to data from other streams and wetlands throughout the region such as a study of Puget Sound wetlands that found an average conductivity of 73 µS/cm in non-urban wetlands and 150 µS/cm in highly urbanized basins (Horner et al. 2001a). However, *Sphagnum* dominated peatlands, which are common at Trilogy and Redmond Ridge, typically have much lower conductivity than conventional wetlands. The UPD wetlands fall within the 17-82 µS/cm range observed for British Columbia coastal peatlands (Vitt et al. 1990). It follows that the headwater streams that drain these wetlands would also have naturally low conductivity levels.

Overall, conductivity has increased in UPD wetlands and streams during the clearing, construction, and post-construction phases of the project from an average of 34.8 µS/cm (SD 13.1, n = 55) in 2002 to 45.4 µS/cm (SD = 22.7, n =60) in 2006 with a peak of 46.0 µS/cm (SD = 24.7, n =58) in 2005 (Figure 4.3.1.1). However, the difference in conductivity between years is not significantly different ($p<0.05$) as determined by the non-parametric Kruskal-Wallis test.

Conductivity also increased over time on a site-specific basis (Figure 4.3.1.2). Only a handful of sites exhibited stable or highly variable conductivity from 2002 to 2006 and these include EC 61, BBC 52, and two sites at BBC 45 all of which are in basins with virtually no UPD development to date. Clearing only began at a large scale in late 2005 in these areas. Conductivity at BBC 44 u/s was also fairly stable with only a very slight increase in conductivity, but it is at the very southern extent of Trilogy and will primarily be influenced by the upstream development around

BBC 45 and BBC 52 that has yet to take place in Redmond Ridge rather than by downstream development in Trilogy. Finally, EC 4 shows considerable variability with no increasing or decreasing trend between years. Conductivity in this basin was highest in 2003 and 2004 when clearing and development of divisions 11 and 12 were taking place. Therefore, these peaks within this basin may be attributable to construction activities, which have the potential through land clearing and grading to cause the greatest short-term impact on water quality, rather than post-development effects.

Several sites have shown a jump in conductivity that was especially strong in the last two to three years (e.g., BBC 26 d/s, ADCW1, SR 24a, 53B) corresponding with new clearing and development at Trilogy within these basins. Wetland EC 3 also had substantial increases in 2005 and 2006 compared with previous years. However, clearing and development within this basin began in 2001 and was mostly completed by 2005. It is not clear what could be responsible for this lag in response at this location; however, the outfall design from stormwater pond ECW1B-No1 has been linked to poor infiltration and buffer saturation of the EC 3 wetland. Or, perhaps these systems are approaching a loadings threshold that is manifest as increased conductivity.

BBC 44 center is a sampling location that has had a steady increase in conductivity since 2003. This location is downstream of the outlet to stormwater pond SWD1, which largely drains the Trilogy golf course. In 2002, the conductivity values here were more than 10 µS/cm higher than the upstream wetlands and by 2006 conductivity was at least 18 µS/cm higher. Impacts from SWD1 should be investigated in more detail, especially considering that BBC 44 is a highly sensitive bog. Bogs, and sphagnum-dominated peatlands in particular, are very rare in Western Washington and these ecosystems support unique plant and invertebrate communities (Kulzer et al. 2001). Once bog chemistry is disrupted, it is very difficult to restore and development of restoration techniques are in their infancy with little progress until recently (i.e., beginning in the 1990s) (Quinty and Rochefort 2003, Rochefort et al. 2003).

To compare changes in conductivity to changes in development, we estimated the proportion of each UPD basin that had been built for each year assuming that build-out occurred two years following clearing. These estimates were created from development phasing maps (Kpff Consulting Engineers 2004, Otak Incorporated 2006) and do not take into account development outside the UPD boundaries or the proportion of land area remaining in natural resource buffers. Instead, this estimate provides an approximation of how much UPD development has occurred relative to the total UPD development proposed within each basin. This proxy for development was highly correlated with conductivity at the majority of our water quality sampling locations (Figure 4.3.1.3). Despite only having four to five years of data at most locations, five sample locations have significant correlations. EC 4 was the only location with decreasing conductivity with increasing UPD development. However, as mentioned before, conductivity was highest in 2003 and 2004 at this location when construction was taking place. Due to our assumption of a two year lag between clearing and build-out, these two years were considered to have 0% development. Construction impacts may have temporarily increased conductivity during this time.

These data suggest that conductivity has increased at numerous wetlands and streams associated with the UPDs. Conductivity did not increase within basins with little to no UPD development.

However, without a better understanding of the natural variability of conductivity and sampling consistency between researchers, it is difficult to conclusively link the apparent increases with UPD development.

pH

The 20 monitored streams and wetlands all have acidic conditions with pH ranging from 4.4 to 5.7 (median = 4.7) in 2002, the first year of routine monitoring. These values are lower than the regional median of 6.4 (max = 7.7) for 162 non-urban wetlands (Horner et al. 2001a) and below the 6.5 to 8.5 state water quality standards for freshwater (WAC Chapter 173-201A 1997). However, these values are comparable to pH levels found in *Sphagnum*-dominated peatlands that are naturally acidic (Kulzer et al. 2001).

Changes in stormwater pH following development were not predicted to impact stream and wetland water quality due to treatment within the stormwater ponds and release via infiltration or spreader devices (Beak Consultants Incorporated 1995). However, overall pH has increased in UPD wetlands and streams from an average of 4.8 in 2002 (SD = 0.28, n = 51) to 5.2 in 2006 (SD = 0.48, n = 60) (Figure 4.3.1.4). The mean pH in 2006 is significantly different than the mean pH of all other years ($p<0.001$, ANOVA). The increase of pH has been greater in the streams than in the on-site wetlands. The average pH of the streams in 2002 was 4.9 (SD = 0.24, n = 16) with a peak average of 5.6 in 2006 (SD = .31, n = 21), whereas wetland pH rose from a mean of 4.8 in 2002 (SD = 0.28, n = 35) to 5.1 (SD = 0.46, n = 39) in 2006 (Figure 4.3.1.4).

These increases are also apparent on a site specific basis over time (Figure 4.3.1.5). Most sites had peak pH values in 2006. Wetland SR 24c and the two Unnamed Creek sites (53B and 53C) did have slight decreases in pH in 2006, down from peak 2005 levels. Wetlands EC 61 and EC 4 did not have increased pH in 2006, but these locations have had more variability than most since 2002 and no development has taken place within the basin of EC 61. As mentioned previously, almost no development has taken place within the BBC 52 and BBC 45 basins with clearing only beginning in 2005. However, the three sampling sites on these two wetlands also show small increases in pH in 2006 compared to previous years.

Like conductivity, increases in pH were repeatedly correlated with increases in development within each UPD basin (Figure 4.3.1.6), with five sample locations having significant correlations.

Changes in pH should be monitored closely over time to determine if increases observed in 2006 continue. The plants and animals found in these systems are adapted to the naturally acidic conditions and changes in pH will likely alter the fragile chemical balance that shapes the entire ecosystem. However, without a better understanding of the natural and field variability of pH, it is difficult to determine whether the apparent changes are associated with increased development within the basins. Furthermore, while 2006 was the first year demonstrating statistically significant increases in pH, increases were observed both in basins with and without UPD development. Therefore, changes in climate or precipitation regimes could be influencing pH throughout the region.

In addition, it is difficult to get accurate readings at such low pH levels. We have relied on litmus paper because we have been unable to get consistent readings using meters which frequently do not stabilize on a value. However, litmus paper provides a very coarse measurement of pH and relies on the observer to allow the paper to fully change color, a subjective end point, before making a reading. In 2006 and for all subsequent years, litmus paper was placed in the water on arrival and the pH was read just before leaving each location after all other sampling was finished in order to maximize the amount of time for the paper to completely change color. Collecting samples for lab processing of pH should be considered if more accurate readings are deemed necessary.

Metals

Metals are generally more soluble, and therefore more bioavailable to harm organisms at lower pH, alkalinity, and hardness (Horner et al. 1994). Most of the wetlands sampled are naturally low pH and alkalinity systems and since the streams we sample drain these wetlands, they generally have low pH too. Therefore, these systems are particularly vulnerable to relatively small amounts of metal inputs because there is little bicarbonate and carbonate for the metals to bind to leaving the metals primarily in their toxic ionic state (Horner et al. 1994). Close monitoring of metal levels and potential sources in these systems is essential.

Baseline measurements of dissolved metal concentrations from the early 1990s, when corrected for hardness, repeatedly exceeded Washington State water quality standards (WAC Chapter 173-201A 2003) for lead, zinc, and copper at a number of sample locations (Appendix 4.3.1.2). Data from the Puget Sound Wetlands and Stormwater Management Program (PSWSMP) from the late 1980s through the mid 1990s suggest that high levels of soil metals from 1988 to 1990 were not uncommon throughout the Puget Sound region (Reinelt and Horner 1990, Horner et al. 2001a). By the mid 1990s, they observed substantial declines associated with reductions in industrial air pollution point sources and automobile pollutants including the removal of lead as a fuel additive and the advent of emission controls and catalytic converters (Horner et al. 2001a).

In 2006, total metal concentrations were measured for each sample location. The (U.S. Environmental Protection Agency 1996)conservatively recommends converting total metals to dissolved concentrations by multiplying them by 0.97. We observed few metals exceedances in 2006 relative to the baseline data, however given the lower concentrations regionally as compared to the 1980s and 1990s, any level above the state standards should be investigated to determine the source. Acute water quality standards were exceeded for zinc at BBC 45 upstream and gauge 53C on Unnamed North and chronic zinc standards were violated BBC 44 upstream (Table 4.3.1.2). At wetland SR 24b, copper was also in violation of acute water quality criteria. These locations should be monitored closely in future years; however, at wetlands BBC 44, BBC 45 upstream, and SR 24b, the violations were only for one of the three sample periods in 2006 and the water quality returned to acceptable levels by the final sample period in each case.

In contrast, zinc concentrations at the stream gauge 53C on Unnamed North exceeded acute state standards for each of the three sample periods. Only 15 stream samples out of 1,995 collected around King County exceeded acute zinc standards (Figure 4.3.1.7). These samples averaged hardness of only 18.5 - $mgCaCO^3/L$ (SD = 13.2, n = 15). Due to concerns regarding the steep, highly erosive hill that Unnamed Creek flows across before draining into the Snoqualmie Valley, very little water from the developments on the plateau drain directly into the creek. Instead, the drainage is directed into several stormwater ponds, which overflow into a bypass system. As a result, it is difficult to attribute these exceedances to the extensive clearing and development that has been taking place on parcel O since 2004.

To try to identify potential sources for the repeated zinc violations, two samples were collected from Unnamed North in September 2006 at the normal 53C location and just upstream of the gauge. KCEL analyzed the samples for both dissolved and total levels of metals. This time samples did not exceed state standards, but water levels were extremely low. Zinc is an abundant trace mineral that occurs naturally, but higher than normal levels are associated with roadway runoff (Beak Consultants Incorporated 1995). However, the source of the three violations at Unnamed North has not been pinpointed, and additional tests should be considered for 2007 to determine if a bottom-less galvanized culvert may be contributing to toxic levels of zinc (Figure 4.3.1.8). In September 2006, there was no surface water running through the culvert, therefore it was not possible to sample at locations up- and downstream of this potential source.

Temperature, Turbidity, and Dissolved Oxygen

Discrete, point measurements of dissolved oxygen, temperature, and turbidity could be helpful in identifying possible causes of specific biological anomalies (e.g., high amphibian or fish mortalities) (Appendix 4.3.1.3). However, without continuous data, more samples to account for temporal and measurement variation, or insurances to standardize the time of day and position in the water column that sampling occurs, it is difficult to accurately assess trends for these parameters over time for a particular site (e.g., dissolved oxygen concentrations or temperature may be low first thing in the morning, but much higher that same afternoon. Therefore, for the purposes of this report, we will focus on trends across all streams and wetlands rather than for site-specific data for dissolved oxygen, temperature, and turbidity.

Adverse temperature impacts were not predicted to streams or wetlands because the UPDs do not directly discharge to temperature sensitive streams and stormwater runoff that flows into stormwater ponds is returned to wetlands via interflow or infiltration into groundwater aquifers (Beak Consultants Incorporated 1995). In addition, most intense storms that would generate minor surface flows primarily occur under low temperature conditions.

Temperatures for all streams and wetlands showed very little change across years, hovering around an average of 8 ºC each year (Figure 4.3.1.9), well below state standards of 16 and 18 ºC for class AA and class A streams (WAC Chapter 173-201A 1997). 2003 was significantly different ($p<0.05$) from 2004 and 2005 temperature values according to the non-parametric Kruskal-Wallis test and Mann-Whitney pair-wise comparisons, however there were no increasing or decreasing trends across years.

Baseline turbidity were very low (mean 2.0 nephelometric turbidity units (NTU) for Redmond Ridge) (Herrera Environmental Consultants Inc. 1992) and no adverse turbidity impacts were predicted from UPD development due to flat topography and mitigation methods such as stormwater pond detention, discharge via level spreaders and interflow, and groundwater infiltration (Beak Consultants Incorporated 1995). 2004 and 2005 turbidity were significantly different ($p<0.05$) from all other years (Figure 4.3.1.10), however there was no increasing or decreasing trend across years as turbidity fluctuated between mean levels of 1.6 to over 4 NTU, below the state standard of 5 NTU for class AA and class A waters. With the exception of 2005, wetlands had lower turbidity than streams as expected since the lack of flow allows suspended sediment to settle out of the water column.

One of the potential secondary construction impacts identified in the EIS process was lowered dissolved oxygen concentrations in the wetlands related to nutrient inputs through sediment transport (Beak Consultants Incorporated 1995). However, these impacts were predicted to be effectively mitigated by wetland buffers, a lack of definitive surface channels, flat topography, and a thick forest duff layer.

Dissolved oxygen levels increased from 5.9 - mg/L in 2002 to 8.9 - mg/L in 2006 (Figure 4.3.1.11). Oxygen levels in 2005 and 2006 were significantly different ($p<0.05$) from all other years except between 2002 and 2005. As expected due to lentic conditions and increased decomposition, we observed lower oxygen in wetlands (mean 5.1 - mg/L, minimum 0.24 - mg/L) than in streams (mean 9.0 - mg/L, minimum 5.0 - mg/L).

The sampling locations on Adair and the two Unnamed Creek tributaries within the Snoqualmie River drainage, met class A water quality standards with dissolved oxygen exceeding 8.0 - mg/L 93% of the time. All four violations were on Unnamed Creek West at gauging station 53B. The sampling locations within the Bear and Evans Creek basins met class AA water quality standards above 9.5 - mg/L only 39% of the time. Colin South at gauging station 02D had three violations out of 12 sampling periods all with over 8.4 - mg/L dissolved oxygen. Rutherford at gauging station 18F had nine violations ranging from 6.3 to 9.3 - mg/L and Evans Creek Middle at gauging station 18B had eleven violations ranging from 5.0 to 9.2 - mg/L. Both the Rutherford and Evans Middle sites are immediately downstream of wetlands EC 3 and EC 5, respectively. Therefore, there is little opportunity for the slow moving wetland waters to become aerated as they transition to flowing waters and their dissolved oxygen levels are more representative of wetland oxygen levels.

Typically, decreases in dissolved oxygen concentrations are associated with water quality degradation in streams. However, in bogs increases in dissolved oxygen could be an early indicator of ecosystem changes where increases in the naturally low pH and nutrient levels could support conditions (e.g., decreased tannic acids) favoring increased photosynthesis that results in higher oxygen concentrations. To explore this hypothesis, we graphed the oxygen levels for each of the different sampling periods separately (Figure 4.3.1.12). Increases in oxygen concentrations were observed regardless of time of year. Changes in photosynthesis would not be expected during the first two sample periods in late January to early February and late February to early March when low levels of photosynthesis are expected in response to low winter light conditions.

Sources of measurement error include oxygen calibration methods and how vigorously the probe was stirred to ensure water flow across the measurement membrane. The lack of consistency in field personnel between years amplifies these sources of error. As a result, it is difficult to irrefutably link increases in dissolved oxygen concentrations to UPD development.

Nevertheless, the trend indicates increases in dissolved oxygen, which could suggest ecosystem changes. Steps should be taken to ensure more consistent measurement methods and this parameter should be monitored closely in future years. Changes in the sampling protocol to encompass quarterly sampling and field repetitions may be warranted to better determine the significance and source of these changes.

Threshold Exceedance Criteria

Wetland water quality threshold exceedance criteria (King County 1999, 2001) are violated if:

1. There are negative deviations from baseline parameters and conditions significant to wetland function.
2. There are extreme fluctuations in measurements signaling system instability or unusual inputs.
3. Alkalinity differs from pre-development alkalinity.

Stream water quality threshold exceedance criteria (King County 1999, King County 2001) are violated if:

1. Water quality degradation is observed including changes from baseline pH, temperature, turbidity, or other conditions.
2. State and federal water quality standards are violated.

These threshold exceedance criteria are not highly quantitative and therefore some level of interpretation is required. The apparent trend of increasing conductivity, dissolved oxygen, and pH at the majority of wetland and stream sites which have experienced UPD development does suggest water quality degradation or changing conditions. In addition, there were repeated violations of zinc levels at Unnamed North. State standards for human-caused variation in pH (0.2 or 0.5 units for class AA and class A waters, respectively) and for dissolved oxygen levels were repeatedly violated.

Corrective Actions

Due to the sampling design, we lack a detailed understanding of the natural variability in conductivity and pH between years. The lack of consistent field personnel introduces additional sources of variability. Nevertheless, the apparent increases in conductivity, dissolved oxygen, and pH across wetland and stream locations warrants an assessment of stormwater pond functioning to determine whether any additional BMPs can be implemented to further mitigate the impacts of UPD development. If future testing for metals implicates the culvert under NE Vine Maple Way as contributing to elevated levels of zinc in Unnamed Creek, the galvanized culvert should be replaced with a plastic culvert.

Discharge of water from stormwater ponds through level spreaders is an accepted construction BMP (Washington State Department of Ecology 2001, 2005) to help reduce increased turbidity associated with runoff. This is a construction-phase activity beyond the scope of post-development monitoring report. However, care should be taken to ensure that water is pumped at a rate and volume that allows complete infiltration and prevents a surface connection as was observed during spring 2006 (see Appendix 4.1.2.1). In addition, to ensure surface water is discharged rather than sediment-laden bottom water, the pumping should be done with a float attached to the hose.

Table 4.3.1.1 Metals levels exceeding state water quality standards for 2006.
Standards change based on hardness values (WAC Chapter 173-201A 2003). Results from the lab were only for total metals. To convert to estimated dissolved metal concentration, total metal concentration was multiplied by 0.97 (U.S. Environmental Protection Agency 1996). 53C is on Unnamed North.

Location	Date	Hardness (mg $CaCO_3$/L)	Parameter	Total Metal (µg/L)	Dissolved Metal (µg/L)
53 C	1-25-26	11.4	Zinc	0.0259	0.0251
53 C	2-28-06	12.5	Zinc	0.0320	0.0310
53 C	4-20-06	13.7	Zinc	0.0325	0.0315
BBC 44 u/s	1-25-06	4.7	Zinc	0.0083	0.0081
BBC 45 u/s	2-28-06	4.3	Zinc	0.0110	0.0107
SR 24b	2-28-06	16.6	Copper	0.0456	0.0442

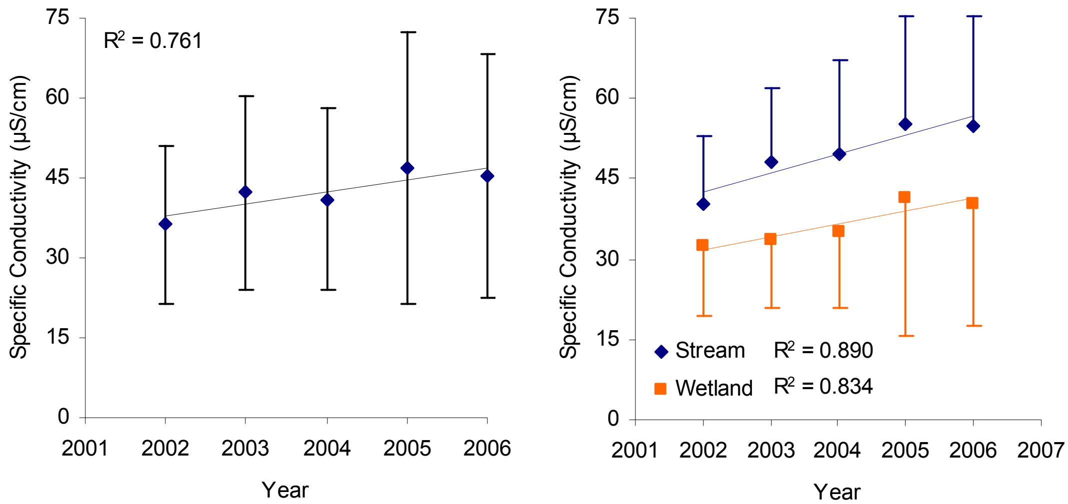


Figure 4.3.1.1 Average conductivity among all UPD sample sites (n=20) from 2002 to 2006 (left), and average conductivity of streams (n=7) and wetlands (n=13) (right).
Error bars represent one standard deviation. Mean conductivity was not significantly different (p=0.08) between years according to the Kruskal-Wallis non-parametric test.

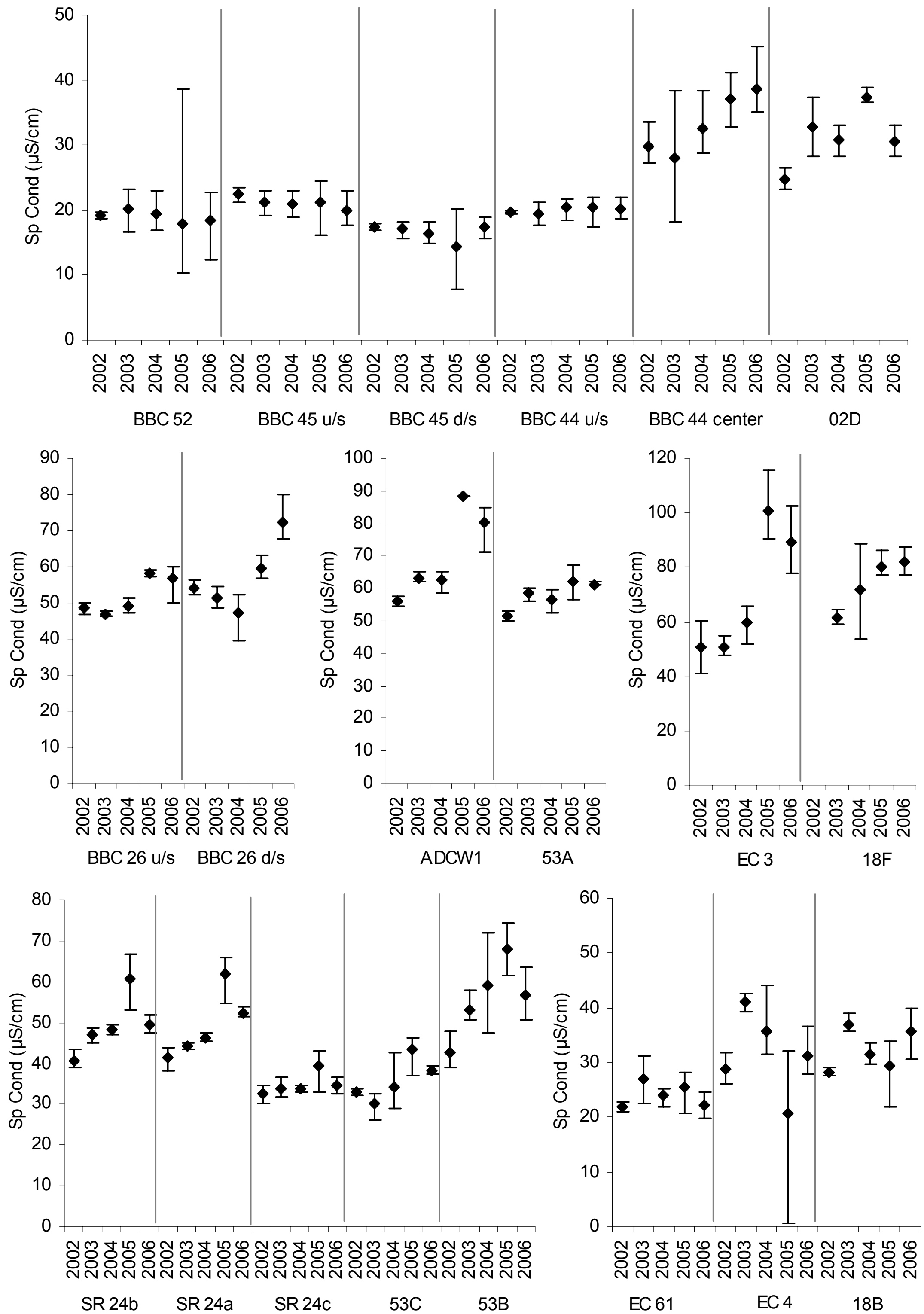


Figure 4.3.1.2. Basin by basin comparison of mean specific conductivity (Sp Cond) values from 2002 to 2006.

Each graph is organized approximately upstream to downstream (e.g., BBC 52 is the upstream most site on a string of wetlands ending at Colin Creek South and gauge 02D (top graph)). Error bars extend to the maximum and minimum values of 3 annual samples.

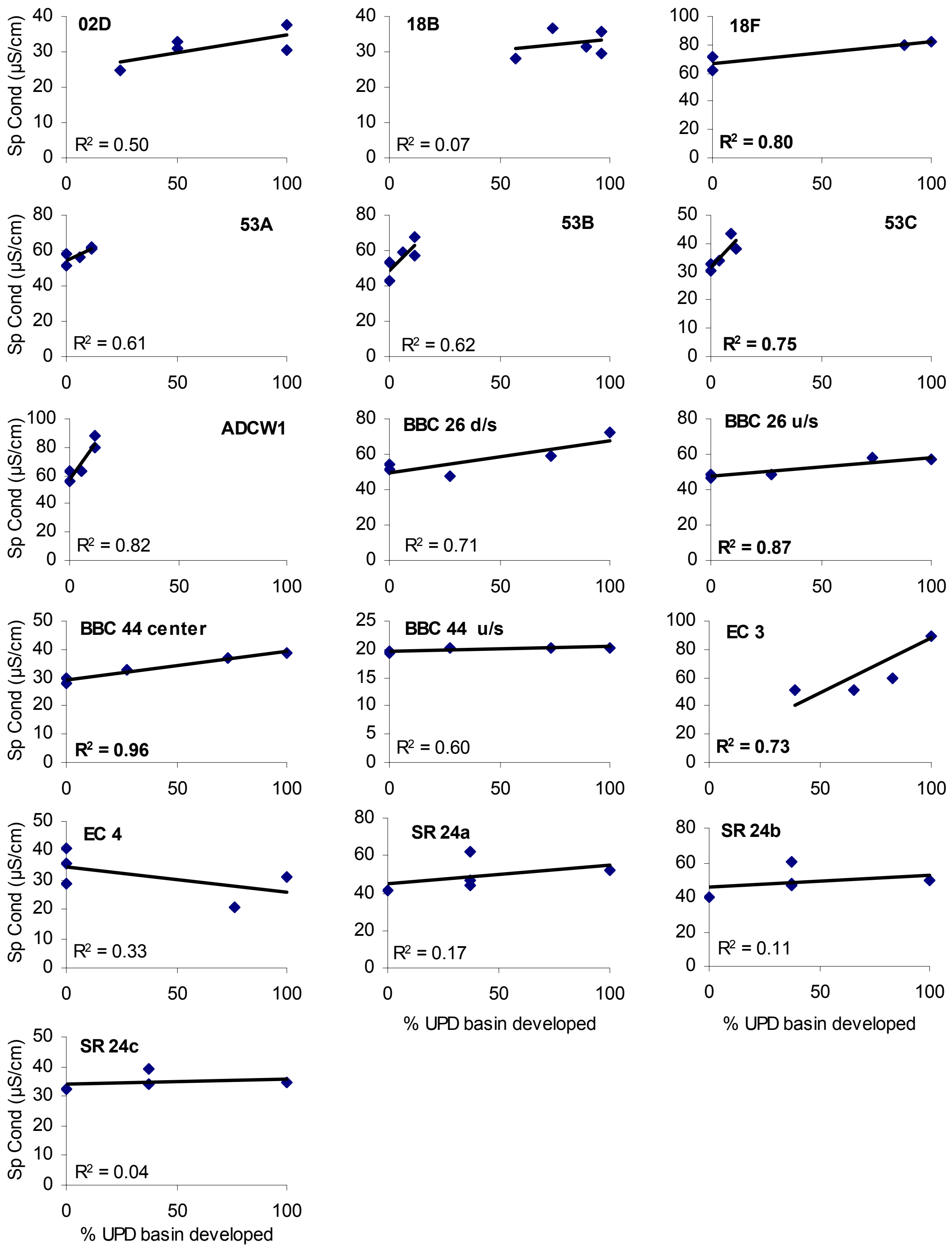


Figure 4.3.1.3 Correlations between basin development and specific conductivity (Sp Cond). Significant correlations ($p < 0.05$) report the R^2 value in bold. BBC 52, BBC 45, and EC 61 locations are not shown because no development beyond some initial clearing has taken place in these basins through 2006. Basin development was estimated from development phasing maps (Kpff Consulting Engineers 2004, Otak Incorporated 2006). Conductivity was averaged for three sample dates between January and April each year generally from 2002 to 2006.

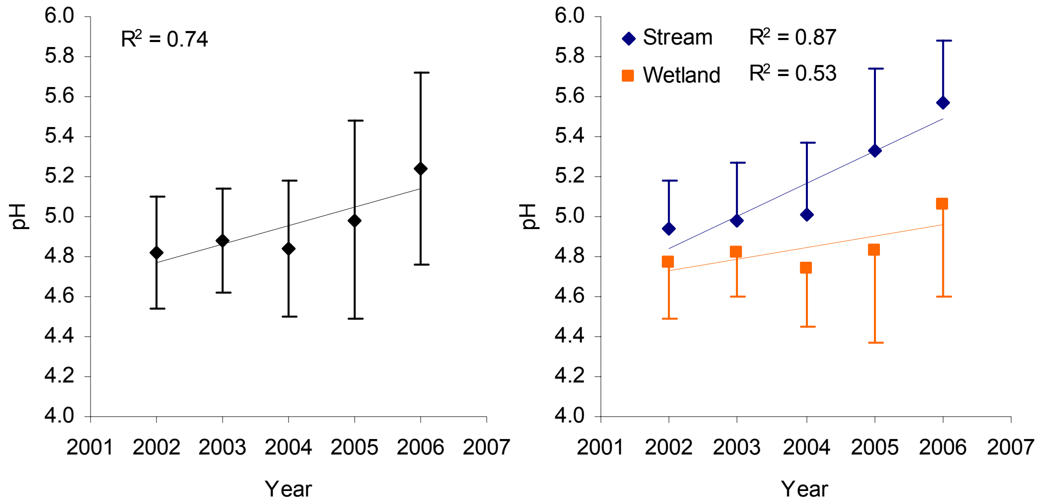


Figure 4.3.1.4 Average pH among all UPD wetland and stream sites (n=20) from 2002 to 2006 (left) and average pH of streams (n=7) and wetlands (n=13) (right).
Error bars represent one standard deviation. For streams and wetlands combined, 2006 pH was significantly different than all other years ($p<0.001$) as determined by an ANOVA test with Bonferroni pair-wise comparisons.

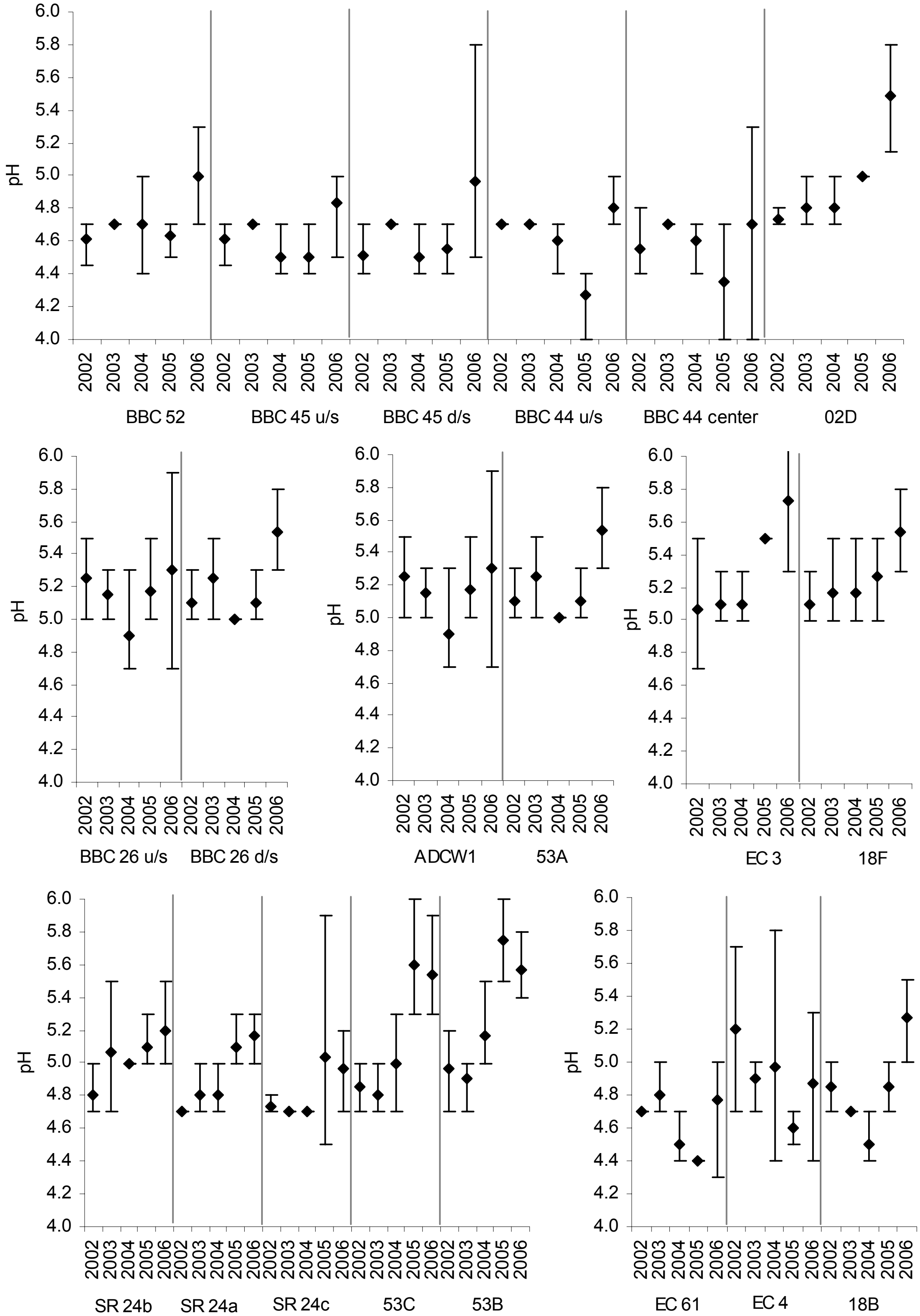


Figure 4.3.1.5 Basin by basin comparison of mean pH from 2002 to 2006.
Each graph is organized approximately upstream to downstream (e.g., BBC 52 is the upstream most site on a string of wetlands ending at Colin Creek South and gauge 02D (top graph)). Error bars extend to the maximum and minimum values of 3 annual samples.

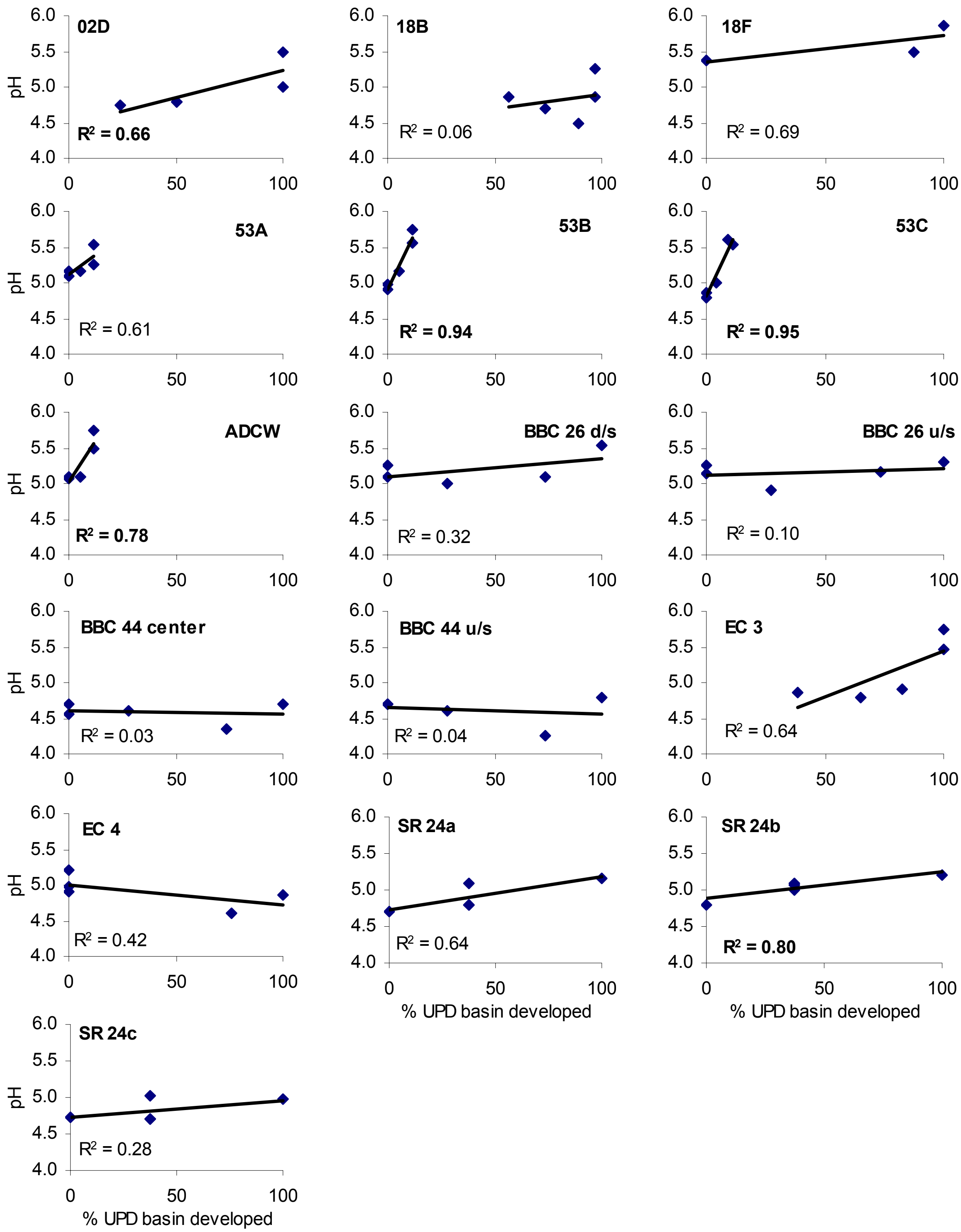


Figure 4.3.1.6 Correlations between basin development and pH.
Significant correlations ($p < 0.05$) report the R^2 value in bold. BBC 52, BBC 45, and EC 61 locations are not shown because no development beyond some initial clearing has taken place in these basins through 2006. Basin development was estimated from development phasing maps (Kpff Consulting Engineers 2004, Otak Incorporated 2006). pH was averaged for three sample dates between January and April each year generally from 2002 to 2006.

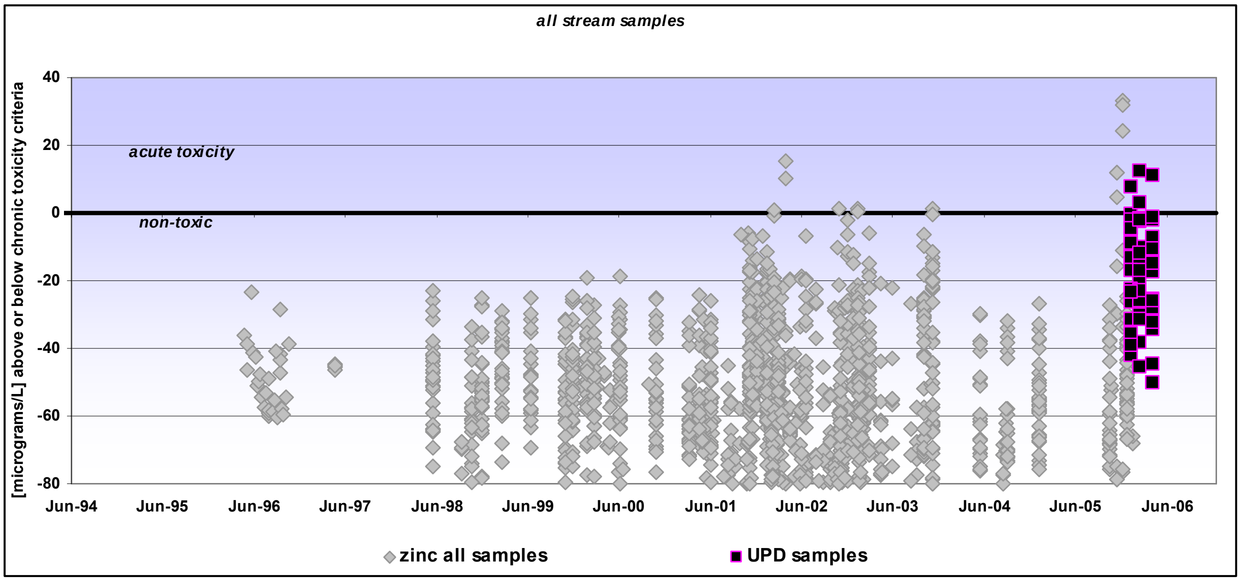


Figure 4.3.1.7 UPD zinc toxicity relative to county-wide stream data.

Figure 4.3.1.8 Galvanized culvert at the upstream extent of Unnamed North under Vine Maple Way. Looking through the culvert facing downstream (left) and the downstream outlet of the culvert (right).

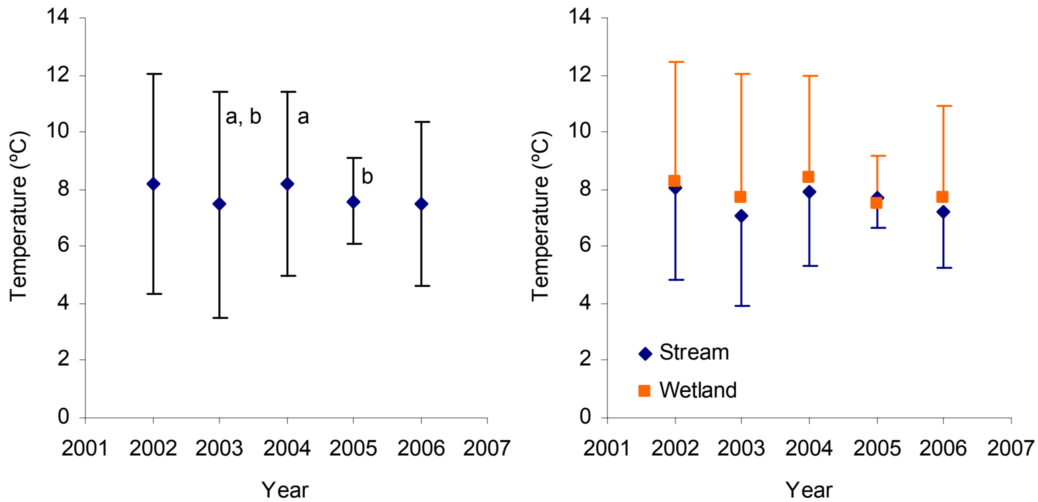


Figure 4.3.1.9 Average temperature among all UPD wetland and stream sites (n=20) from 2002 to 2006 (left) and average temperature of streams (n=7) and wetlands (n=13) (right).
Error bars represent one standard deviation. Letters represent years that were significantly different ($p<0.05$) as determined by a Kruskal-Wallis non-parametric test followed by Mann-Whitney pair-wise comparisons.

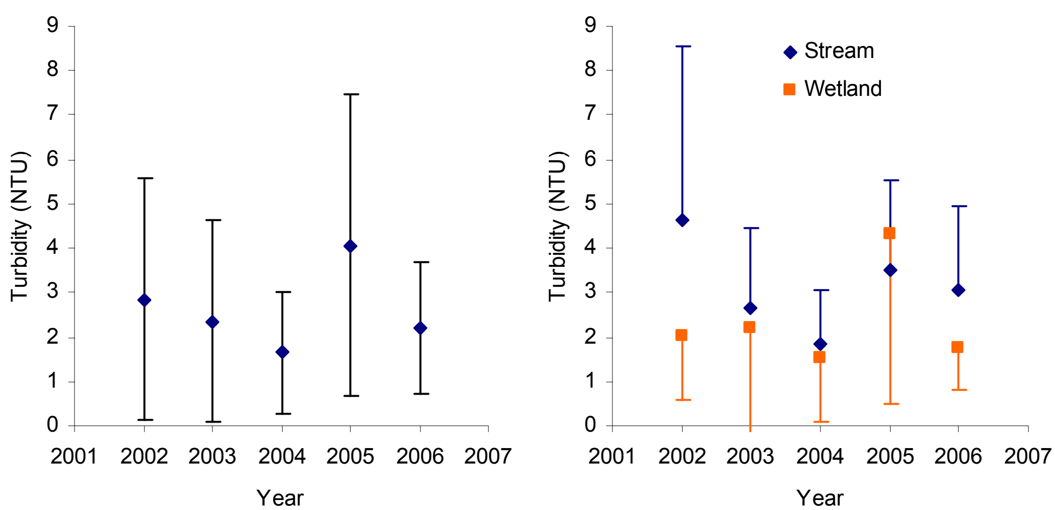


Figure 4.3.1.10 Average turbidity among all UPD wetland and stream sites (n=20) from 2002 to 2006 (left) and average turbidity of streams (n=7) and wetlands (n=13) (right).
Error bars represent one standard deviation. 2004 and 2005 turbidity were significantly different ($p<0.05$) than all other years as determined by a Kruskal-Wallis non-parametric test followed by Mann-Whitney pair-wise comparisons.

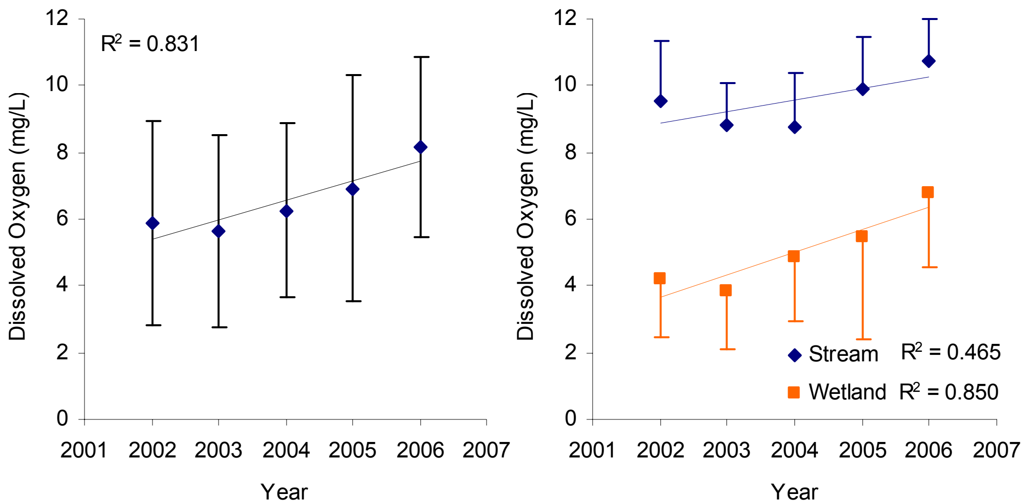


Figure 4.3.1.11 Average dissolved oxygen concentrations among all UPD wetland and stream sites (n=20) from 2002 to 2006 (left) and average dissolved oxygen of streams (n=7) and wetlands (n=13) (right).
Error bars represent one standard deviation. 2005 and 2006 dissolved oxygen levels were significantly different ($p<0.05$) than all other years except 2002 and 2005 as determined by a Kruskal-Wallis non-parametric test followed by Mann-Whitney pair-wise comparisons.

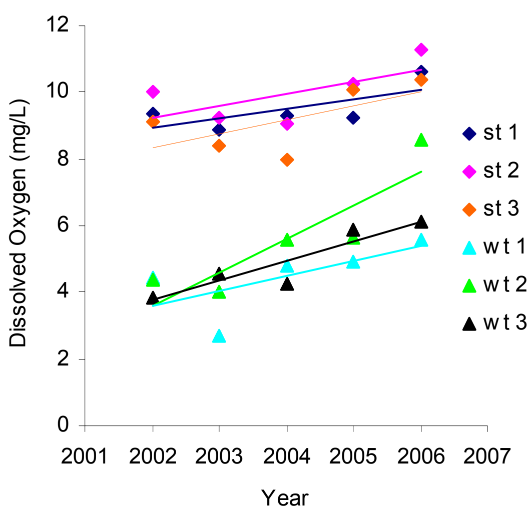


Figure 4.3.1.12 Dissolved oxygen concentrations at wetlands (wt) and streams (st) during three sampling periods.
Sampling periods were: “1” – late January to early February, “2” – late February to early March, and “3” – mid to late April.