Appendix A.

Key non-regulatory habitat protection programs and selected attributes

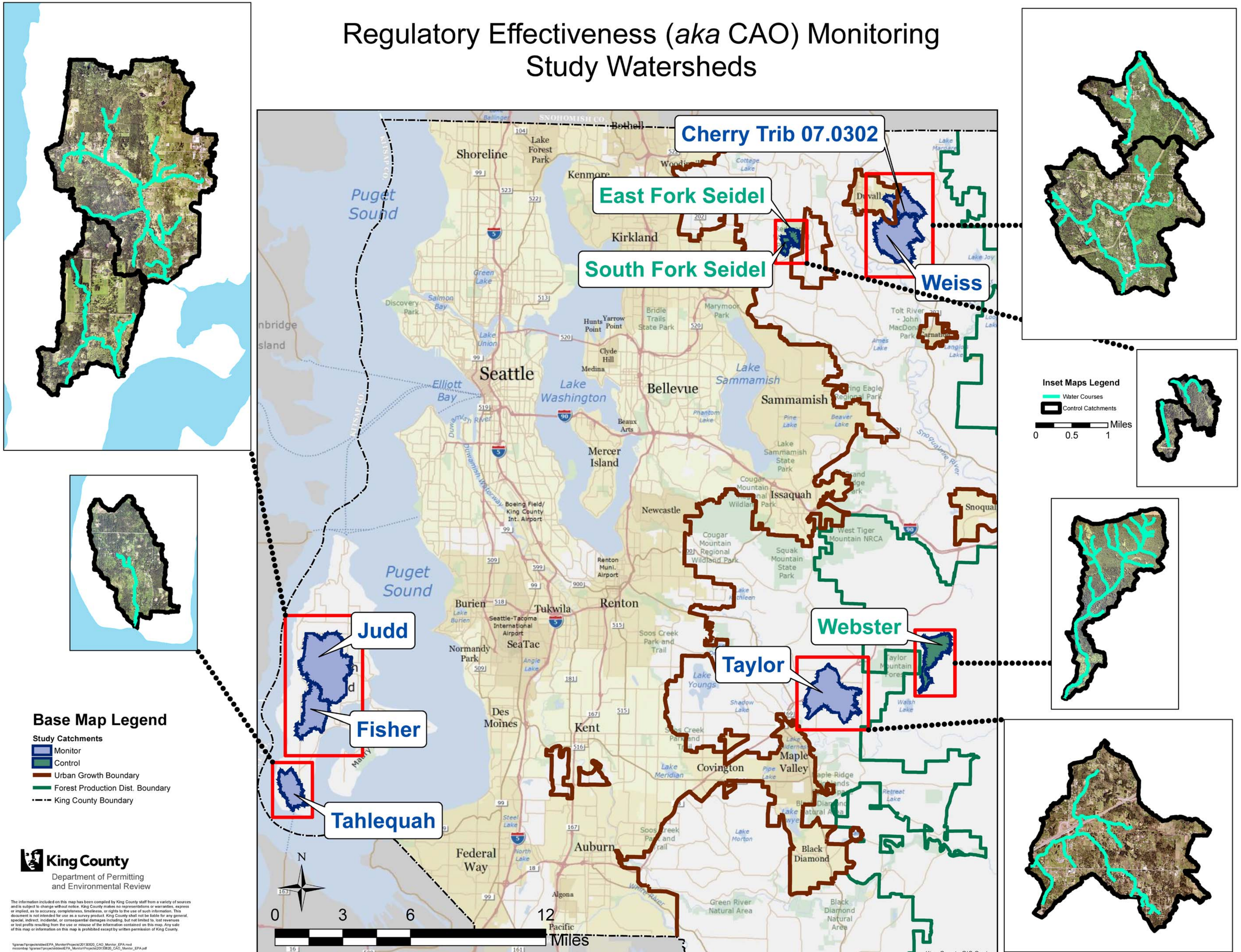
Key non-regulatory habitat protection programs and selected attributes (from King County 2012a).

Program	Focal Areas	Purpose	Permanent protection	Area (ha) in King County (2011)*	Notes
Current Use Taxation (CUT) - Forestland	Forested, > 20 acres	Tax incentive to preserve land in forestry.	No. Can be reversed with payment of back taxes and penalties.	102,400	Majority also in TDR and Open Space Charter Amendment Programs
Current Use Taxation (CUT) - Agriculture	Agricultural lands	Tax incentive to preserve land in agriculture.	No. Can be reversed with payment of back taxes and penalties.	11,782	Majority also in Farmland Preservation Program
Timberland	Forested, 5 to 20 acres. Small-lot forests on land zoned Rural Residential	Tax incentive to preserve land in forestry.	No. Can be reversed with payment of back taxes and penalties.	1,404	
Public Benefit Rating System	Primarily areas zoned as Rural Residential	Points-based tax incentives to preserve and manage land for variety of benefits, such as stream buffers, ground water protection areas, threatened or endangered wildlife, public recreation and historic property.	No. Can be reversed with payment of back taxes and penalties.	4,044	
Transfer of Development Rights (TDR)	Primarily Forest Production District and areas zoned residential in Rural Forest Focus Areas	Prevent future development or change in land use.	Yes. Development rights permanently removed.	57,277	Majority also in CUT Forestland and Open Space Charter Amendment Programs
Farmland Preservation Program	Agricultural lands	Preserve agricultural lands. Prevent change in land use.	Yes. Development rights permanently removed.	5,302	Majority also in CUT Agriculture program
Open Space Charter Amendment	County-owned lands and private forest lands that have permanent conservation easements	Strengthen protection of designated open space lands by requiring supermajority of County Council to authorize any future changes in use.	Nearly, subject to super-majority vote of County Council.	62,481	Majority also in CUT Forestland and TDR Programs

*These designations are not all exclusive and therefore, some of these areas may overlap.

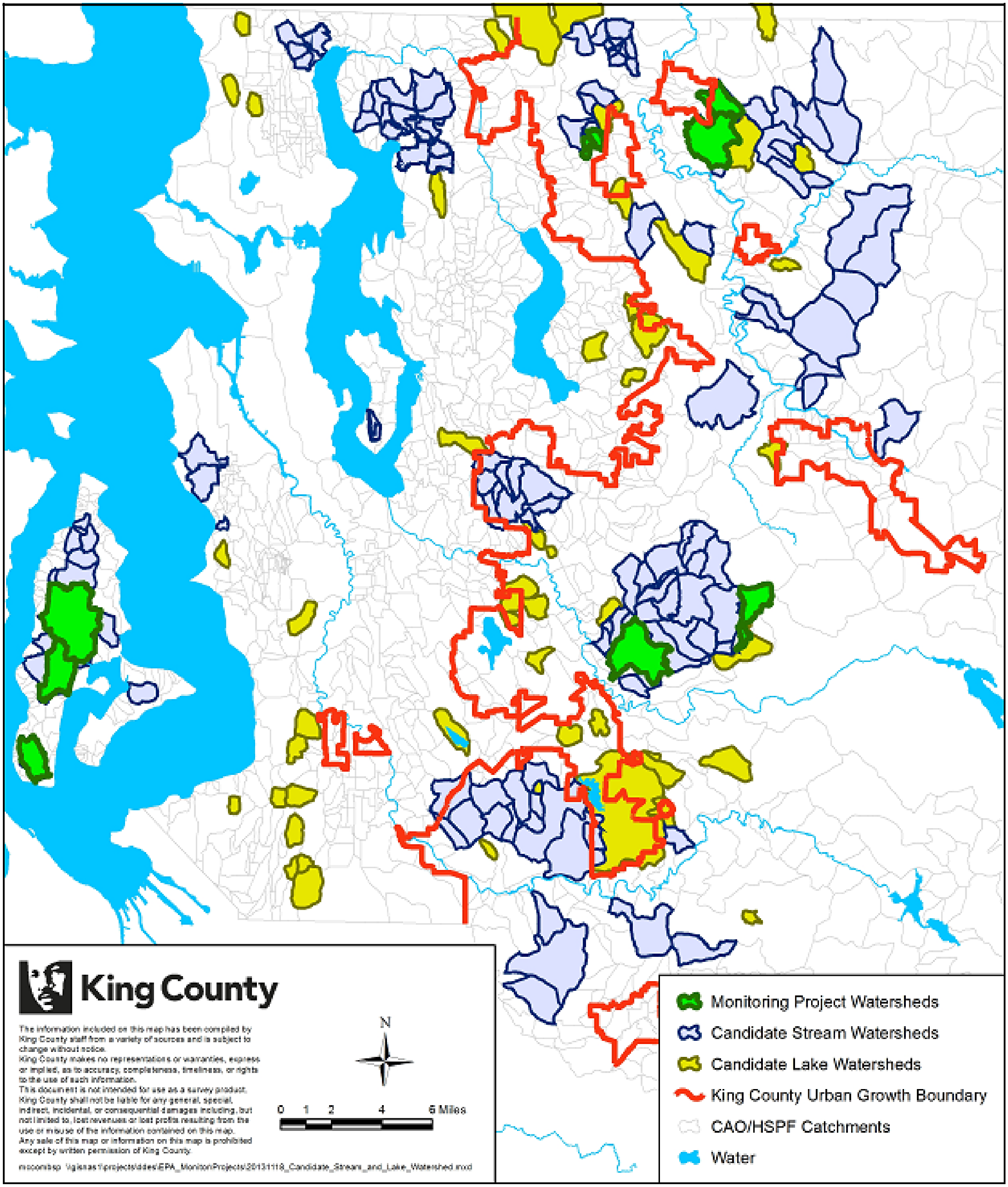
Appendix B.
Map of Study Watersheds

Regulatory Effectiveness (*aka* CAO) Monitoring Study Watersheds



Appendix C.

Watersheds Considered for Regualtory Effectiveness Monitoring


King County
The information included on this map has been compiled by King County staff from a variety of sources and is subject to change without notice.
King County makes no representations or warranties, express or implied, as to accuracy, completeness, timeliness, or rights to the use of such information.
This document is not intended for use as a survey product.
King County shall not be liable for any general, special, indirect, incidental, or consequential damages including, but not limited to, lost revenues or lost profits resulting from the use or misuse of the information contained on this map.
Any sale of this map or information on this map is prohibited except by written permission of King County.
N
0 1 2 4 6 Miles
Monitoring Project Watersheds
Candidate Stream Watersheds
Candidate Lake Watersheds
King County Urban Growth Boundary
CAO/HSPF Catchments
Water

Appendix D.

Michalak et al. 2013

Implications of land-cover change history for monitoring present and future ecological condition in nine basins on the urban fringe of Seattle, Washington

Julia Michalak[1], Gino Lucchetti[2], Josh Latterell[2], and Ray Timm[2]

Abstract:

Historic land use is a potentially significant factor determining present day watershed condition. Previous research has shown that historical land uses can have lasting effects on watershed condition that are manifest in present day hydrologic and water quality variables. However, the relative importance of past land uses such as the extent and intensity of forestry or agriculture within a basin, remains poorly understood. As part of a larger project to monitor the effectiveness of King County's Critical Areas Ordinance, we reconstructed land-cover conditions over approximately 100 years in nine small watersheds (80 – 1200 ha) in the Puget Sound region. We used these data to explore three questions: 1) how has forest cover changed overall within these watersheds; 2) how has forest cover changed within the riparian zone since 1936; and; 3) what land-cover changes are common to all watersheds and when and how do land-cover histories diverge?

We found that overall, the watersheds share a common history of forest-cover change, though the timing and extent of the change varied by watershed. The watersheds were primarily forested prior to 1900. All the watersheds lost between 50 and 100% of their forest cover between 1900 and 1948 and existing cleared lands either transitioned back to forest or were converted to agriculture during this time. Between 1936 and 1948, all the watersheds except Taylor and East Seidel retained 40% to 60% of their riparian forest cover even though the watersheds overall were only 20 to 40% forested. The retention of forested buffers within the riparian zone potentially reduced the impacts of early logging on overall watershed condition and contemporary water quality. Between 1948 and 1965, forest cover increased substantially in all watersheds, and by 1986, all watersheds were 60-100% forested. Since 1986, forest cover has declined slowly primarily due to conversion to rural residential land cover. In Cherry, Weiss and Tahlequah watersheds, development occurred primarily on previously forested lands without intervening agricultural land use. Comparatively, in Fisher, Judd and Taylor watersheds much of the early residential development occurred on previously agricultural lands. In addition, these latter watersheds developed to a greater extent than the other three treatment watersheds.

This analysis reveals potentially important variation in land-cover history among watersheds, which today have very similar land-cover characteristics. Identifying commonalities and variation in land use history is potentially critical for understanding both present day conditions and the overall trajectory of watershed change in the future.

Introduction

Numerous studies have demonstrated that watershed land use and land-cover composition correlates with water quality and ecological conditions in aquatic systems (Basnyat et al. 2001, and Pan et al. 2003, Groffman et al. 2004, Brett et al. 2005, Handler et al. 2006, Zampella et al. 1999). In particular, the amount and configuration of urban land cover within a watershed is significantly correlated with indicators of stream health (Walsh et al. 2005, Alberti et al. 2007).

[1] University of Washington Urban Ecology Research Lab, 432 Gould Hall, 3949 15th Ave NE, Seattle, WA.
[2] King County Department of Natural Resources and Parks, 201 S. Jackson St., Suite 600, Seattle, WA.

Understanding the relationship between urban land cover patterns and watershed condition is critical in order to protect and improve stream health in urbanizing regions. However, recent studies have found that the spatial complexities of watershed land use history can confound our understanding of the relationship between present day land cover and stream health (King 2005, Brown et al, 2009, and Harding et al. 1998). Watershed studies that include historical land use have found that prior land use and land cover conditions may have a lasting effect on stream ecology (Harding et al. 1998, Brown et al, 2009). As a result, historical land-cover conditions should be considered in stream monitoring studies. Ultimately, an improved understanding of the relationship between historical and present-day land cover and watershed condition is needed to adequately protect and manage urbanizing watersheds.

In 2005, King County (the County) updated its land use regulations as required by Washington State's Growth Management Act (GMA) to protect environmentally critical areas, including streams, lakes and wetlands. The regulations protect vegetative buffers surrounding critical areas, limit clearing and grading, and regulate the amount of stormwater runoff from developed surfaces (King County Code chapters 21A.24, Title 9, 16.82, and 21A.25). To assess the effectiveness of these regulations, the County initiated a multi-year (2008 to 2012) study of land-cover change and in stream conditions in nine sub-catchments (80 to 1,200 ha) located along the rapidly urbanizing fringe of Seattle. The study was conducted in six "treatment" catchments with relatively high future development potential indicated by parcels for which no known land-altering permits had been issued since 1989 and parcels whose value of improvements was ≤ 20 percent of the total parcel value based on the King County Assessor database. The resulting study watersheds selected to have relatively high similarity of land uses, regulations, and development potential over time and high sensitivity to development-based hydrologic alteration because of common geology, climate, drainage area, and gradients. The remaining three catchments were selected as controls as they are currently forested and protected from clearing and development. Over these five years, the County monitored flow flashiness, conductivity, temperature, benthic macro invertebrates, and channel hydraulic complexity as indicators of hydrology, water quality, biology and physical habitat quality, respectively. Simultaneously, the County quantified land use and land-cover change in each of the nine study watersheds. The County then compared in stream conditions and land-cover change in developing (treatment) and protected (control) catchments over time to identify any potential adverse effects of new development on watershed condition in these catchments. However, in order to truly isolate the effects of new development, it was necessary to understand historical conditions within these catchments that may still influence present day in stream dynamics.

Previous studies have found that land use history can have a lasting effect on in stream conditions (King 2005, Brown et al, 2009, and Harding et al. 1998). Specifically, watersheds with a history of logging may exhibit legacy effects such as a) reduced extent, size and diversity of riparian vegetation, b) little or no in-channel large wood and little or no new recruitment, c) reduced hydraulic complexity and quantity and quality of pools, d) channelization and bank hardening, and e) reduced soil perviousness from soil compaction, and f) altered flow paths roads and agricultural drainage ditches. In addition, the trajectory of land-cover change over time can have a significant influence on statistical relationships between present day urbanization and in stream condition. For example, Brown et al. (2009) found that the intensity of urban development in watersheds with an agricultural history had little to no effect on in stream conditions. In contrast, when forested watersheds are converted directly to urban development without an intervening period of agriculture, even small amounts of development had a measurable impact on stream macroinvertebrate communities.

Given the importance of historical land use in shaping present day in stream conditions, reconstructing the historical conditions within the nine study catchments is potentially central to understanding present day stream dynamics. Historical land cover within the study catchments could affect the outcomes of this study in several important ways. First, historical land use could influence the quality of the control catchments. These catchments are currently completely forested. However, present day in stream conditions may still be recovering from previous land uses such as logging or agriculture. Depending on the intensity and persistence of previous land use disturbances, the condition of reference streams may be more similar to treatment catchments than initially expected. Secondly, historical land use conditions may affect the starting conditions for the treatment watersheds, ultimately having an effect on the response of treatment catchments to new development. For example, if treatment watersheds experienced significant agricultural development prior to urbanizing, these watersheds may be starting from a degraded condition and show little to no change in response to new development.

In this study, we reconstructed land-cover conditions over a period of approximately 100 years in nine study catchments. The purpose of this study is to characterize the land use history of each catchment to establish the extent and trajectory of land-cover change in each catchment overall and the extent of forest loss within the riparian zone. Current theory suggests that protecting a buffer of vegetation adjacent to the stream channel can protect stream integrity despite forest loss in upland portions of a watershed (Sweeney 1993, Vuori and Joensuu 1996). Once this information is obtained, we can investigate the following questions:

1. Does land use history differ between the control and treatment catchments?
2. Does land use history differ among treatment catchments?
3. Did maintenance of forest riparian cover during periods of deforestation differ among catchments?
4. Do differences in land use history correlate with measures of present day watershed condition (i.e. the “starting conditions” of the nine catchments)?
5. Does the maintenance of riparian forest cover over time influence present day watershed condition?
6. To what extent have these streams recovered from previously degraded conditions?

Ultimately, understanding the historical land use conditions within these study catchments will contribute to the County’s ability to interpret observed starting conditions and change over time. Landscapes are constantly changing whether from management or natural processes–understanding historical land-cover helps us to understand trajectory of landscape conditions. Trajectory should affect the watershed’s resilience to further impacts or ability to ‘absorb’ development without significant changes in watershed health. A better understanding of the historical effects and legacies of land use change is necessary in order to understand the cause-and-effect relationships between development and stream health.

Methods

Study Sites

The selected study watersheds are located in the lowlands (predominantly < 500 feet in elevation) of the central eastern portion of Puget Sound. The study area covers the developing, low-lying western portion of King County, an area of common geologic history, flora, fauna, and human uses. Study watersheds contain small headwater alluvial streams originating on low-gradient upland plateaus, dropping across steep side-slopes to low-gradient base levels set by a major river, lake, or Puget Sound. Upland and riparian forests consist of second-growth conifers (mainly Douglas-fir, western hemlock, and western red cedar) and to a lesser extent deciduous trees (mainly big leaf and vine maple, red alder, and black cottonwood). Hydrology is rain-dominated, with naturally flashy flows during winter and low summer base flows. Aquatic productivity is typically limited by low nutrient availability, low summer flows, high winter flows, and, during winter, light.

Relatively small headwater watersheds (Strahler 1^{st} to small 3^{rd} order) with perennial, fish-bearing streams were chosen because prior studies demonstrate that they are sensitive to development-driven change in hydrology-mediated responses. A pool of candidate treatment watersheds was identified in an unbiased but nonrandom manner by screening for areas with relatively high future development potential indicated by parcels for which no known land-altering permits had been issued since 1989 and parcels whose value of improvements was ≤ 20 percent of the total parcel value based on the King County Assessor database. Subsequent selection of the study's six treatment watersheds was based on the following selection criteria:

- the presence of a past or existing flow, water quality or benthic invertebrate monitoring site;
- predominance of underlying glacial till-based geology, chosen because of its greater sensitivity to hydrologic change relative to glacial outwash, the other dominant surface geology in King County lowlands;
- absence of lakes, ponds and relatively large areas of wetlands, because these features may mask or reduce the magnitude of land-use driven hydrologic effects. Although effort was made to avoid wetlands it was not possible to select study watersheds with no wetlands as they are present throughout King County because of the relatively flat topography, moist climate, and prevalence of hydric soils in lowland Puget Sound; and
- lack or presence of only minor areas of urban zoning or areas under the regulatory control of other local jurisdictions to avoid confounding effects associated with the application of multiple land use regulations.

From the above criteria, the resulting study watersheds were assumed to have relatively high similarity of land uses, regulations, and development potential over time and high sensitivity to development-based hydrologic alteration because of common geology, climate, drainage area, and gradients. Reference watersheds were situated in municipal watersheds or nature reserves with no recent, existing or anticipated future development.

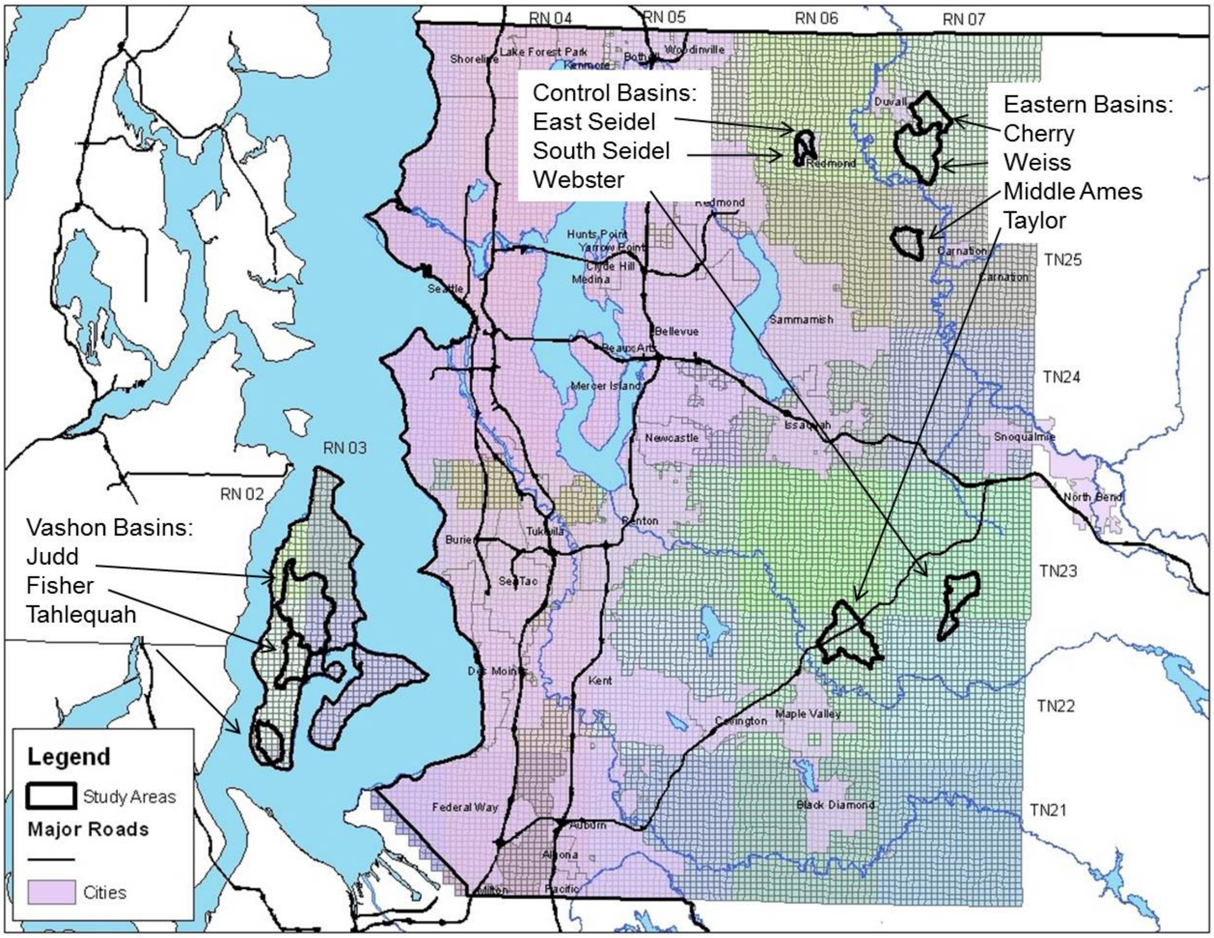


Figure 1 shows the location of the study catchments in King County. For presentation purposes, the catchments are categorized into three geographical and treatment groups: Vashon island catchments, eastern treatment catchments, and the three control catchments.

Data Collection

We used historical maps, aerial photographs, and classified Landsat TM land-cover data to reconstruct land-cover composition and configuration over six time periods within the nine watersheds. We searched the archives of the following organizations and agencies in order to identify all maps, records, and aerial photos that could provide information about land cover (forest cover, clearing, agriculture etc.) and land use (housing development, ranching, farming, road development etc.): the University of Washington map and special map collections (UW), the University of Washington Urban Ecology Research Lab (UW UERL), King County Department of Development and Environmental Services (KC DDES), King County Natural Resources Conservation Service (KC NRCS), King County Road Services Division (KC RSD), and the King County Archives.

We identified seven datasets that provided the most detailed and comparable land cover information and encompassed the majority of the study region (see Table 1). During the earlier time periods, not all datasets were available for all basins. It is also important to note that the scale and resolution of the datasets vary. In particular, the USGS Land Classification map was created at a very coarse scale, and thus may not show smaller patches of forest that may have been present.

Table 1 Final datasets included in the analysis. Not all datasets were available for all basins

Data set	GLO (1857-1892)	Timber Cruise (1907)	USGS Land Classification Map (1911)	Aerial photos - 1936	Aerial photos - 1948	Aerial photos - 1965	Land-cover data 1986-2007
Data Source	KC RSD, UW	King County Archives	UW	UW, KC RSD, KC DDES, KC Archives	KC NRCS	UW	UW UERL
Scale/ resolution	Twnship 10 miles: 1 in	40 acre tract	1:125,000	1:800	1:20,000	1:60,000	30 meter pixels
Cherry	Yes	Yes	-	-	Yes	Yes	Yes
East Seidel	Yes	Yes	-	Yes	Yes	Yes	Yes
Fisher	Yes	-	Yes	Yes	Yes	Yes	Yes
Judd	Yes	-	Yes	Yes	Yes	Yes	Yes
South Seidel	Yes	Yes	-	Yes	Yes	Yes	Yes
Tahlequah	Yes	-	Yes	Yes	Yes	Yes	Yes
Taylor	Yes	Yes	-	-	Yes	Yes	Yes
Webster	Yes	Yes	-	-	Yes	Yes	Yes
Weiss	Yes	Yes	-	-	Yes	Yes	Yes

Classification of past land-cover

We used visual interpretation of maps and photos to reconstruct long-term (1900 – 1965) historical land cover. The Timber Cruise and USGS maps were georeferenced in ArcGIS 9.3 based on the Public Land Survey System section grid boundaries. Aerial photographs were scanned to a pixel resolution of 1 meter and orthorectified using ERDAS software. Once maps and aerial photos were georeferenced, land-cover polygons were digitized in ArcGIS. For the maps, we classified land cover based on polygons delineated by the map's creators. For the aerial photographs, we developed a common classification system based on visually distinctive patch types. A patch was considered a relatively homogeneous land-cover type that could be reasonably distinguished from its surrounding land cover (Robinson et al. 2005). For consistency, digitization was performed by one analyst at a scale of 1:10,000 for all time periods. Given the resolution of the aerial photos, one hectare was considered the smallest, consistently classifiable unit. Classification was reviewed by an independent GIS analyst at King County.

To quantify more recent (1986 – 2007) land-cover changes, we used satellite-based land-cover data. These were generated by the University of Washington Urban Ecology Research Lab using Landsat Thematic Mapper (TM) imagery and supervised classification with spectral unmixing (for more details see Alberti et al. 2006). The same interpretation methods were used for all datasets resulting in a consistent classification into 12 land-cover classes with a spatial resolution of 30 meters. We used ArcGIS software to quantify land-cover composition for each basin for all time periods. We decided not to use aerial photos for this time period because digitizing polygons is time consuming and satellite data were available. Also in more recent history, classifying land cover in urban regions is more complex due to the diversity and complexity of urban forms. As a result, differentiating between patch types based on visual assessments is more challenging and satellite-based analysis provides a consistent method for quantifying land cover. To assess the comparability of aerial photo interpretation compared with the satellite-based land-cover data, the County independently digitized land-cover polygons using high resolution (15 cm pixels) orthorectified aerial photos for 2007. Land-cover statistics calculated using this polygon layer correlated will with results from the satellite-generated land-cover data for 2007.

Data Limitations

There are several important limitations and sources of error to consider when interpreting historical datasets. The earliest records of land cover, i.e. the GLO and Timber Cruise surveys, are from hand-drawn maps created in the field by potentially different individual authors. The Timber Cruise maps include a wide variety of land-cover types. However, associated records do not provide any description or definitions of these land-cover types. It is possible, perhaps likely, that these land-cover categories were created in the field without any consistent classification criteria.

The 1911 Land Classification map is a small scale map (1:125,000). This coarse resolution means that detailed pockets of forest cover would not be mapped. For example, this map shows the entire Island of Vashon as converted to agriculture. While it is likely that the Island was dominated by agriculture at that time, perhaps even close to 100% converted, it seems unlikely that all forest cover was completely cleared.

The more modern datasets including the aerial photos and satellite imagery are more reliable and consistent. While the aerial photos vary in their resolution, the limited number of land-cover types present in these areas during this time period (1936-1965) further increases the reliability of these data. Some error is introduced when aerial photos are georeferenced. However, comparisons between digitized land-cover layers from two independent georeferencing efforts demonstrated that this error is small (< 5%).

Finally, some error is introduced in the satellite data classification process. An error assessment is conducted in which randomly selected pixels are compared to aerial photos to assess validity. The overall accuracy for each of the land-cover layers used in this analysis was as follows: 1986 – 62%, 1991 - 85%, 1995 - 86%, 1999 - 88%, 2002 - 72%, 2007 - 98%. For more details on accuracy assessment methods see Alberti et al. 2006.

Land-cover Change Analysis

We estimated land-cover change over time to characterize the history of each basin. We used the six datasets that provided the most comprehensive, comparable, and reliable coverage for the study basins: aerial photos from 1936, 1948, and 1965 and land-cover raster data layers from 1986, 1995, and 2007. To facilitate comparison across years, polygons digitized using aerial photos were converted to raster grids. Because our smallest mapped polygon was 1 hectare, we converted polygon data into 90 meter raster grids by converting polygon data directly to a 90 meter raster and resampled the 30 meter raster grids to 90 meters. Our second approach to increase consistency was to aggregate our land-cover classes into six general categories: forest, agriculture, cleared, built, wetland, and water (see Tables 2 and 3). Both of these data manipulations reduced the precision of our data, but increased comparability across years. We used cross tabulation in IDRISI Taiga (Clark Labs 2009), to calculate land cover transitions over time for each basin.

Table 2 Original satellite based land-cover classification and reclassificiation categories

Final reclassified land-cover class	Original land-cover class	Description
Built	Heavy Urban	>80% Impervious area
	Medium Urban	50-80% Impervious area
	Light Urban	20-50% Impervious area
Cleared	Cleared for development	Land currently being developed
	Grass	Developed grass and grasslands
	Clear cut forest	Clearcut forest
	Snow/bare	Snow/bare
Forest	Deciduous/mixed forest	10-80% Deciduous or mixed forest
	Conifer forest	>80% Coniferous forest
	Regenerating forest	Re-growing forest
Agriculture	Agriculture	Row crops, pastures
Wetland	Non-forested wetland	Non-forested Wetlands

Table 3 Original aerial photograph classification and reclassification categories

Final reclassified land-cover class	Original aerial photo land cover type	Description
Built	Buildings	Buildings not associated with agricultural fields
Built	Developed	Impervious Surface dominant (parking lots, rooftops), high(er) density regularly spaced housing not associated with agriculture.
Cleared	Forest – Sparse	Sparse (individual trees distinguishable across more than 50% of the polygon) individual trees covering 10-50% of polygon
		Extremely Sparse – individual trees cover <10% of polygon
	Shrub/ Regenerating forest	Medium darkness between grass and forest – covering at least 70% of the polygon – smooth, dense texture
	Grass	Medium Light – cleared of forest, but not as dark as regenerating forest
	Cleared - unknown	Open treeless areas often in regular (straight line) shapes or with sharp edges, near roads or buildings, not clearly attributable to a particular purpose.
Cleared	Cleared – Eroded/bare	Star-like shapes associated with logging roads
	Cleared for Timber	Visible downed timber, in forestry area with little development or agriculture. Logging roads visible.
Forest	Forest-Clumped	Clumped (Individual trees form clumps and blocks but overall the polygon is > 10% and less than 60% forested)
	Forest-Contiguous w/ gaps	Gaps occasionally visible but otherwise contiguous (>60% forested)
	Forest – Contiguous	Spaces are not visible between trees

Agriculture	Orchard	Regularly spaced trees
	Cleared for Agriculture	Cleared area surrounding one or more building structures
	Row Crops	Regular rows (lines) in cleared land
	Mixed Ag	Multiple small patches of ag (orchard, row crop, buildings, unknown purpose)
Water	Open Water	Lakes, large rivers, wetlands

Forest Buffer Analysis

We estimated the percent of forest cover at increasing distances to the stream channel for 1936, 1948, 1965, 1986, 1995, and 2007 to determine whether the study basins retained forest within the stream buffer over time despite more widespread de-forestation. For this analysis, we converted a vector hydrography layer created by King County to a 90 meter grid and calculated both the Euclidean distance and the hydrologic distance from the stream channel. Regulatory stream buffer widths are often measured using Euclidean distance. However, topography alters the flow path of water over land and therefore measures that incorporate topographic variability are potentially more accurate. To measure hydrologic distance, we used a digital elevation model to measure the distance from each pixel to the stream channel via the most likely overland flow path. Distances for both methods were binned into six distance classes: 0-90, 90-180, 180-270, 270-360, 360-450, > 450 meters from the stream. Forest cover composition was summarized for each distance class.

Results

Overall basin history

Overall, we found that the study watersheds share a common history of land-cover change, though the timing and extent of the change varied by watershed. Initially, all watersheds were presumed to be fully or near-fully forested then lost between 50% and 100% of their forest cover before 1948. Across all watersheds, forest cover increased substantially between 1948 to 1965. Since 1986, the treatment watersheds have been experiencing a slow but steady conversion of forest to residential and commercial development (see Figure 1). For presentation purposes, we have grouped the nine watersheds into three categories: 1) Vashon island catchments include Fisher, Judd, and Tahlequah basins; 2) eastern catchments include Taylor, Cherry and Weiss creeks; and 3) control catchments include Webster, East and West Seidels (Figure 1).

However, there are potentially important differences in the timing and progression of land-cover change among individual basins. Early records of forest cover including the USGS land classification map and the Timber Cruise report suggest that the three Vashon Island catchments (Fisher, Judd and Tahlequah) and Taylor catchment had all experienced significant forest clearing. According to the USGS land classification map, Vashon Island was primarily converted to agriculture by 1911. This may be an overestimate of clearing (see note on limitations for this dataset above) since aerial photos from 1936 show the three Vashon catchments retaining between 10 and 30% forest cover. The Timber Cruise records indicate that Taylor basin was significantly deforested (~70%) by 1907. In contrast, Timber Cruise maps recorded that the Webster catchment retained approximately 55% forest cover and the remaining catchments (Cherry, Weiss, East and South Seidel catchments) all retained 65-100% forest cover at this time.

In addition, three of the basins that were cleared earliest, Judd, Fisher, and Taylor, also recovered the least amount of forest between 1948 and 1965 relative to the other watersheds.

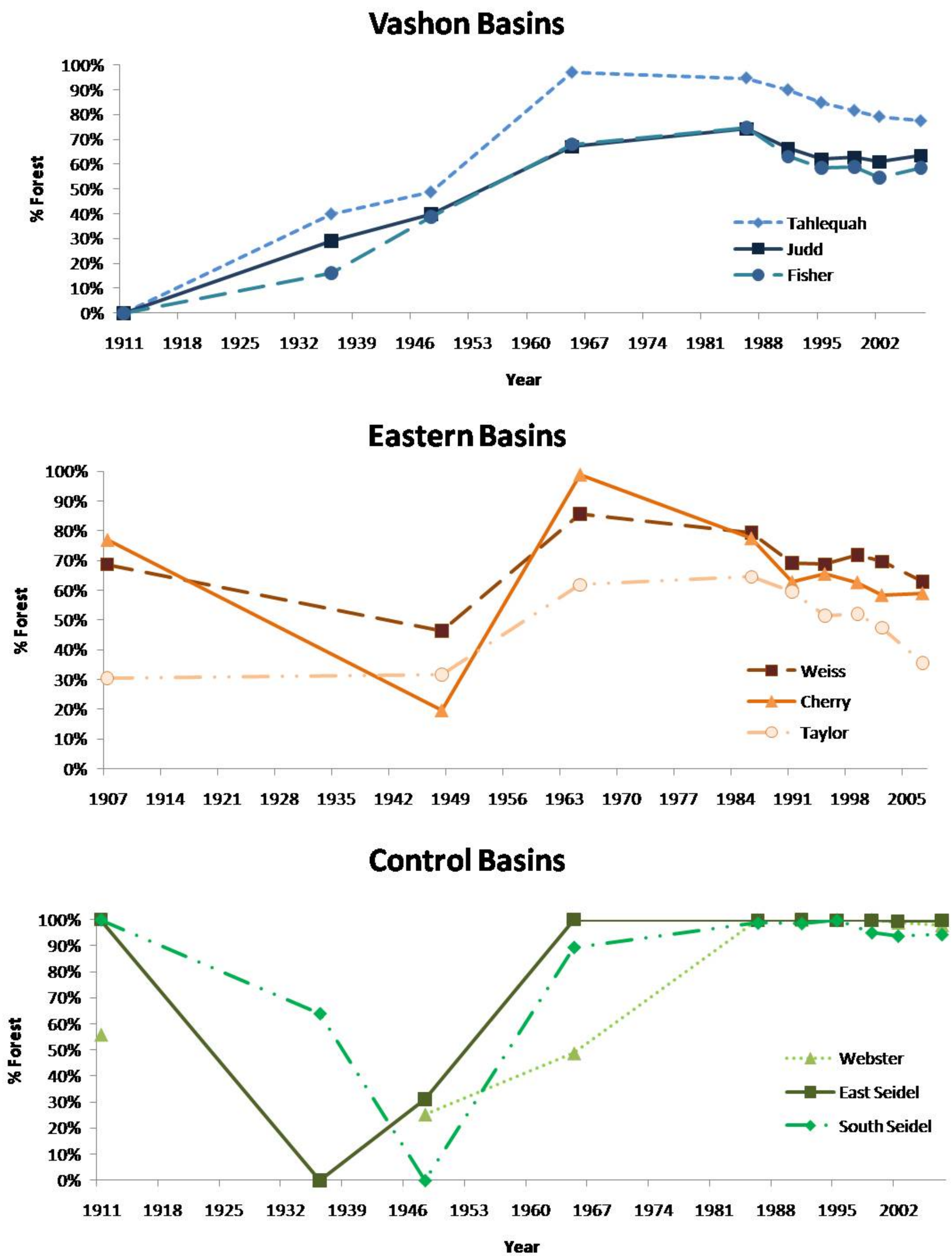


Figure 2 Forest cover change from 1907/1911 to 2007. Note that no data were available for Webster, Weiss, Cherry and Taylor basins from 1936.

Land-cover change analysis

In addition to finding a similar broad trajectory of forest-cover change across time, we also observed general trends in land-cover transitions over time across all watersheds (Table 4). All the study watersheds had been primarily cleared of forest at some point between 1936 and 1948. During the period from 1948 to 1965, the dominant land-cover trend was for cleared land to return to forest cover. Very little additional area was converted to agricultural areas after 1965. In contrast, some agricultural lands were converted into built lands during this time and some forested areas were directly converted into built areas. Beginning in 1986, a more significant conversion to built lands occurred. In contrast to the previous time period (1965 – 1986), development activities converted forested or cleared lands rather than agricultural lands. Also during this time, forest losses began to exceed forest gains.

Table 4 Land-cover transitions for each time period included in the analysis. Total forest loss and gain are also shown for each time period. By 1936, the Vashon catchments and East Siedel had all been cleared extensively. Consequently, the transition time period from 1936 to 1948 shows very little additional forest cover loss (which all occurred in South Seidel). Similarly, the remaining basins were at their least forested point in 1948, data were not available for Weiss, Cherry, Taylor, and Webster basins in 1936, leading to a smaller over all study area for that time period.

	1936-1948		1948-1965		1965-1986		1986-1995		1995-2007	
Transition	**Hectares**	**%**	**Hectares**	**%**	**Hectares**	**%**	**Hectares**	**%**	**Hectares**	**%**
Forest to Cleared	87	3%	67	1%	201	4%	302	6%	136	3%
Forest to Agriculture	4	0%	56	1%	1	0%	4	0%	2	0%
Forest to Built	0	0%	5	0%	17	0%	237	5%	323	7%
Forest Persistence	601	24%	1605	33%	3323	68%	3604	74%	3257	66%
Cleared to Forest	329	13%	1841	38%	455	9%	126	3%	180	4%
Cleared to Agriculture	40	2%	137	3%	0	0%	6	0%	53	1%
Cleared to Built	0	0%	7	0%	8	0%	198	4%	121	2%
Agriculture to Built	0	0%	0	0%	72	1%	2	0%	1	0%
Agriculture to Forest	4	0%	85	2%	365	7%	2	0%	2	0%
Agriculture to Cleared	28	1%	53	1%	338	7%	5	0%	2	0%
Total Forest Gain	**332**	**13%**	**1926**	**39%**	**831**	**17%**	**131**	**3%**	**202**	**4%**
Total Forest Loss	**90**	**4%**	**129**	**3%**	**226**	**5%**	**550**	**11%**	**478**	**10%**
Total Study Area	**2512***		**4900**		**4900**		**4900**		**4900**	

Individual basin narratives

1850 to 1907/1911 (Primary Forest and Clearing Phase)

According to the General Land Office Survey, the early history (1857 – 1892) of land-cover change within the study catchments is dominated by deforestation. According to a land classification map created by USGS, by 1911, the majority of forested land on Vashon Island forest had been cleared and converted to agriculture. By contrast, maps of land cover and land use created during a timber cruise in 1907 indicate that the extent of forest clearing was variable in the eastern basins. The Seidel watersheds had experienced no forest clearing while the Webster, Cherry and Weiss systems retained over 50% of their forest cover. The Taylor watershed experienced the most significant deforestation, retaining only 30% forest cover. Taylor was also the only eastern watershed with some agriculture in 1907.

Agricultural development occurred early and was extensive in the Vashon basins compared to the other basins. The Vashon watersheds had a significant and extensive period of agricultural cultivation during these early years. The Taylor catchment was the only eastern basin with even small amounts of agriculture (3% of basin area) during this time.

1936 to 1948 (Continued Clearing and Early Forest Regrowth Phase – data available for Vashon catchments and East and South Seidels only)
By 1936, Fisher, Judd, and Tahlequah and East Seidel basins had been extensively cleared and retained only 16%, 29%, 40%, and 0% of their forest cover respectively. South Seidel retained 64% of its original forest cover. Between 1936 and1948, forest cover increased between 14-24% in each of the Vashon and East Seidel catchments while the remaining forest in South Seidel had been cleared. Overall during this time, 333 hectares (13% of the study area with data for 1936) of previously cleared land had re-forested while 85 new hectares (3%) of forest land were cleared resulting in a *net increase of approximately 250 acres of forest in these five basins*. In addition, approximately 40 (2%) hectares of cleared land was converted to agriculture.

1948 to 1965 (Maturing Regrowth Phase)
During this time period, forest cover in all the study catchments increased substantially. By 1948, approximately 1800 hectares (35% of the study area) had returned to forest cover, while only 70 new hectares (1%) had been cleared, *resulting in a net increase of 1730 acres of forest (35%) between 1948 and 1965*. Afforestation did not occur evenly across all basins. By 1965, forest cover in the Webster, Taylor, Fisher, and Judd watersheds ranged from 49% and 67%. In contrast, forest cover in remaining catchments exceeded 80%.

Agricultural land cover peaks during this time period due to *a net increase of approximately 55 hectares of agricultural land cover* (approximately 130 hectares of cleared land converted to agriculture, and 85 hectares of agricultural land reverted to forest). In the Taylor, Fisher and Judd systems a small percentage of cleared land (4-7%) was converted to agriculture. By 1965, agriculture increased to 33%, 10%, and 25% of land in each of these three basins, respectively. In addition, 1965 is the first time (within the constraints of the data available) that housing and commercial ("built") land cover is recorded in these catchments (approximately 10 hectares).

1965 to 1986 (Secondary Forest and Development Phase)
By 1986, the study watersheds were largely forested. The control watersheds (Webster, East Seidel and South Seidel) were protected from development and forest clearing, and retained 99% of their forest cover. They remain 99% forested to the present day. For all watersheds, there was a *net increase of 605 hectares of forest* cover during this time period as agricultural land and cleared lands reverted to forest cover. A new trend emerged during this time period as about 20 hectares of forest and 70 of agriculture were converted to residential and commercial development. All transitions to built land cover occurred in the Judd and Taylor basins (3% each). While forest and developed land covers increase, agriculture experienced the greatest net loss of cover (780 hectares). Of this loss, 365 hectares was due to reversion of agricultural lands to forest and 70 hectares were converted into built cover.

During this time period, the control and treatment catchment histories begin to diverge. By 1986, the three control catchments are fully forested and remain to until present day. For the treatment basins, forest cover peaks in 1986. The Tahlequah catchment reverts to a nearly fully (95%) forested state. The Weiss, Cherry, and Fisher catchments reach a peak of 75-77% forest cover. Taylor and Judd have the lowest amount of forest recovery (65% and 74% respectively). In addition, residential and commercial development begins in Taylor and Judd. Judd, Fisher and Taylor catchments lost a significant amount of agricultural land cover to abandonment (cleared; 13%, 11%, and 12%) and aforestation (10%, 10%, and 20%).

1986 to 1995 (Development Phase)
During the time period between 1986 and 1995, forest cover begins to slowly decline as it is replaced by residential and commercial development. Approximately 300 hectares of forest land were cleared and 240 hectares of forest were converted to development. *Overall, there was a net loss of approximately 420 hectares of forest cover.* Built land cover increased by almost 440 hectares. At this point, forest cover remained above 50% for all basins, and at 52%, the Taylor basin had the least forest cover. Otherwise, the basins were all near (59% for Fisher) or above 60% forested.

The increase in residential and commercial development was not evenly distributed across the basins (Table 6). The Cherry, Weiss, and Tahlequah basins showed less than 10% developed land cover while the Taylor, Fisher, and Judd basins all showed greater than 15% developed land cover. Virtually all the developed land cover was converted from land that was forest or cleared in 1986. Cherry and Weiss are the only basins that showed any increases in forest cover (12% and 7% respectively). The Fisher, Weiss, Taylor, and Judd basins all showed additional forest clearing (10%, 9%, 8% and 7% respectively). Otherwise, all changes were due to conversion of cleared and forested land to development

Table 5 Percent of each basin transitioning from cleared, forest or agriculture to built lands between 1986 and 1995.

	Cleared to Built	Forest to Built	Agriculture to Built
Cherry	2%	6%	0%
E. Seidel	0%	0%	0%
Fisher	6%	6%	0%
Judd	7%	4%	0%
S. Seidel	0%	0%	0%
Tahlequah	1%	5%	0%
Taylor	7%	10%	0%
Webster	0%	0%	0%
Weiss	1%	3%	0%

1995 to 2007 (Continued Development Phase and Just Prior to Start of Effectiveness Monitoring)
Between 1995 and 2007, built land cover increased by about 420 hectares. About 320 hectares of forest and 120 hectares of cleared land were converted into built. In turn, approximately 180 hectares of cleared land reverted to forest, while 140 hectares of forest were cleared. Around 50

hectares of cleared land were reclassified as agriculture. This reclassification may be at least partially due to classification error, as distinguishing between cleared and agricultural land can be difficult.

Within all treatment catchments, significant land-cover change is primarily from forest into built. Taylor experienced the most significant increase in development (20%) with the majority of this conversion occurring on forested land (Table 5). The Weiss basin underwent the next most significant increase in development, also primarily on forested lands. Cherry and Tahlequah both experience small amounts of development of forested lands. Fisher and Judd both experience a small increase in development which occurs both on previously cleared and previously forested lands.

Table 6 Percent of each basin transitioning from cleared or forest to built lands between 1995 and 2007. No agricultural lands converted to built during this time period.

	Cleared to Built	Forest to Built
Cherry	1%	8%
E. Seidel	0%	0%
Fisher	5%	3%
Judd	3%	2%
S. Seidel	0%	0%
Tahlequah	0%	5%
Taylor	4%	16%
Webster	0%	0%
Weiss	2%	10%

Riparian forest-cover analysis results

Though the catchments were primarily deforested by 1936 (Vashon and East Seidel) and 1948 (Cherry, Weiss, Taylor, South Seidel, and Webster), riparian areas (defined here as areas within 90 meters of the stream channel) generally retained proportionally more forest cover than other distance classes and the study area as a whole (Figure 3). Indeed, 1965 is the only year during which riparian areas are not more forested than other distance classes, and this finding is only the case when Euclidean distance measures are used. This is likely because by 1965 several of the basins were nearly 100% forested. Since 1986, riparian areas again showed a greater proportion of forest cover than other distance classes or the study area as a whole. Excepting 1965, these patterns hold for both Euclidean and hydrologic distance measures. The 2007 land-cover data in particular reveal a steady decrease in proportional forest cover with increasing distance from stream channels.

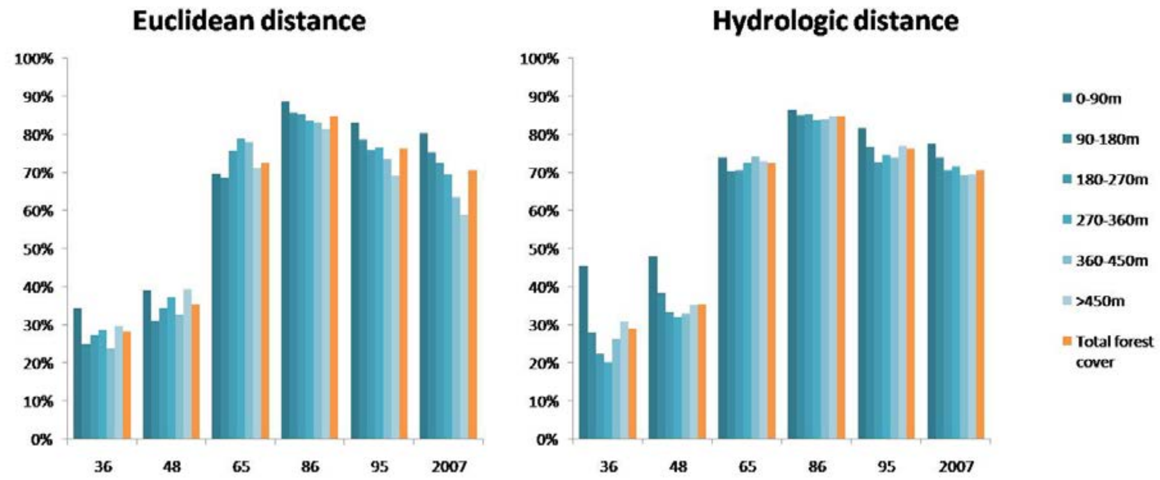


Figure 3 shows the percent of each distance class in forest cover for the entire study area. Total forest cover refers to the total forest cover across all basins for that year. Distance class is measured by Euclidean distance from the stream channel and hydrologic distance measures the distance of the water flow path to the stream channel based on basin topography.

The relatively high proportion of forest cover retained in riparian areas can be seen in most, but not all of the individual basins. In addition, results from individual basins reveal potentially important differences between Euclidean and hydrologic measures of distance. In 1936 and 1948 all the basins except Taylor and South Seidel showed higher proportion of riparian forest cover. In addition, the Judd and Webster basins do not show this pattern for Euclidean distance measures, but do show higher percentages of forest cover in the riparian zone using hydrologic distance measures (Figure 4). Webster in particular has a very steep topographic profile, which likely explains the significant difference in forest cover measures between the two distance measures.

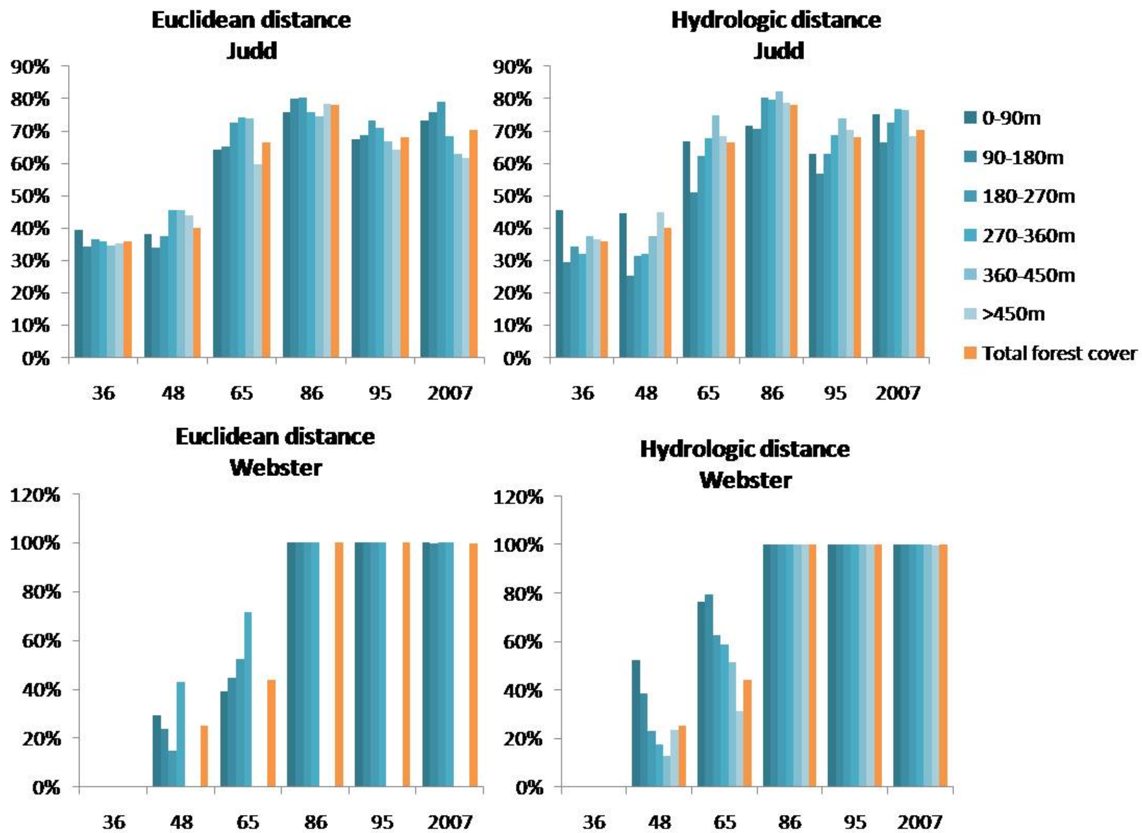


Figure 4 compares forest cover results measured using Euclidean and hydrologic distance for Judd and Webster. These two basins show the greatest qualitative difference for forest cover results using the two distance measures. Total forest cover refers to the total % forest cover for the basin overall.

From 1986 to 2007, most of the basins again showed a pattern of declining forest cover as distance from the stream channel increases (Figure 5). This is the case for the Fisher, Taylor, Weiss, and Cherry basins (using hydrologic distance measures). The Webster, East Seidel and South Seidel catchments are all virtually 100% forested, and so logically show no patterns of forest cover by distance from stream. Tahlequah was also primarily forested during these years, and shows no relationship between distance from the stream and forest land cover. The Judd basin also retained a moderately high level of forest cover compared to other distance classes during these years, but this pattern was not as pronounced as those seen in many of the other basins.

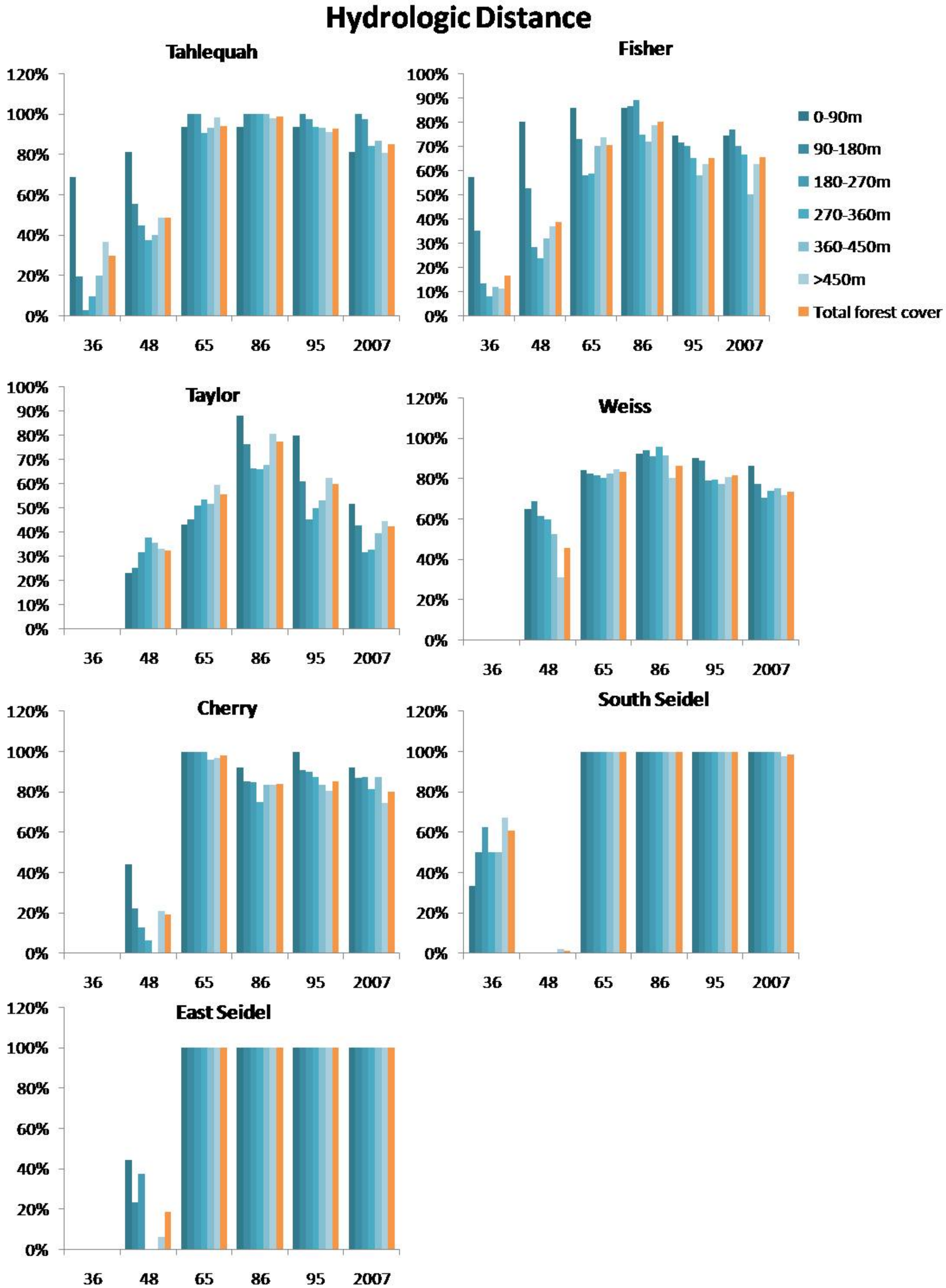


Figure 5 shows the results of the forest cover by hydrologic distance from stream channel for all basins not shown in Figure 3. Taylor, Weiss, and Cherry basins show 0% forest cover in 1936 because data were not available for those areas in that year.

Discussion

Land-cover history and change

Numerous studies have found that land-cover changes correlate with changes in watershed condition. Factors known to relate to watershed condition include total forest cover throughout the basin (Booth et al. 2002), riparian forest cover (Naiman et al. 2000), impervious surface (Walsh et al. 2005, Booth et al. 2002, Alberti et al. 2007), agriculture (Cuffney et al. 2009, Harding et al. 1998) and timber harvest (Moore and Wondzell 2005, Fuchs et al. 2003, Haggerty et al. 2004). Most previous studies have generally focused on relating stream responses to changes in a single type of land cover at a snapshot in time. Our analysis isolates individual land-cover impacts by quantifying transitions through multiple stages of dominant land use and forest cover patterns for each study watershed. Reconstructing the long-term history of a watershed raises new questions about the relative importance of the duration, intensity, and time that has elapsed since a given impact as well as the interactions among multiple impacts over time. These findings confound simple relations between watershed condition and land-cover characteristics for any moment in time because there may be legacy effects of previous land uses.

The study watersheds all experienced significant deforestation prior to 1965, a period of afforestation up until 1986. Currently, the treatment catchments are on a relatively slow trajectory of deforestation based on the rate of changes observed from 1986 to 2007. However, the timing, extent and spatial distribution of forest cover loss differed in modest but possibly important ways for each basin. Previous studies have found that logging can significantly alter aquatic macroinvertebrate communities and increase peak stream flows (ref e.g., Stencil et al). However, the degree to which, the effects of sedimentation and wood removal on ecosystem function remains unknown. Other studies have demonstrated that logging effects diminish within the first two decades following logging as forest cover regenerates (). The Tahlequah, Cherry, Weiss and all three control catchments experienced an intense but relatively short period of deforestation, and all recovered to more than 80% forest cover between 1965 and 1986 (Figure 2), at least two decades prior to the start of the Critical Areas effectiveness study. According to previous studies (Fuchs et al. 2003, Haggerty et al. 2004, Moore and Wondzell 2005), the hydrology of these basins has likely recovered from logging effects. In contrast, the Judd, Fisher and Taylor basins experienced similar deforestation, but only recovered to approximately 65-75% forest cover between 1965 and 1986. To what extent this subtle difference in forest recovery may lead to differences in watershed condition is unclear.

Watershed condition, specifically the presence of agriculture, prior to development likely influences the sensitivity of individual watersheds to urbanization (Brown et al. 2009). We found that Fisher, Judd and Taylor had the most significant agricultural history of all our study catchments. Residential and commercial development in these catchments occurred on both previously cleared (or agricultural) lands and previously forested lands. In contrast, the majority of development that occurred in the Cherry, Weiss and Tahlequah catchments occurred on forested land, with little or no intervening agricultural stage (Tables 5 and 6). This differential history may influence the sensitivity of these catchments to future residential and commercial development. Brown and others (2009) suggest that aquatic macroinvertebrate communities in basins with an agricultural history have already sustained significant declines prior to urban development, and therefore urban development does not have a strong additional effect. In contrast, forested basins are likely to have relatively intact aquatic macroinvertebrate communities that will be significantly altered as the basin develops. Based on the findings of Brown et al. (2009), we therefore would expect that Fisher, Judd and Taylor would be starting out in a more degraded macroinvertebrate community and therefore show less of a decrease in BIBI over time compared to Cherry, Weiss, and Tahlequah as development in these catchments increases.

Riparian forest-cover analysis

Retaining forest cover within the riparian zone can protect streams from land-cover changes occurring throughout the basins (Sweeney 1993, Vuori and Joensuu 1996), although protection depends to some extent on the width of the buffers (Haggerty et al. 2004). During the logging period from 1936 to 1948, all the basins except Taylor and East Seidel retained 40 to 60% percent of their riparian (defined as areas within 90 meters of the stream channel) forest cover even though by 1948 these basins overall were only 20 to 40% forested (Figures 4 and 5). The retention of forest cover within the riparian zone may have reduced the impacts of early logging on overall watershed condition. If this is the case, most of the basins in the study should have maintained relatively good watershed condition, even during the periods of extensive logging in 1930s and 1940s. The exceptions to this are the Taylor and East Seidel basins, which both retained very little riparian forest during early deforestation. The lack of riparian forest may not have had much long-term impact for East Seidel, which was completely re-forested by 1965. However, the extent of afforestation in the Taylor basin only reached a level of about 65% in 1986, before starting to decline. In more recent decades, more riparian forest has been retained in the Taylor basin relative to the basin overall.

Distance was measured using both hydrologic and Euclidean distance. For some basins, the two measures resulted in qualitatively different results. However, overall, the trend towards greater protection of riparian forest cover (as opposed to forest cover in the rest of the catchment) was more pronounced when distance was measured using hydrologic rather than Euclidean distance (Figure 4). Because hydrologic distance is based on topography, it seems plausible that this measure may be capturing patterns of forest cover within each basin with greater accuracy than Euclidean distance. Topographic features such as steep slopes near small streams may have made logging up to the stream edge difficult.

Temporal intercorrelations in land-cover change

These findings demonstrate that Fisher, Judd and Taylor basins were cleared earlier and remained deforested longer, experienced more agricultural development, and began to develop earlier and to a greater extent than the other three treatment basins (Cherry, Weiss, and Tahlequah). Based on these historical characteristics, at the start of the Critical Areas Effectiveness study in 2007, the Fisher, Judd and Taylor catchments should have the most impaired watershed condition of all the study catchments. However, it is important to note that although there are real differences in land-cover history among the basins, it is not yet known whether these differences lead to differential ecological outcomes (though this is the focus of the broader study of which this is a part).

These findings also raise an additional important consideration that pertains to all studies attempting to link land cover to ecological condition: that land-cover characteristics within and across time periods are not independent. King and others (2009) found that intercorrelations among commonly used predictor variables challenge our ability to detect causal relationships between land cover and ecological stream indicators. In this study, we find similar issues inherent to comparing historical land use to current watershed condition. For example, Judd, Fisher and Taylor did not recover as much forested land as the other six basins. This is likely because these basins were also converted to agricultural land use, leaving less land available for afforestation. This correlation makes it difficult to distinguish between the effects of lower overall forest cover and the presence of agriculture. In addition, when development first started in these areas, existing agricultural areas were more likely to convert to residential and commercial development than to forested lands. Therefore, those basins with an agricultural history may be more prone to develop earlier. Early settlement in a particular area is likely due to a combination of factors including the underlying geomorphology of the basin and proximity to more developed cultural centers, in this case the City of Seattle.

Lastly, changes in forest cover are not independent over time. In this study, areas that were forested in one time period were cleared in the next and regenerating during the yet the time period. This correlation challenges our ability to relate historic to present day conditions because the direction of the relationship switches depending on the time period selected. Studies investigating the effects of historic conditions on present day conditions often select one point in time (e.g. Harding et al. 1998) because it is difficult to obtain data and to analyze more than one time period. However, these studies may be missing important information about impacts that occurred before or after the time period selected for the analysis.

Conclusions

The land cover history of any given watershed potentially has significant implications for the current as well as the trajectory of watershed condition over time. Significant intercorrelations among land-cover characteristics within and across any given point in time challenges our ability to attribute changes in watershed condition to any particular land use or land-cover change event. However, understanding the long-term history of a given watershed is likely to be informative in explaining observed changes over time. In particular, understanding the variation in land-cover change history is potentially important for understanding and explaining variation in the relationship between land-cover change and watershed condition. Although it is difficult and time consuming to reconstruct land-cover conditions for all watersheds, it is worthwhile to at least develop a broad understanding of historic land-cover conditions for those areas that will be intensively monitored. There is a significant need for long-term monitoring of watershed condition, and building an understanding of the overall trajectory of the system is likely to be important in properly interpreting the results of those monitoring efforts.

Citations:

Alberti M., Marzluff J.M., Waddell P. 2006. Modeling interactions among urban development, land-cover change, and bird diversity. NSF Final Report BE/CNH 0120024.

Alberti M, Booth D, Hill K, Coburn B, Avolio C, Coe S, and Spirandelli D. 2007. The impact of urban patterns on aquatic ecosystems: an empirical analysis in Puget lowland sub-basins. Landscape and Urban Planning 80:345-361.

Basnyat, P., L. D. Teeter, K. M. Flynn, and D. G. Lockaby. 2000. The use of remote sensing and GIS in watershed level analyses of non-point source pollution problems. Forest Ecology and Management 128:11, 65-73.

Booth, D.B., D. Hartley, and R. Jackson. 2002. Forest cover, impervious surface area, and the mitigation of stormwater impacts. Journal of the American Water Resources Association 38:835-845.

Brett, M.T., G.B Arhonditis, S. E. Mueller, D. M. Hartley, J. D. Frodge, and D. E. Funke. 2005. Non-point-source impacts on stream nutrient concentrations along a forest to urban gradient. Environmental Management 35:330-342.

Cuffney, T. F., R. A. Brightbill, J. T. May, and I. R. Waite. 2009. Responses of benthic macroinvertebrates to environmental changes associated with urbanization in nine metropolitan areas. Ecological Society of America – Preprint.

Brown, L.R., Cuffney, T. F., Coles, J.F., Fitzpatrick, F., McMahon, G., Steur, J., Bell, A.H., and J.T. May. 2009. Urban streams across the USA: lessons learned from studies in 9 metropolitan areas. J. N. Am. Benthol. Soc. 28(4): 1051-1069.

Fetherson, K. L., R. L. Naiman, and R. E. Bilby. Large woody debris, physical process, and riparian forest development in montane river networks of the Pacific Northwest. Geomorphology 13:133-144.

Franklin, J. F., Cromack, K. Jr., Denison, W.; McKee, A.; Maser, Chris, Sedell, J., Swanson, F., Juday, G. 1981. Ecological characteristics of old-growth Douglas-fir forests. Gen. Tech. Rep. PNW-GTR-118. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Groffman, P. M., N. L. Law, K. T. Belt, L. E. Band, and G. T. Fisher. 2004. Nitrogen fluxes and retention in urban watershed ecosystems. Ecosystems 7:393-403.

Handler, N. B., A. Paytan, C. P. Higgins, R. G. Luthy, and A. B. Boehm. 2006. Human development is linked to multiple water body impairments along the California coast. Estuaries and Coasts 29:860-870.

Harding, J. S., E. F. Benfield, P. V. Bolstad, G. S. Helfman, and E. B. D. Jones III. 1998. Stream biodiversity: The ghost of land use past. Proceedings of the National Academy of Sciences 95:14843-14847.

Haydon, S.R., R. G. Benyon, and R. Lewis. 1996. Variation in sapwood area and throughfall with forest age in mountain ash (Eucalyptus regnans F. Muell.). Journal of Hydrology 187:351-366.

King, R. S., M. E. Baker, D. F. Whigham, D. E. Weller, T. E. Jordan, P. G. Kazyak, and M. K. Hurd. 2005. Spatial considerations for linking watershed land cover to ecological indicators in streams. Ecological Applications 15:137-153.

Moore, R. D. and S. M. Wondzell. 2005. Physical hydrology and the effects of forest harvesting in the Pacific Northwest: a review. Journal of the American Water Resources Association 41:763-784.

Naiman, R. J., R. E. Bilby, and P. A. Bisson. 2000. Riparian ecology and management in the Pacific Coastal Rain Forest. Bioscience 50:996-1011.

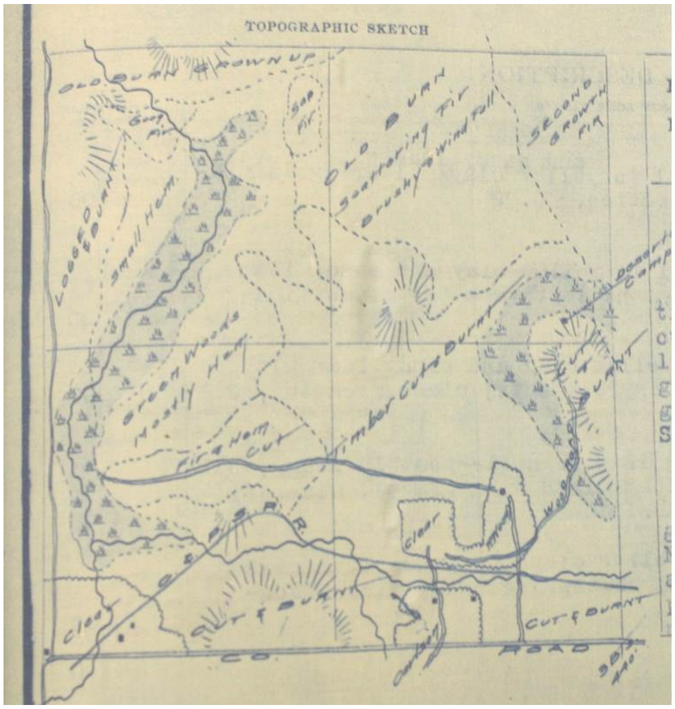
Pan, Y., A. Herlihy, P. Kaufmann, J. Wigington, J.van Sickle, and T. Moser. 2003. Linkages among land-use, water quality, physical habitat conditions and lotic diatom assemblages: A multi-spatial scale assessment, Hydrobiologia 515:59-73.

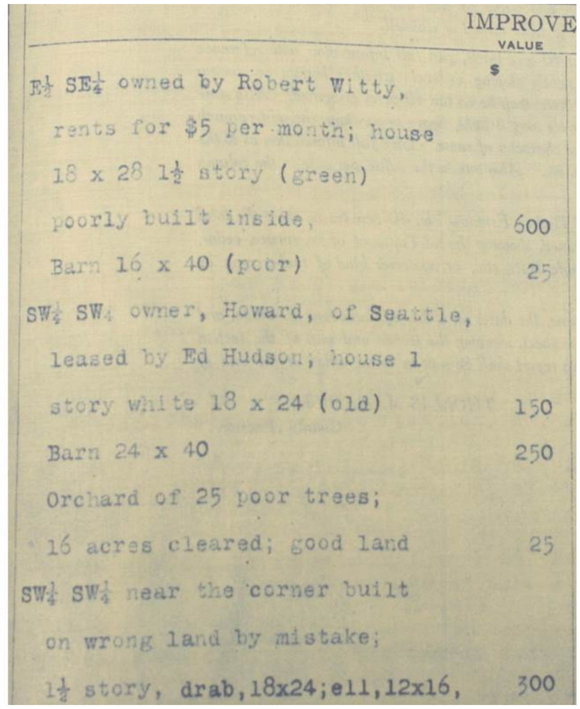
Valett, H. M., C. L. Crenshaw, and P. F. Wagner. 2002. Stream nutrient uptake, forest succession, and biogeochemical theory. Ecology 83: 2888-2901.

Walsh, C. J., A. H. Roy, J. W. Feminella, P. D. Cottingham, P. M. Groffman, R. P. Morgan II. 2005. Journal of the North American Benthological Society 24:706-723.

Zampella, R. A., N. A. Procopio, R. G. Lathrop, and C. L. Dow. 1999. Relationship of land-use/land-cover patterns and surface water quality in the Mullica River basin. Journal of the American Water Resources Association 43:594-604.

Appendix A: Data examples





IMPROVE
VALUE
$

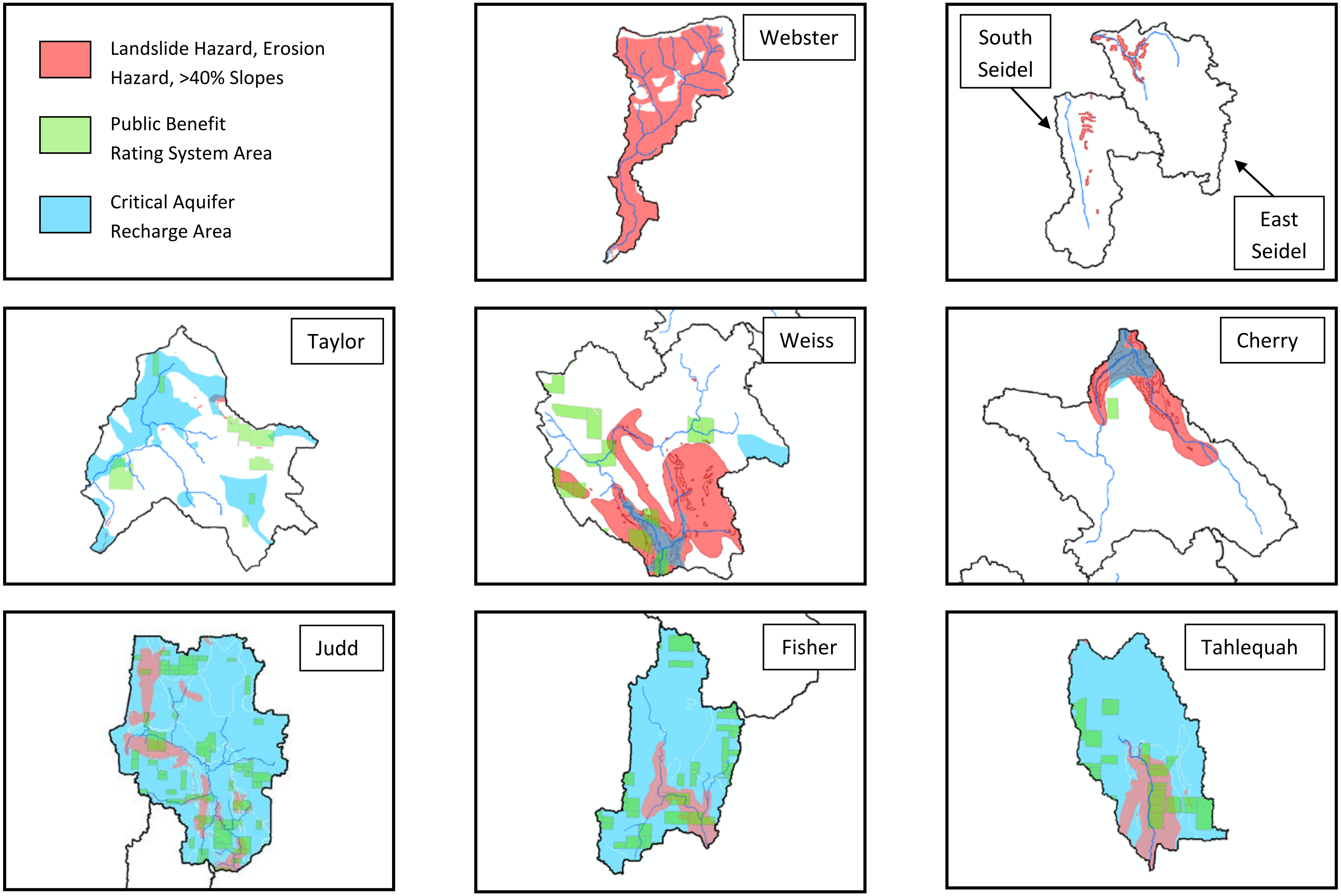
E½ SE¼ owned by Robert Witty,	
rents for $5 per month; house	
18 x 28 1½ story (green)	
poorly built inside,	600
Barn 16 x 40 (poor)	25
SW¼ SW¼ owner, Howard, of Seattle,	
leased by Ed Hudson; house 1	
story white 18 x 24 (old)	150
Barn 24 x 40	250
Orchard of 25 poor trees;	
16 acres cleared; good land	25
SW¼ SW¼ near the corner built	
on wrong land by mistake;	
1½ story, drab, 18x24; ell, 12x16,	300

Figure 6 Timber cruise map from TN22 RN6 and associated notes.

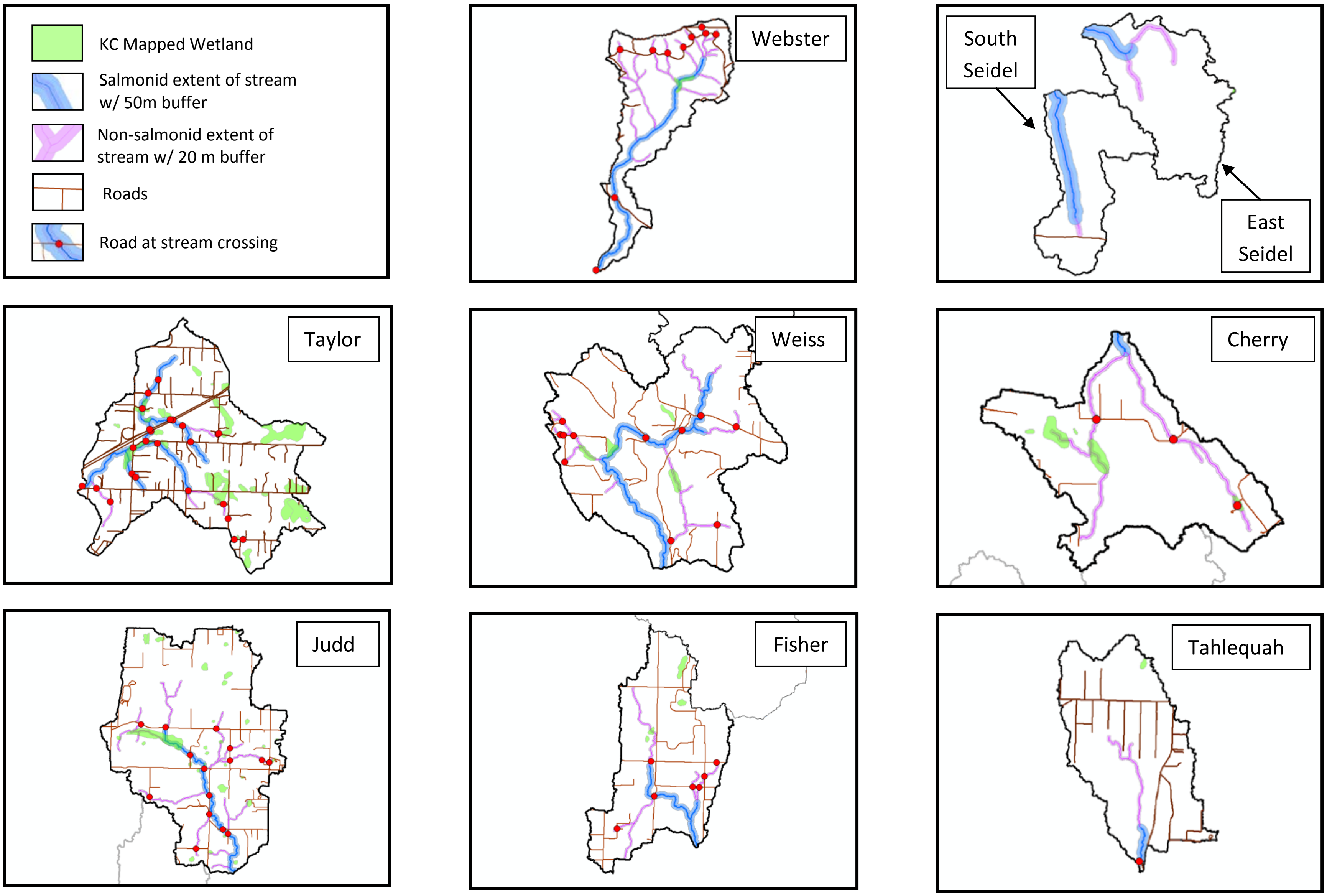
Appendix E.
Study watershed map panels

- **Steep Slopes, PBRS, CARAs**
- **Wetlands, Buffers, Roads, & Crossings**
- **Basin Soil Permeability based on Surficial Geology**
- **Basin Soil Types**
- **Orthophotos for each photo year, locations of landcover change, and full build out**

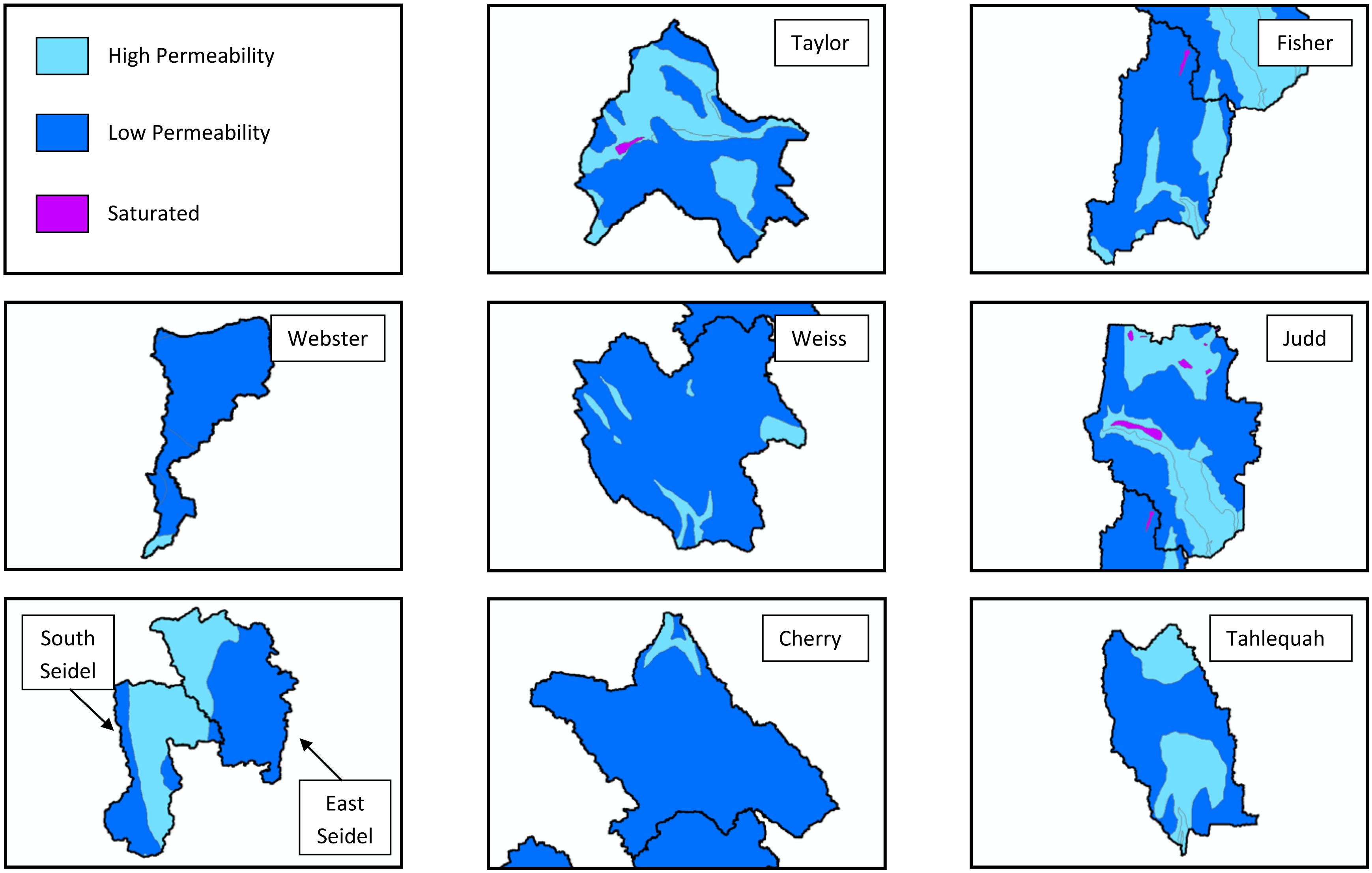
Steep Slopes, PBRS, CARA's



Wetlands, Buffers, Roads, & Crossings

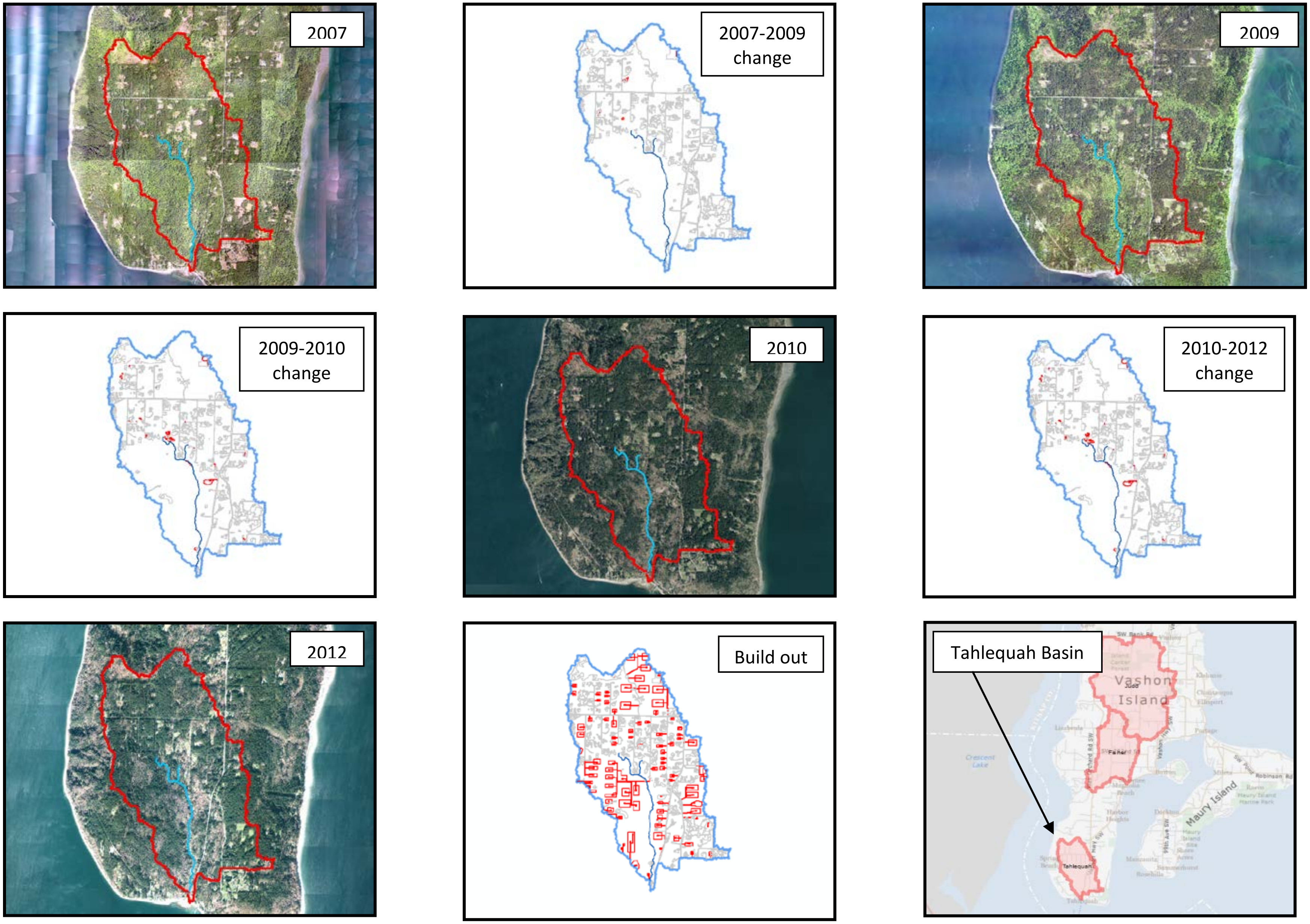


Basin Soil Permeability based on Surficial Geology

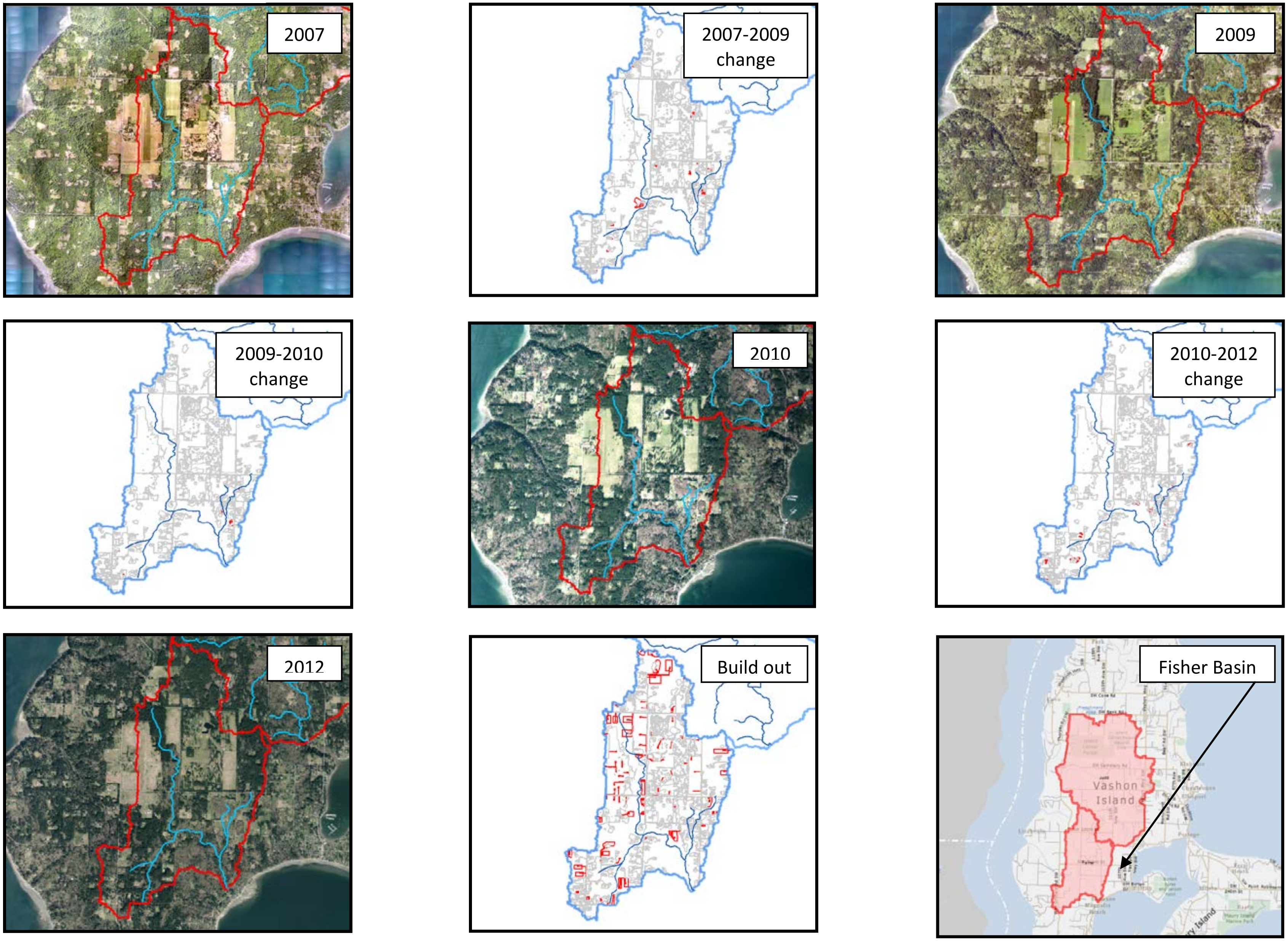


Basin Soil Types

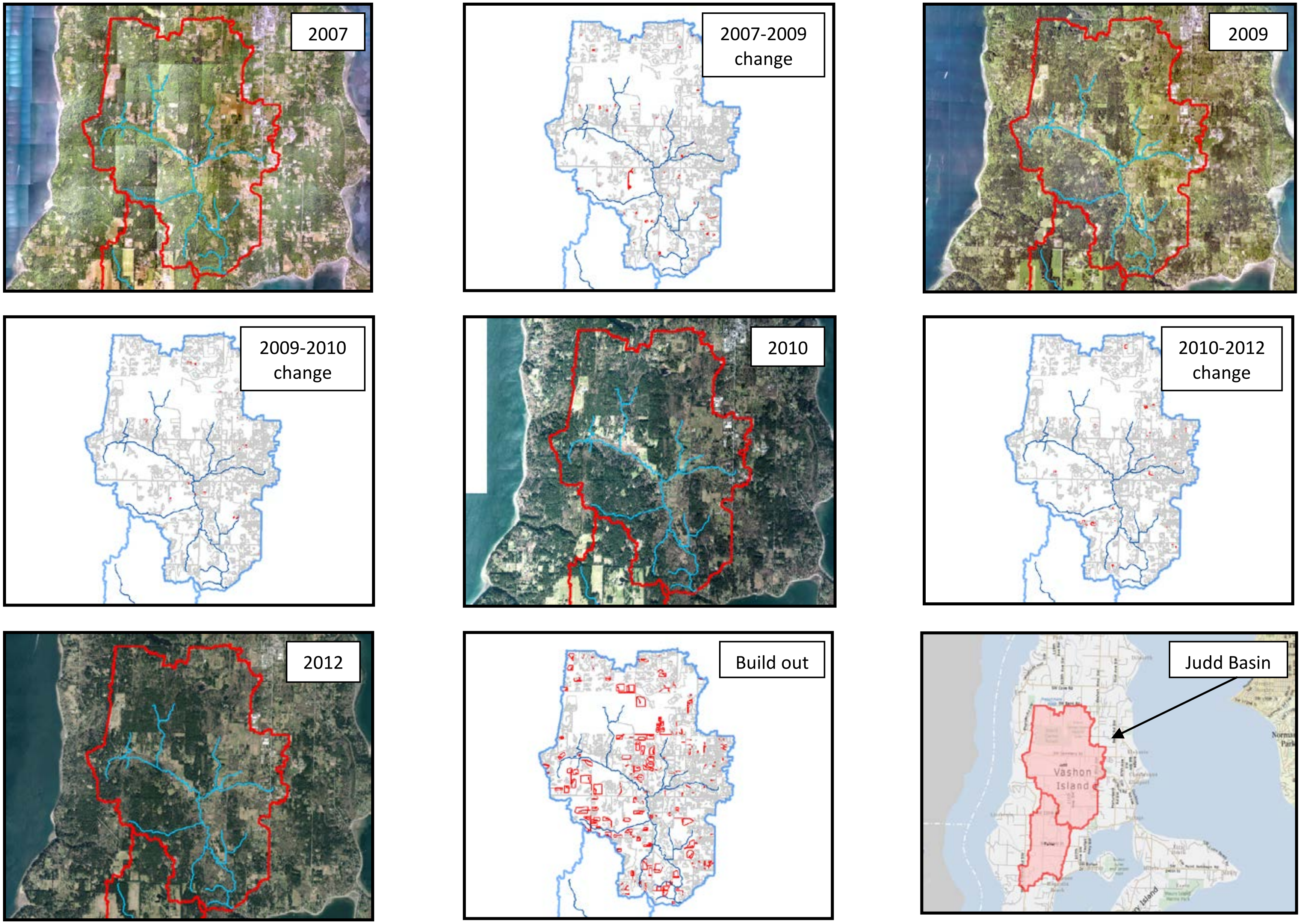
Tahlequah Landcover



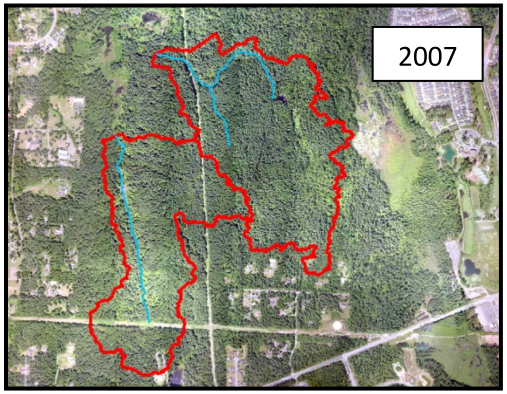
Fisher Landcover

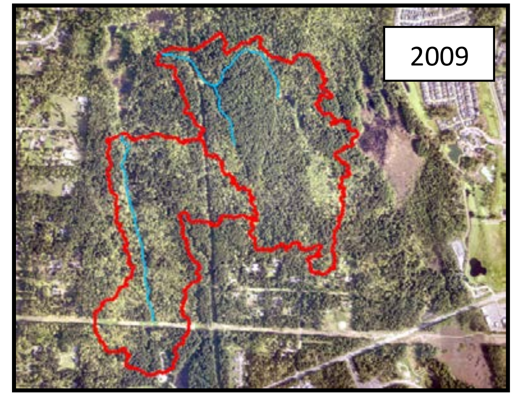


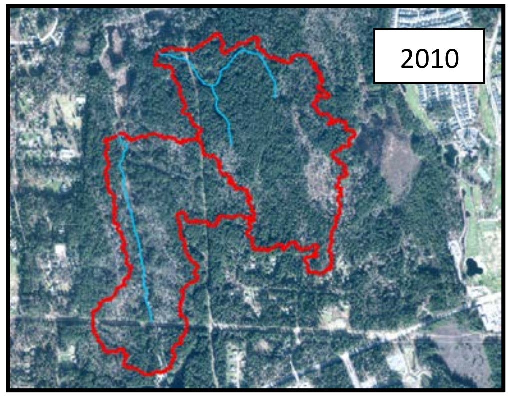
Judd Landcover

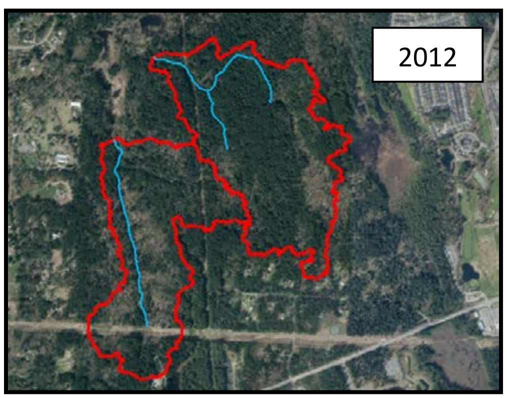


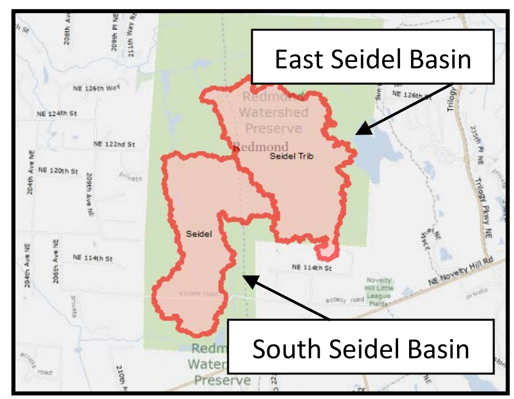
South Seidel & East Seidel Landcover



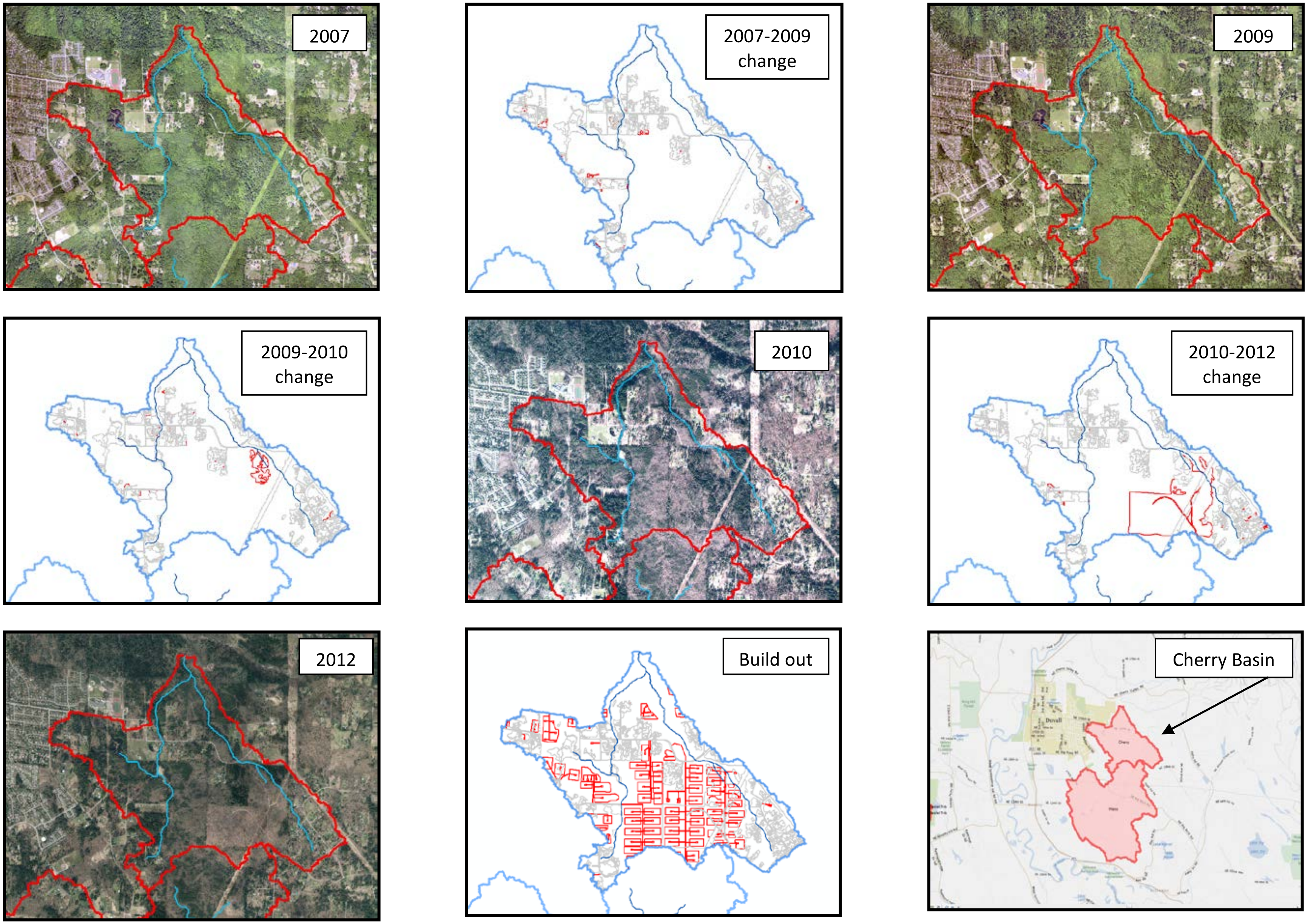




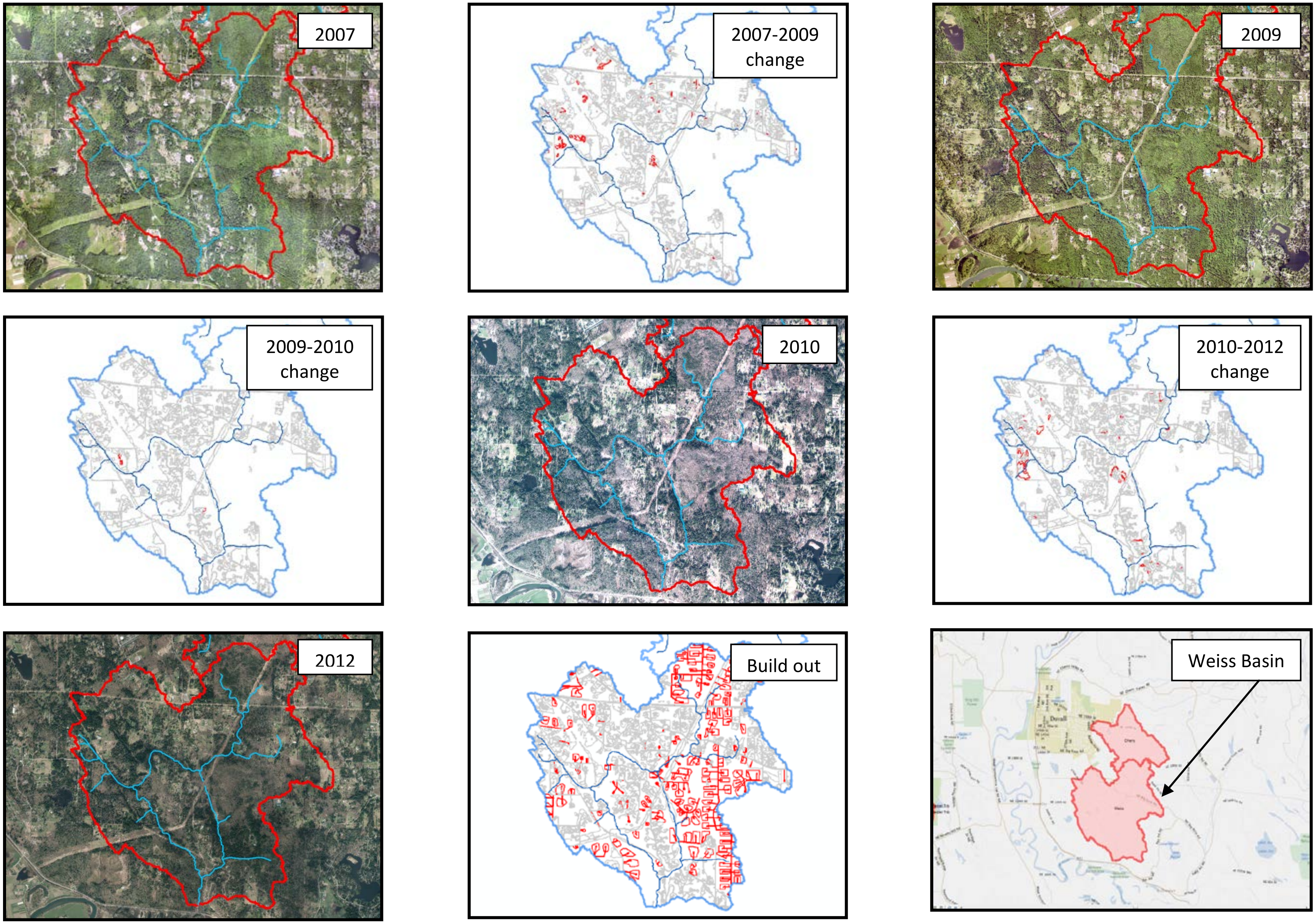




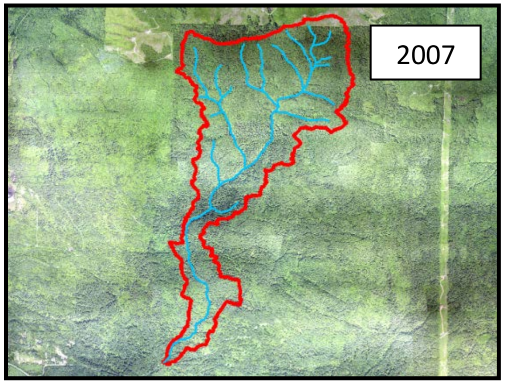
Cherry Landcover

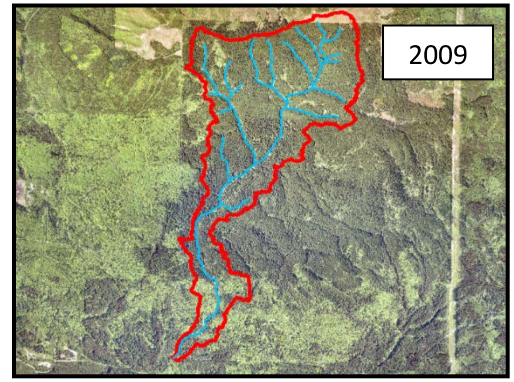


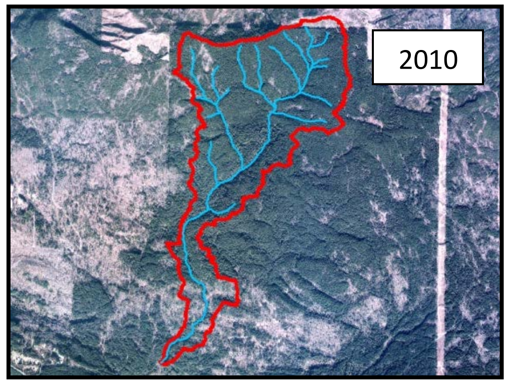
Weiss Landcover

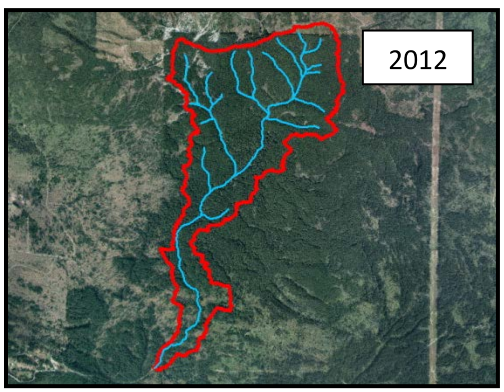


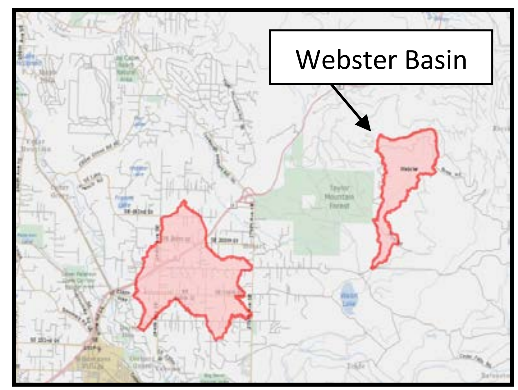
Webster Landcover



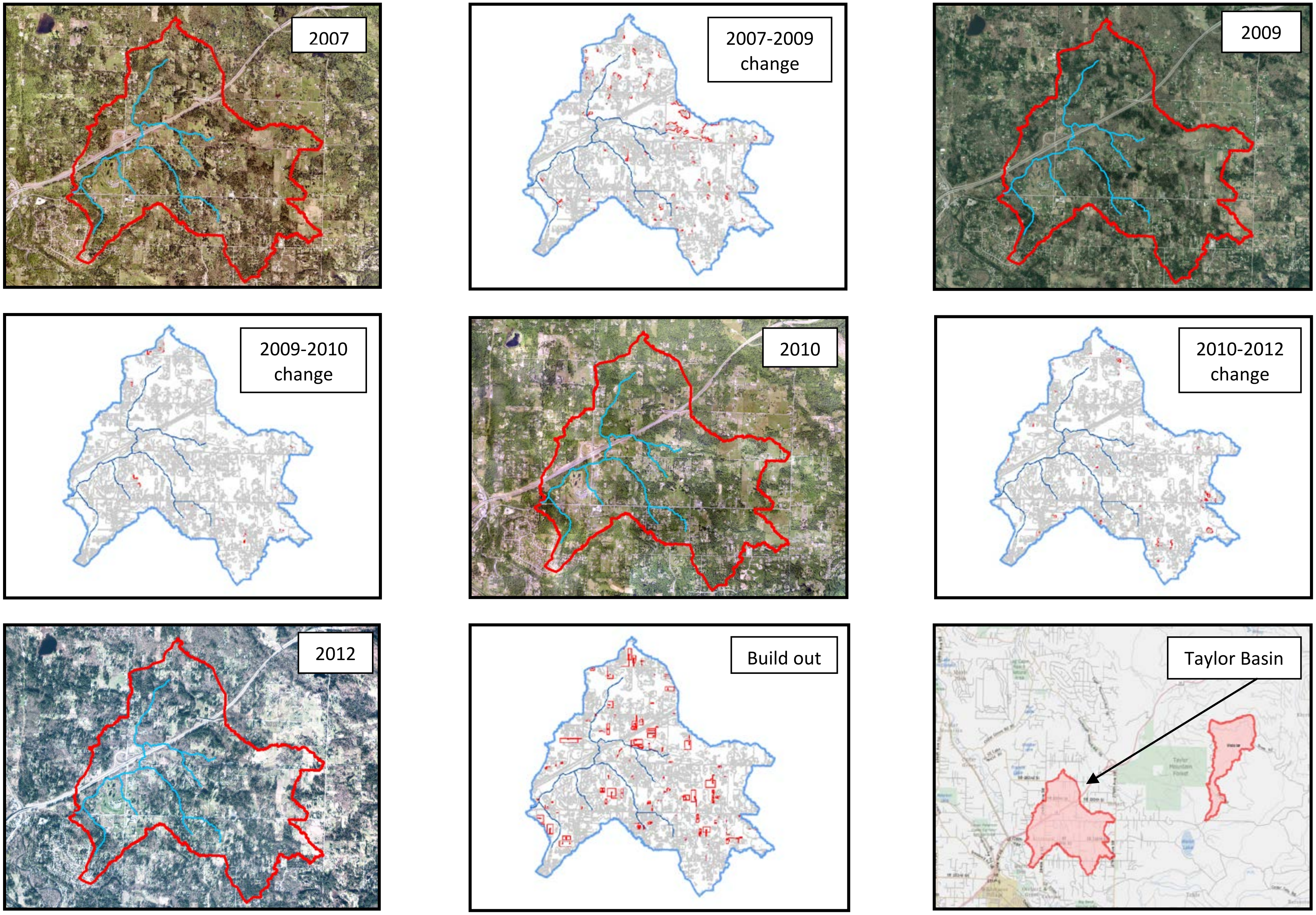








Taylor Landcover



Appendix F.

Lucchetti et al. 2013

Monitoring physical changes in stream channels with salt tracers

2013

Gino Lucchetti[1], Joshua J. Latterell, Raymond Timm, Chris Gregersen

King County Department of Natural Resources and Parks

Water and Land Resources Division

Suite 600, 201 South Jackson Street

Seattle, WA 98144 USA

King County

Department of Natural Resources and Parks
Water and Land Resources Division
King Street Center, KSC-NR-0600
201 South Jackson Street, Suite 600
Seattle, WA 98104

Alternate Formats Available

206-296-6519 TTY Relay: 711

[1] Corresponding author: Gino Lucchetti, Gino.Lucchett@kingcounty.gov, Office: (206) 477-4759, FAX: (206) 296-0192.

SUMMARY/ABSTRACT

1. The purpose of this study was to test the feasibility of using salt tracers for detecting in-stream effects of upstream development, or 'urban stream syndrome'. The study is one part of a larger study to test the effectiveness of comprehensive land use regulations designed to protect aquatic resources.
2. The study was conducted over five years in nine small streams with mostly undeveloped, rural watersheds in King County, Washington. One small urban stream was added to determine whether wood placement causes a measurable increase in flow resistance.
3. A salt solution was added to the top of the study reach, as a slug, and conductivity meters were used to measure the time required to reach peak conductivity at multiple points downstream. Reach-averaged advective velocity (U) was then calculated, as a proxy for total flow resistance (f_t).
4. Reaches were approximately 200 m long, divided in half, and instrumented with loggers recording conductivity at 5- or 15-minute intervals.
5. An estimate of U_{MAD} – the mean advective velocity at mean annual discharge – was derived from rating curves based on multiple salt releases in each stream and year. Same-day replicate surveys were used to estimate measurement precision.
6. A total of 306 independent salt runs were measured between 2008 and 2013.
7. Tracer surveys were very precise. Median measurement error was 2%, and median coefficient of variation (c_v) was 1.5% among sites and replicates.
8. Despite being highly precise, values for U_{MAD} over five years showed no coherence in the direction and magnitude of change between sites. This study established a baseline; more studies at a later date are needed to test for development impacts.
9. Wood placement reduced U_{MAD} in one of three reaches; the effects of wood were only evident where it was in contact with and affecting the shape of the streambed.
10. We conclude that using salt to measure changes in U_{MAD} is advantageous because it offers greater precision than traditional methods, provided that rating curves are derived for each stream and year of the study.
11. The method's greatest strength is also a limitation: U_{MAD} can detect and integrate many physical changes, but is diagnostic of none. As a result, U_{MAD} is best seen as a complement to traditional surveys, not a replacement.

Key words: Stream survey, development, Washington, solute dynamics, hydraulics, geomorphology, habitat, King County, development, stressor-response

INTRODUCTION

Stream monitoring is needed to detect, understand, and avoid human development impacts on streams. Monitoring has been instrumental in detecting ecological damage resulting from harmful development practices (Walsh et al. 2009). In some cases, these findings help to motivate major changes in land use regulations, as in the case of King County – a local government including metropolitan Seattle, its exurbs, and rural areas. Presumably, improved regulations are more effective at protecting streams from development impacts, though little evidence exists to validate this assumption.

King County updated its land use regulations in 2005, enacting major changes to more effectively protect aquatic resources in developing rural areas. This update was required by Washington State's Growth Management Act (GMA). Commonly referred to as the Critical Areas Ordinance (CAO), these land use regulations include three ordinances – Critical Areas, Clearing and Grading, and Stormwater. The CAO included significant environmental protections, such as larger buffers around streams and stronger clearing limits and stormwater management requirements. The ordinances work in combination with restoration, protection and stewardship efforts to protect the environment from potential adverse effects of development, in its various forms. The CAO was based on a synthesis of best available science (BAS; see http://www.metrokc.gov/ddes/cao/), yet it was controversial and repeatedly contested in public and legal debates. These debates continue and uncertainty persists as to whether the new regulations are necessary or sufficient to protect aquatic resources and beneficial uses.

Implementation of the CAO in 2005 created a unique opportunity to determine whether stringent land use regulations can prevent the onset of 'urban stream syndrome' (Meyer et al. 2005, Walsh et al. 2005, Walsh et al. 2009) in rural watersheds not yet altered by development. Unlike 'urban' watersheds, it may yet be feasible to protect the valuable aquatic resources in rural watersheds and processes that sustain them, which are thought to be at highest risk from land development pressures. Accordingly, King County partnered with the U.S. Environmental Protection Agency in 2008 to begin a 5-year study to determine whether the new land use regulations (i.e., CAO) were sufficient to prevent the onset of 'urban stream syndrome' (Meyer et al. 2005, Walsh et al. 2005, Walsh et al. 2009) – as an indication of effectiveness of the regulations at the watershed scale.

One of the near-universal symptoms of urban stream syndrome is channel simplification (Walsh et al. 2009). Development or urbanization tends to increase runoff efficiency, and the frequency of small flood events (DeGasperi et al. 2009). Consequently, when small flood events become more frequent, stream channels consistently widen, pools get deeper, scour increases, and channel complexity declines (Table 1 (Walsh et al. 2009)). Channel simplification reduces habitat quality, and simplified channels discharge more efficiently, contributing to flashiness in downstream areas. Accordingly, streams that are effectively protected by land use regulations should not exhibit symptoms of urban stream syndrome, including evidence of channel simplification.

Our goal was to determine if upstream development was associated with channel simplification. If so, it would be evidence that the CAO was not effectively insulating streams from development impacts, and would potentially need to be revised. Our broader study measured changes in flow regime, conductivity, and benthic invertebrates, all of which have been shown to be responsive to land use driven change (Walsh et al. 2009). But in this paper, we focused on field-testing the utility of solute tracers. Our rationale was that tracers have the potential to detect small changes in channel complexity and serve as an 'early-warning system' for urban stream syndrome.

In this study, we first field-test a solute tracer method to measure five years of change in the hydraulic complexity of six stream channels draining developing rural watersheds and in three streams draining forested watersheds. The six 'treatment' watersheds had a common history of clear-cut logging, followed by agricultural land uses. Existing development levels vary among the treatment watersheds, but each remains relatively undeveloped, as compared to urban watersheds. However, each watershed is expected to experience substantial future development. These attributes made them useful places for tracking development-related changes. Changes in the treatment watersheds were compared to changes in three forested 'reference' watersheds that are protected as park lands or municipal watersheds. The reference watersheds had also been historically logged, as were virtually all of the lowland forests in Puget Sound. However, there was little or no existing development in these watersheds, and they were each mostly covered with 2nd-growth native forests.

In addition to measuring changes in developing and reference watersheds, we also use tracers to measure changes in stream complexity as the result of large wood placement in a low-order, urban stream with moderate gradient. Large wood is commonly added to streams to restore habitat for salmonids (e.g., by trapping sediment and scouring pools). The aim of the second part of the study was to better understand the sensitivity of tracer methods to an experimental manipulation that increased flow resistance. In both applications – detecting development impacts and restoration benefits – tracer methods could reduce measurement error and eliminate observer biases that impact other widely-used channel survey techniques (Woodsmith et al. 2005, Whitacre et al. 2007, Roper et al. 2010).

Measurement error and observer bias are both persistent problems in stream surveys (Kaufmann et al. 1999). For example, there is often little consistency among trained observers classifying channel units (Roper and Scarnecchia 1995), owing in part to ambiguous or subjectively defined units. It is typical for different observers measuring the same stream to produce estimates with confidence intervals ranging between 26-43% of the mean (Wang et al. 1996). Even when using the same protocols, between-crew survey precision of selected indicators of channel condition may range from 4% to 46% among variables (medians) (Woodsmith et al. 2005). Comparisons of stream monitoring protocols demonstrate a need for greater precision, consistency, broader transferability, and responsiveness to human activities in stream (Roper et al. 2010).

One of the problems created by measurement error and observer bias is an increase in the number of years or sites that must be monitored to reliably detect trends or change (Roper et al. 2002, Roni et al. 2005). For example, the total sample size required to detect a 20% difference in commonly used indicators (gradient, substrate size,% fines) with relatively weak confidence (1-β of 0.10) and was over 400 (Roper et al. 2002). Consequently, the cost-effectiveness of the monitoring effort is reduced (i.e., knowledge per unit effort) and the lead time for responding to environmental problems is reduced. This is a significant problem for local governments charged with monitoring and managing aquatic resources.

The problem stems, in part, from economic recessions of the past decade that have strained the budgets of local governments nationwide. This threatens funding for stream monitoring. As a result, monitoring is likely to remain underfunded even though it is widely recognized as indispensable (Bernhardt et al. 2007). In the absence of more reliable funding, it will be critical for managers to use survey techniques that are cost-effective, precise, and consistent. We believe tracer methods hold significant promise, in that they may meet these criteria.

In this study, we test the feasibility of using a solution of plain table salt (NaCl) as a tracer for monitoring the hydraulic responses to physical changes in small streams. The salt causes a change in the conductivity of the stream water, which can be precisely measured using conductivity loggers. The reach-averaged velocity of the stream can be estimated by tracking how long it takes for the conductivity measurements to peak at the downstream end of the reach. This is a useful measurement because flow velocity is a fundamental channel property, affected by discharge, slope, and channel properties. It is a key determinant of stream power, and the friction coefficient. Most importantly, it determines habitat suitability and the community composition of stream organisms (Hart and Finelli 1999). Velocity is also one of the most sensitive – albeit, variable – properties of flow, because it depends on and integrates so many factors (Knighton 1998). These properties make flow velocity a meaningful and important indicator of environmental change, provided that it can be accurately measured.

Specifically, the use of salt tracers allows channel simplification – a pervasive symptom of urban stream syndrome – to be directly quantified as total flow resistance as the Darcy-Weisbach friction factor (f_t; Eq. 1).

Eq. 1: $f_t = (8gRS_e)/U^2$ or $f_t = (8gQS_s)/WU^3$

where g is 9.81 m s^{-2}; R is mean hydraulic radius (i.e., cross-sectional area/wetted perimeter); S_e is slope (m/m) or reach-mean energy gradient; S_s is the reach-mean water surface slope; and U is mean advective velocity in reach (m/s). In low-order streams of the Pacific Northwest, the key components of flow resistance are, in order of importance (Curran and Wohl 2003):

1. Spill resistance: Occurs at steps in streambed, energy is dissipated through turbulence. Associated with flow acceleration and deceleration.
2. Grain resistance: Shear generated by grains along boundary
3. Form resistance: Results from drag forces or pressure differences between the upstream and downstream sides of obstacles (e.g., wood, boulders, bars)

Total flow resistance f_t primarily reflects reach-averaged flow velocities U, because computed values of f_t are very sensitive to variation in U. Accordingly, in low-order streams of the Pacific Northwest, U is well-correlated with f_t. Most of the information content or 'signal' in f_t is contained in U; in other words, U can be used as a proxy for detecting changes in total flow resistance (Eq. 2). If inter-sample variation in discharge and slope is controlled or standardized, U objectively characterizes channel properties.

Eq. 2. $f_t = 2.0787U^{1.554}$, r^2=0.84 (Curran and Wohl 2003)

A tracer, like salt, can be used to estimate U in simple fashion. Releasing a tracer, as a slug, near the top of a study reach permits measurement of the nominal transport time (T_n); the travel time between the top and bottom, as indicated by time of peak on tracer concentration or specific conductance curves. In keeping with convention (Curran and Wohl 2003), we used the peak travel time instead of centroid, because it is more practical and considered to be more appropriate for evaluating flow resistance (as opposed to subsurface flows). Reach-averaged velocity (U) can then be estimated by dividing the T_n by the reach length, measured continuously along the channel thalweg.

STUDY GOALS

This study had two primary and interrelated goals:

1. To field test salt tracers as a method for detecting in-stream effects of upstream development. Accordingly, we used plain table salt as a tracer to measure reach-averaged advective velocity for five years in nine small streams draining rural watersheds.
2. To better understand the sensitivity of a salt tracer to an experimental stream channel manipulation (i.e., wood placement) that increased flow resistance in a simplified urban stream channel.

HYPOTHESES

(H1) We hypothesized that a tracer-based indicator of channel complexity (U) is more cost-effective and more precise than indicators from other protocols in widespread use in Washington State (e.g., EMAP). Our rationale was that tracers would require less time for sampling and analysis, and replicate samples would be more precise, evidenced by lower coefficients of variation among replicates. Our logic was that tracers are more objective than stream channel surveys, and remove the observer bias and transcription errors. If so, tracers could potentially detect development impacts to streams faster than other techniques.

(H2) We hypothesized that U is sensitive to increases in channel roughness or flow-obstructions. Specifically, U should decrease after wood placement, to the extent that wood increases spill resistance(Curran and Wohl 2003). If so, this indicates that changes in U may indicate the loss or addition of wood interacting with flows in upstream reaches.

STUDY AREA

The study area occurs in King County, Washington, an area of common geologic history, flora and fauna, and human uses. The area has been altered by historic logging (mid-1800s to mid-1900s) and agriculture, and contains the largest cities and most densely populated metropolitan areas in the Puget Sound region. Even so, a majority (66%) of land in King County jurisdiction remains forested and is dominated by rural and forest production land uses, excluding federal wilderness areas. Winters are warm and wet, averaging 35-57 inches of precipitation, annually. Surficial geology is mostly glacial till and outwash, or basal till with some volcanic ash. Soils are commonly gravels, sands, and loams. Annual temperatures average approximately 10^0 C.

Six treatment and three reference study streams were chosen deliberately, instead of drawn at random from a larger population (Figure 1; Table 1). We sought low-elevation streams draining rural watersheds, with a high potential for new development. Development potential was assumed to be inversely related to the number of parcels developed after 1989, and positively related to the number of parcels with improved value of less than 20% of the total value. In other words, these watersheds contained relatively large numbers of undeveloped or only partially developed parcels. We also required the selected watersheds to have good access; many did not, as they are in private ownership. The reference sites are protected, municipal watersheds with no development and minimal management. They are assumed to represent a potential past and future condition of the treatment watersheds, in the absence of rural development.

These nine watersheds are 2nd- and 3rd-order alluvial streams originating on low-gradient upland plateaus, dropping across steep side-slopes to low-gradient base levels set by a major river, lake, or Puget Sound. Upland and riparian forests consist of second-growth conifers, mainly Douglas fir (*Pseudotsuga menzeisii*), western hemlock (*Tsuga heterophylla*), and western red cedar (*Thuja plicata*) and, to a lesser extent, deciduous trees, mainly big leaf maple (*Acer macrophyllum*) and vine maple (*Acer circinatum*), red alder (*Alnus rubra*) and black cottonwoods (*Populus trichocarpa balsamifera*). Hydrology is rain-dominated, with naturally flashy flows during winter and low to intermittent flows during summer. All stream reaches in the study have perennial flows and are fish-bearing.

Figure 1. Study sites.

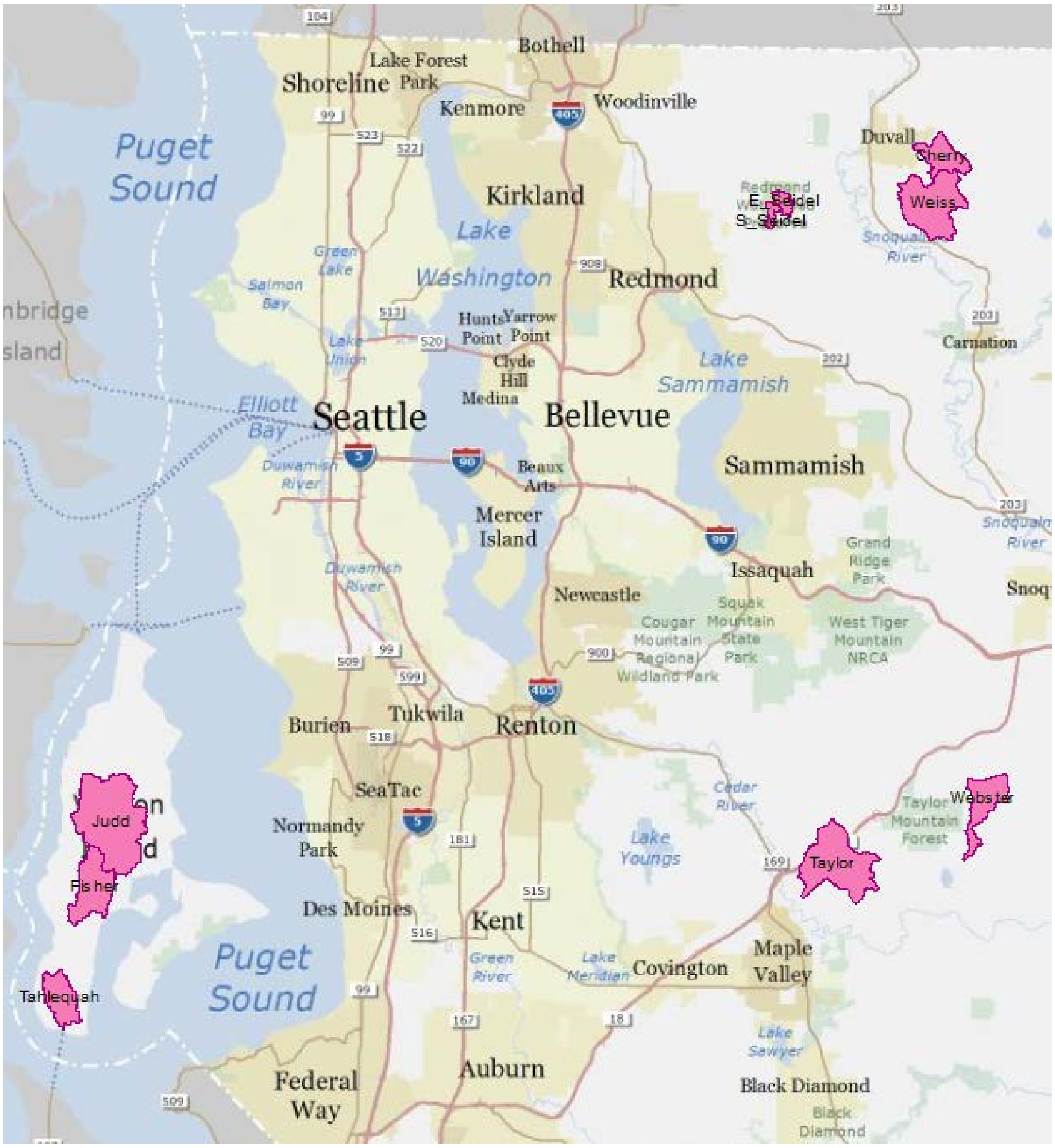


Table 1. Study streams.

Treatment	Stream	Elev. (ft.)	MAD (cfs)	Drainage Area (ha)	Study reach	2009 Reach length (m)	2009 Channel width (m)	Channel slope (m/m)
Development	Cherry	100-575	1.45	300	Upper	82	3.1	0.052
					Lower	98	3.7	0.064
	Taylor	405-735	3.20	936	Upper	94	5.0	0.009
					Lower	108	5.2	0.011
	Judd	15-470	5.82	300	Upper	86	7.1	
					Lower	100	5.9	
	Fisher	20-440	1.58	512	Upper	86	3.8	0.021
					Lower	100	3.7	0.023
	Tahlequah	5-415	0.79	331	Upper	70	2.7	0.022
					Lower	62	3.1	0.024
	Weiss	65-575	6.23	764	Upper	90	3.9	0.011
					Lower	99	4.5	0.024
None (Reference)	East Seidel	360-580	0.40	75	Upper	92	1.9	0.068
					Lower	82	2.5	0.061
	S. Seidel	340-565	0.50	62	Upper	100	1.9	0.020
					Lower	110	2.4	0.020
	Webster	835-2600	9.43	397	Upper	131	8.4	0.053
					Lower	80	6.8	0.044
Wood placement	Des Moines *2012 meas.	0-300	7.8	1500	Control	75	7.0	0.03
					Reach 1	59	8.2	
					Reach 2	82	7.3	
					Reach 3	104	7.3	

The restoration project tested in this study took place in ravine reach (RM1.6 to RM 0.9) of Des Moines Creek, in the Cities of Des Moines and SeaTac, Washington (O'Rollins 1999). This creek drains an urbanized basin of approximately 5.8 square miles and empties directly into marine waters of Puget Sound at the mouth. The dominant discharge in this stream responsible for the majority of sediment transport is in the range of 80 to 150 cfs (2.3 to 4.2 m3/s) (Merit and Booth 1999). Prior studies identified problems with flashy flows and lack of instream structure, so a series of restoration project was performed to detain and bypass flows, revegetate the riparian areas and add large wood to the park, wetland and ravine reaches. Woody debris was added in four different efforts several years. The woody debris installation efforts analyzed by the tracer study are located in the ravine and wetland reaches. The restoration project located in the ravine reach consisted of adding multiple, channel-spanning logs to the channel. Logs were clustered together and attached to others in the cluster by heavy chain, for stability. The project was completed in summer of 2010.

REACH LAYOUT

Study reaches were located in places where landowners granted access, and near locations suitable for stream gage installation. Study reaches were subdivided in half to form an upper and lower reach. One reason for using two adjacent reaches is to determine whether they show similar types and magnitude of change. Another reason is to determine whether changes are scale-dependent. For example, a change may be evident in one reach, but could be masked when the two reaches are combined for analysis.

A Before-After-Control-Impact (BACI) design was used to test the effect of wood placement on stream velocity (U) in the 'ravine' reach of Des Moines Creek. A control reach was established immediately upstream from the first wood structure. Downstream of the control reach, the stream was divided into three contiguous 'treatment reaches' that contained one log structure each. This layout was intended to allow independent evaluation of the effects of each structure on U .

APPROACH

Our approach was to 1) estimate U with tracers in each stream over a range of flows to estimate the relationship between discharge and U, and 2) to compare changes in U at median daily discharge across years in order to understand interannual variability in the metric between years, and 3) to compare the precision of tracers with that of more commonly-used stream surveys.

TRACER METHODS

Reach-averaged velocity (U) was estimated by releasing a slug of salt solution (tracer, hereafter), upstream from each study reach and measuring the resulting changes in stream conductivity at the top and bottom of each reach. The tracer was a solution of approximately 200 g of plain, non-iodized table salt (NaCl) dissolved in 4L of either tap water or stream water (*M. Roberts, pers. comm.*). The aim was to measure T_n, or nominal transport time, which is simply the time required for the peak conductivity to be reached at each sonde.

Two sondes were used; a YSI 600XLM with 1 μS resolution and a YSI Professional Plus (Model 6050000) with a 0.1 μS resolution. Sondes were periodically calibrated prior to field sampling with a standard solution; typically, once every two weeks during the sampling season. Sonde clocks were synchronized with atomic watches. The time of installation in the stream was noted (to 1 sec) to verify and correct sonde times, if necessary. Background conductivity was recorded prior to releasing the salt solution and sondes were removed after stream conductivity returned to near background levels, meaning the salt solution had passed. Sondes logged conductivity of stream water at either five or 15-second intervals; shorter intervals were needed during high flow conditions, when the salt slug is moving quickly, and consequently, conductivity levels change rapidly.

Once sondes were installed, the tracer was released at the upstream boundary of the upper reach, and the time of release was recorded (to 1 sec). A 25-m mixing reach was used at the restoration sites, but no mixing reaches were used at the development and reference sites. Tracer dosage – or volume of salt solution added to the stream – was typically four liters or less, but varied according to a visual estimate of the stream discharge at the time of the survey. After the first year of surveys, tracer dosages were optimized by a combination of trial-and-error, and by evaluating the relationships between peak size, solute volume, and discharge during initial surveys. Our goal was to release enough tracer solution to generate a peak of 50 to 100 μS above background levels. The reason was that the sondes are accurate to approximately 1 μS (though the resolution may be greater), so the resulting peaks would have 50 to 100 distinct values. Having many distinct values affords greater precision in determining the time at which the tracer is first detected, peaks, and then flushes out of the study reach. Small peaks lead to problems with a 'stair-step' conductivity curve, containing many consecutive measurements of the same value. Stair-steps on the curves can make it difficult to precisely identify the timing of peak tracer concentration.

Multiple tracer releases were performed on each stream in each year, in order to estimate a flow-standardized value for T_n or U. The values were flow-standardized because they are highly sensitive to variation in discharge, but it is virtually impossible to survey the same discharge level within or between years. Instead, multiple surveys established a univariate regression relationship between discharge and T_n for each stream in each year of our study. The relationship was used to estimate the value of T_n at the mean annual discharge, as well as the 25th (below normal) and the 75th (above normal) percentiles. We performed tracer surveys across a wide range of flows in each stream and year to precisely estimate the relationship between discharge and T_n. This approach allows T_n to be compared among streams and years. Permanent, continuous stream gages were previously installed on all study streams, providing continuous records of stream discharge at each site.

Same-day replicate tracer surveys were performed in 2009 to estimate the precision of tracer method. Replicates were completed at least once at each site; after the first tracer slug passed the downstream sonde, another slug was released. Comparing U between replicates quantifies the minimum level of measurement error in the method; stream morphology is identical between surveys, stream discharge is virtually unchanged, and observer bias is minimal or nonexistent. Remaining variation between samples must be attributed to either random variation in stream hydraulics, imprecise clock synchronization, or some unknown observer bias. If precision is high between replicates, it follows that observer bias is similarly low or nonexistent. If so, then the changes between years, at a constant flow, must be indicative of a physical change in reach hydraulics, not observer errors or bias.

THALWEG SURVEYS

Replicate surveys of thalweg length were performed in 2010 by three independent field crews on Cherry Creek and Taylor Creek. The purpose was to estimate the precision of the physical stream survey protocols and to estimate errors from observer biases. Prior studies find that reach length increases up to 10% when measured in fine detail (many sampling points; Curran and Wohl 2003). Each team was led by one 'expert' or highly-trained leader. The same protocols were used by each team to measure the length and depth of the channel thalweg (and other variables not reported here).

TRACER DATA ANALYSIS

DATA ENTRY

Data was manually entered in the first three years of the study. In the fourth year, a database was created to store and efficiently process tracer data. From that time forward, data was uploaded from sondes into the database, which was used for QA/QC, for automatically calculating metrics of interest, and for exporting summary tables.

DATA PROCESSING

The reach-averaged velocity *U* was calculated from each tracer release, based on changes in the conductivity of stream water as the solute passes by the point of observation (Table 2). Each tracer release generated two conductivity curves; an upper reach, and a combined reach curve. The peak travel time T_n was divided by the reach length to estimate *U*; the reach-averaged velocity. Metrics were calculated separately for the upper reach and the combined reach. Values for the lower reach were estimated as the difference between the upper and combined reach.

Table 2. Definitions for tracer metrics. Visual interpretation was used to validate and, if necessary, correct the automatic assessments performed in the database.

Metric	Symbol	Definition	Assessment Rule
Reach-averaged velocity	U	Tracer-based velocity in meters per minute; proxy for total flow resistance (Curran and Wohl 2003)	Divide T_n by the reach length, measured continuously along the channel thalweg.
Peak travel time	T_n	The time at which salt tracer reaches the maximum concentration (peak conductivity value) at the sonde (equivalent to nominal travel time; ref)	The maximum value in the conductivity readings. In case of ties (e.g., a prolonged peak), the first value is used as the peak.

In the six treatment and three reference sites, *U* was standardized to the mean annual daily discharge (MAD). Standardization was needed because metrics are flow-dependent and repeat samples can rarely be taken at the same flow level. A regression was fit between observed values of T_n and discharge, measured by stream gages at each site. The regression produces a 'tracer rating curve', which allows T_n at MAD to be estimated, and compared among years and streams. In most cases, the relationship between T_n and MAD was best described by a power function. The standardized value for T_n can then be divided by thalweg length for each study year to estimate the reach-averaged mean velocity at mean annual discharge, hereafter, U_{MAD} to denote reach-averaged velocity. In the same manner, we also estimated the value of *U* at the 25th and 75th percentile flows; U_{25} and U_{75}, respectively.

The relationship between discharge and T_n or *U* may also be useful for detecting changes in the channel. For example, year-over-year increases in the slope of the tracer rating curve may indicate the stream transports water more quickly, per unit increase in flow. Also, the coefficient of determination (r^2) may be inversely related to stream complexity. For example, tracer metrics may be strongly dependent on discharge, alone, in a simple channel. By contrast, discharge may be a weaker predictor of tracer metrics in a channel with complex features and substantial roughness, which would add variability to the tracer rating curve. Accordingly, the residuals from the regression relationships may be attributable and related to flow obstructions, and bed complexity, or other forms of roughness. In these ways, increases in the slope of the tracer rating curve, and in the coefficient of determination could be indicators of urban stream syndrome.

At Des Moines Creek, no standardization was necessary because treatment reaches had a paired control site sampled at the same discharge, on the same day. At this site, the difference in *U* between the impact and the control was calculated before and after the wood placement. Comparing the difference before and after restoration (i.e., the 'difference of the difference') measures the effect of restoration on *U* in the treatment reach, relative to changes in the control.

RESULTS

A total of 306 independent salt runs were successfully performed between 2008 and 2013, during surveys of the six treatment and three reference streams (Table 3). Eighteen same-day replicate tracer surveys were performed. One person could sample two or three streams per day (0.3 – 0.5 work days per sampling event).

Table 3. Number of tracer runs in which peak travel time (T_n) was recorded for both the upper and combined reach.

Site	2008	2009	2010	2011	2012	**Subtotal**
Cherry	7	4	5	8	12	**36**
Fisher	6	6	6	9	8	**35**
Judd	6	7	9	9	8	**39**
Tahlequah	5	5	7	9	8	**34**
Taylor	5	6	6	9	9	**35**
Weiss	6	8	6	8	8	**36**
E Seidel	6	5	8	7	7	**33**
S Seidel	2	3	7	8	6	**26**
Webster	4	6	7	7	8	**32**
Subtotal	**47**	**50**	**61**	**74**	**74**	**306**

Runs were considered entirely successful if both sondes recorded T_n (Table 3), or partly successful if only the lower sonde recorded T_n (Table 4). If T_n was recorded at both sondes, *U* could be estimated for the lower, upper, and combined reaches. If T_n was recorded at the lower sonde, *U* could be calculated for the combined reach. Runs in which only the upper sonde recorded T_n could only be used to estimate *U* for the upper reach, not the lower or combined reach.

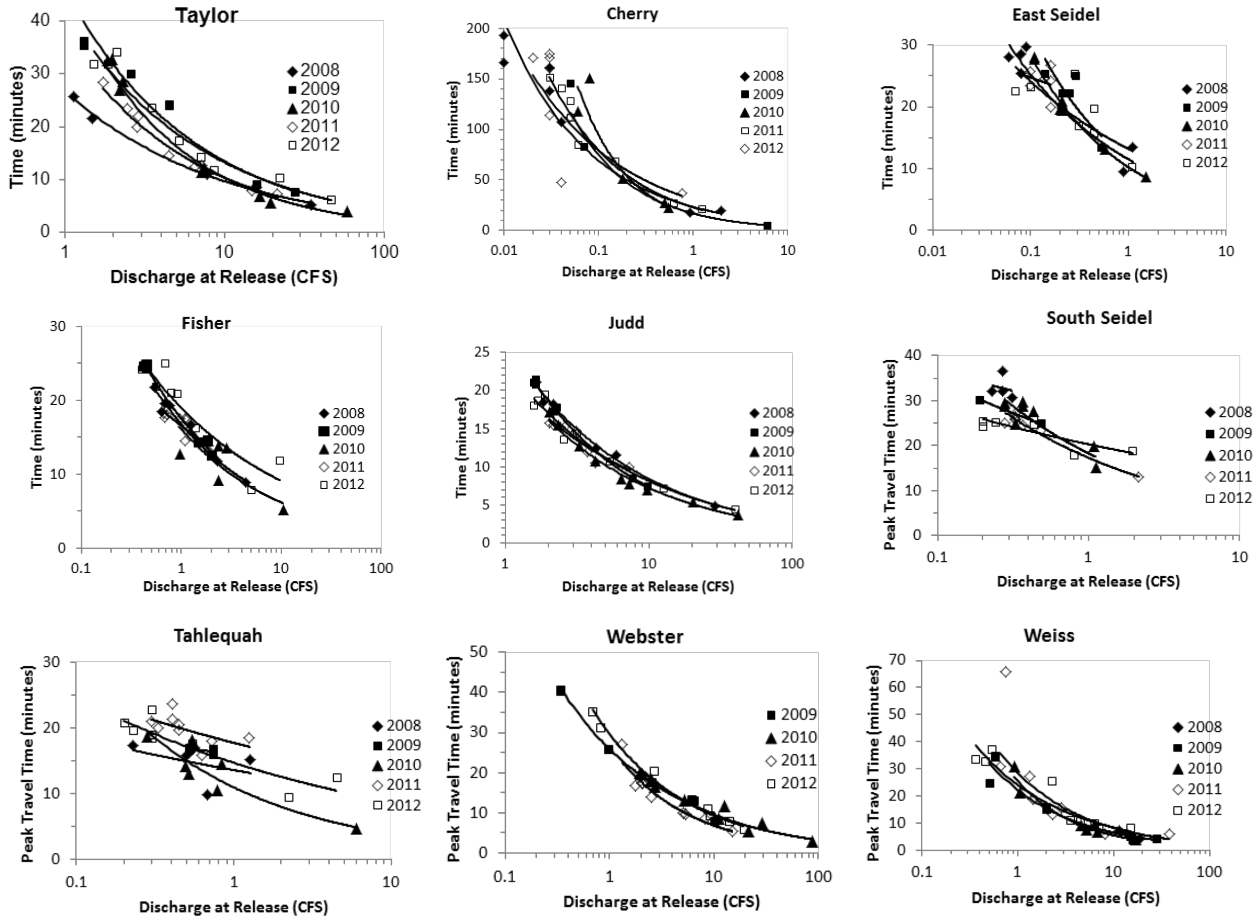
Table 4. Number of tracer runs in peak travel time T_n was recorded for at least the combined reach.

Site	2008	2009	2010	2011	2012	Subtotal
Cherry	7	4	5	8	12	36
Fisher	7	6	6	9	8	36
Judd	6	7	9	9	8	39
Tahlequah	5	4	7	9	8	33
Taylor	6	7	7	9	9	38
Weiss	7	8	6	8	9	38
E Seidel	7	5	8	7	7	34
S Seidel	4	2	8	8	7	29
Webster	5	6	8	9	9	37
Subtotal	54	49	64	76	77	320

RATING CURVES

Either four or five tracer rating curves – one per survey year – were fit to observed values for T_n in each stream, using a power function in every case (Figure 2). The regression model for each curve (Table 5) was used to estimate a single value for T_n at mean annual discharge (Table 9), which was converted to U_{MAD} by dividing by the thalweg length for that year (Table 7).

Figure 2. Rating curves showing the relationship between the travel time of the conductivity peak (y) and discharge (x) through the combined reach (upper and lower) for each stream and year in the study. Note that the X-axis is represented on a logarithmic scale.



The precision of the relationship between T_n (and therefore U_{MAD}) and discharge varied among stream and years, but generally increased with stream size (Figure 2, Table 5). Travel time was most precisely related to flow in Judd, Weiss, and Webster. In each instance, the relationship was also very consistent among years. In contrast, the relationship was relatively imprecise in Tahlequah and South Seidel Creeks. In these instances, and in other measurements taken during extremely low flows, the precision of the relationship was reduced by inaccuracies in the discharge estimate. For example, virtually all of the measurements in Tahlequah and S. Seidel Creeks were taken when flows were less than one cfs. In Cherry Creek, roughly half of the surveys were done at flows in the range of 0.01 to 0.1 cfs, which are very difficult to precisely estimate.

Table 5. Relationships between the travel time of the conductivity peak (y) and discharge (x) through the combined reach (upper and lower) for each stream and year in the study. Sample size (n) and coefficient of determination (r^2) are given for each stream and year.

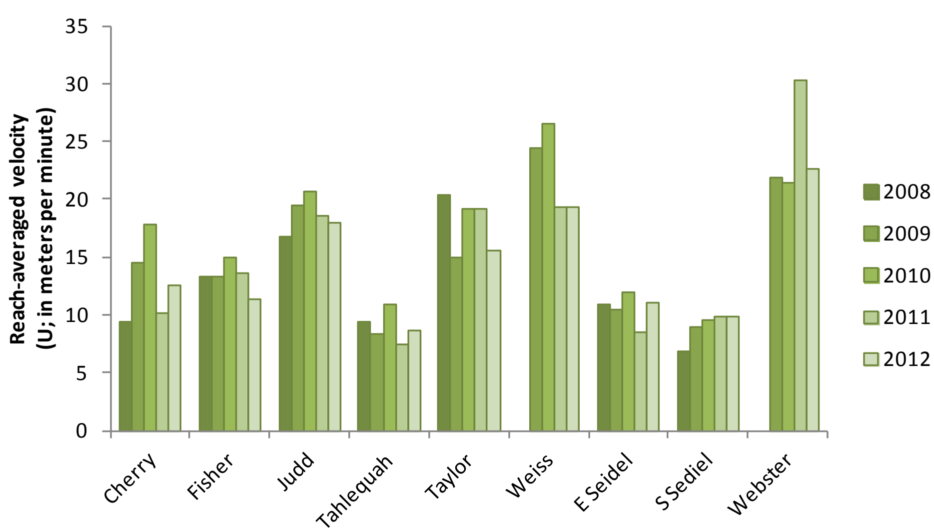
Site	Equation (y)					n					r^2				
	2008	2009	2010	2011	2012	2008	2009	2010	2011	2012	2008	2009	2010	2011	2012
Taylor	$26.76x^{-0.441}$	$46.032x^{-0.534}$	$46.36x^{-0.65}$	$36.928x^{-0.552}$	$42.093x^{-0.502}$	5	7	9	9	9	0.99	0.96	0.98	0.98	0.96
Cherry	$23.229x^{-0.474}$	$16.751x^{-0.647}$	$14.514x^{-0.813}$	$22.855x^{-0.54}$	$18.412x^{-0.557}$	7	4	5	8	8	0.96	0.99	0.95	0.98	0.91
E Seidel	$11.497x^{-0.342}$	$11.428x^{-0.453}$	$10.21x^{-0.444}$	$19.74x^{-0.098}$	$13.197x^{-0.262}$	6	5	8	7	7	0.90	0.72	0.99	0.09	0.62
Fisher	$16.768x^{-0.412}$	$17.526x^{-0.418}$	$16.098x^{-0.416}$	$16.548x^{-0.276}$	$18.979x^{-0.32}$	7	6	6	9	8	0.97	0.98	0.67	0.66	0.79
Judd	$26.399x^{-0.496}$	$27.808x^{-0.577}$	$22.719x^{-0.498}$	$21.652x^{-0.411}$	$23.297x^{-0.457}$	6	7	9	9	8	1.00	1.00	0.99	0.86	0.98
S Seidel	$28.239x^{-0.113}$	$21.389x^{-0.209}$	$18.281x^{-0.394}$	$17.269x^{-0.361}$	$20.273x^{-0.151}$	4	2	8	8	7	0.03	1.00	0.83	0.97	0.65
Tahlequah	$13.558x^{-0.137}$	$15.969x^{-0.083}$	$10.937x^{-0.456}$	$17.717x^{-0.151}$	$14.675x^{-0.227}$	5	4	7	9	8	0.11	0.69	0.90	0.57	0.80
Webster	No data	$25.931x^{-0.425}$	$28.382x^{-0.473}$	$26.741x^{-0.592}$	$29.696x^{-0.503}$		6	10	9	9		0.97	0.93	0.96	0.99
Weiss	$74.077x^{-0.953}$	$22.204x^{-0.544}$	$25.505x^{-0.676}$	$28.443x^{-0.541}$	$24.033x^{-0.463}$	2	8	7	8	9	1.00	0.97	0.97	0.79	0.89

When averaged across all years within each stream, the reach-averaged velocities (U_{MAD}) ranged from 9 meters per minute in Tahlequah and S. Seidel Creeks, to 22 meters per minute in Weiss, followed closely by Webster Creek at 21 meters per minute (combined reach only; Table 6). There did not appear to be closely synchronized changes in U_{MAD} or U_{25} across all the streams, during the period of observation (Figure 3). Multi-year average values for U_{25} ranged from a low of 2 meters per minute in Cherry to a high of 13 meters per minute in Webster Creek (Table 7). The ratio of U_{25} to U_{MAD} averaged 60% across streams and years – meaning the average velocity at the 25th percentile flow was only 60% of the average velocity at the mean annual discharge – but ranged widely among streams. For example, U_{25} was only 13% of the value of U_{MAD} in Cherry Creek, but 88% of U_{MAD} in S. Seidel Creek, illustrating the differences in the flow regime among the study streams.

Table 6. U_{MAD} in meters per minute for each of three study reaches at each site and year.

Reach	Site	2008	2009	2010	2011	2012
Combined	Cherry	9.3	14.5	17.8	10.1	12.5
	Fisher	13.4	13.3	14.9	13.6	11.3
	Judd	16.8	19.4	20.7	18.6	18.0
	Tahlequah	9.4	8.4	10.9	7.4	8.6
	Taylor	20.4	15.0	19.2	19.2	15.5
	Weiss		24.5	26.5	19.3	19.4
	E Seidel	11.0	10.5	12.0	8.5	11.1
	S Sediel	6.9	8.9	9.5	9.8	9.8
	Webster	9.3	21.8	21.4	30.3	22.7
Lower	Cherry	9.0	12.5	14.8	9.7	12.1
	Fisher	14.7	14.2	14.9	13.1	11.9
	Judd	17.0	18.3	19.4	17.2	16.6
	Tahlequah	10.5	7.5	12.8	6.2	8.3
	Taylor	27.5	19.3	22.2	21.7	20.4
	Weiss		21.2	23.6	18.1	16.4
	E Seidel	10.9	10.3	11.2	6.6	11.1
	S Sediel	6.1	9.1	9.7	9.9	10.0
	Webster	9.0	20.5	19.0	28.0	21.5
Upper	Cherry	9.7	17.9	23.0	10.7	13.0
	Fisher	12.1	12.4	14.9	14.2	10.8
	Judd	16.5	20.9	22.5	20.4	20.0
	Tahlequah	8.5	9.4	9.5	8.5	8.8
	Taylor	15.7	12.0	16.6	17.0	12.3
	Weiss	27.7	30.0	30.9	21.0	24.8
	E Seidel	11.0	10.7	12.7	11.2	11.1
	S Sediel	8.0	8.8	9.3	9.6	9.6
	Webster	9.7	22.7	23.3	31.9	22.4

Figure 3. Estimated U_{MAD} (top) and U_{25} (bottom) in meters per minute for the combined reach at each site and year.



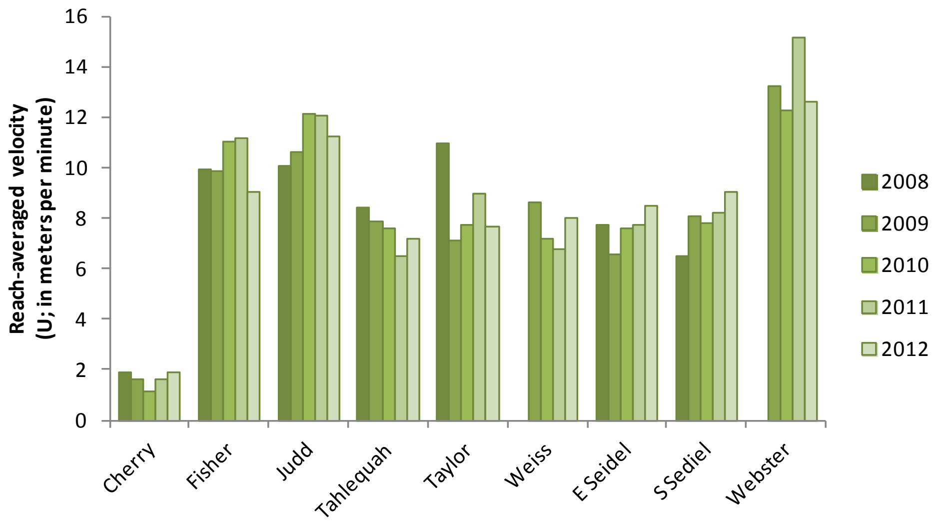


Table 7. U_{25} in meters per minute for each of three study reaches at each site and year.

Reach	Site	2008	2009	2010	2011	2012
Combined	Cherry	1.9	1.6	1.1	1.6	1.9
	Fisher	9.9	9.8	11.0	11.2	9.0
	Judd	10.1	10.6	12.1	12.0	11.3
	Tahlequah	8.4	7.8	7.6	6.5	7.2
	Taylor	10.9	7.1	7.7	9.0	7.6
	Weiss		8.6	7.2	6.8	8.0
	E Seidel	7.7	6.6	7.6	7.7	8.5
	S Seidel	6.5	8.1	7.8	8.2	9.1
	Webster		13.2	12.2	15.2	12.6
Lower	Cherry	1.8	1.4	1.1	1.6	1.8
	Fisher	11.4	10.5	10.8	10.9	9.5
	Judd	10.1	10.6	10.8	11.4	10.1
	Tahlequah	8.4	7.1	8.0	5.0	6.9
	Taylor	16.6	9.0	8.3	10.2	9.4
	Weiss		8.5	6.7	6.4	7.3
	E Seidel	7.6	6.8	7.3	7.4	9.0
	S Seidel	6.7	8.5	8.0	8.4	9.1
	Webster		12.8	11.1	14.4	12.2
Upper	Cherry	1.9	1.9	1.2	1.7	1.9
	Fisher	8.6	9.2	11.3	11.5	8.6
	Judd	10.0	10.6	14.0	12.9	13.0
	Tahlequah	8.4	8.6	7.2	8.2	7.4
	Taylor	7.9	5.8	7.2	8.0	6.3
	Weiss	11.7	8.8	7.8	7.4	9.0
	E Seidel	7.8	6.4	7.9	8.0	8.1
	S Seidel	6.3	7.7	7.6	8.1	9.0
	Webster		13.5	13.1	15.7	12.3

For the remainder of the analysis, We focused on U_{MAD} and U_{25}, but excluded U_{75}. This was warranted because the U_{MAD} was precisely related to U_{75}. (Table 8): correlation coefficients ranged from 0.986 to >0.999. However, correlations were more variable and generally weaker between U_{MAD} and the U_{25} (0.096-0.950; Table 8).

Table 8. Correlation coefficients between tracer peak velocity standardized to mean annual discharge versus the standardized velocities for the 75^{th} and 25^{th} percentile flows. All values are for the combined reach (upper and lower) at each study site across years (2008-2012).

Site	U_{MAD} vs U_{75}	U_{MAD} vs U_{25}
Cherry	0.999	-0.813
Fisher	0.998	0.846
Judd	0.986	0.653
Tahlequah	0.999	0.539
Taylor	0.990	0.791
Weiss	0.997	0.188
E Seidel	0.989	0.096
S Sediel	0.993	0.899
Webster	1.000	0.950

Trajectories or vectors of change in U_{MAD} over time indicate that in many of the study streams, the upper and lower reaches do not change in exactly the same way over time, in spite of experiences nearly identical flows, climatic variation, and upstream land use/land cover (Figure 4). In Figure 4, lines represent the magnitude and direction of change in the U_{MAD} – and surrogate for channel complexity in this analysis. Each point represents U_{MAD} for a given year. Vectors point forward in time, between two consecutive years of sampling. If the vector is close to the diagonal dashed line, complexity is similar between reaches. Vectors that point left indicate gains in complexity in the lower reach; those that point right indicate losses in the lower reach. Vectors that point upward indicate losses in complexity in the upper reach; those pointing down indicate gains in complexity in the lower reach. Vectors that point toward the upper right indicate less complexity in both reaches. Vectors pointing toward the lower left indicate gains in complexity in both reaches.

Accordingly, the vectors of change can be used to compare how the upper and lower reaches have changed with respect to each other, and to their initial condition at the beginning of the study. For example, in Cherry Creek, The both reaches became less complex (i.e., U_{MAD} increased) in the first two years of the study, but the magnitude of change was far greater in the upper reach, as indicated by the position of the vector (e.g., above the dotted line). Then in the third year, complexity increased in both reaches to a level that nearly matched that of the first year. In the final year of the study, both reaches became slightly less complex and at nearly equal rates.

Several stream experienced asynchronous changes in complexity in the upper and lower reaches, but became either more complex over time, or returned to similar levels as the outset of the study. For example, both reaches of Fisher Creek became more complex over time, as did Weiss and Taylor. Webster became less complex temporarily, but returned to its original levels in both reaches by the end of the study. In Tahlequah, the upper reach was virtually unchanged over time, compared to the lower reach, which ultimately returned to original levels by the end of the study. The only stream that demonstrated a consistent loss of complexity that was sustained over time was S. Seidel – one of the reference streams. The rate of change in complexity declined over time, however.

Figure 4. Trajectories or vectors of change in U_{MAD} over time as an indicator of complexity in the upper and lower reaches of the treatment and reference streams, from 2008 to 2012.

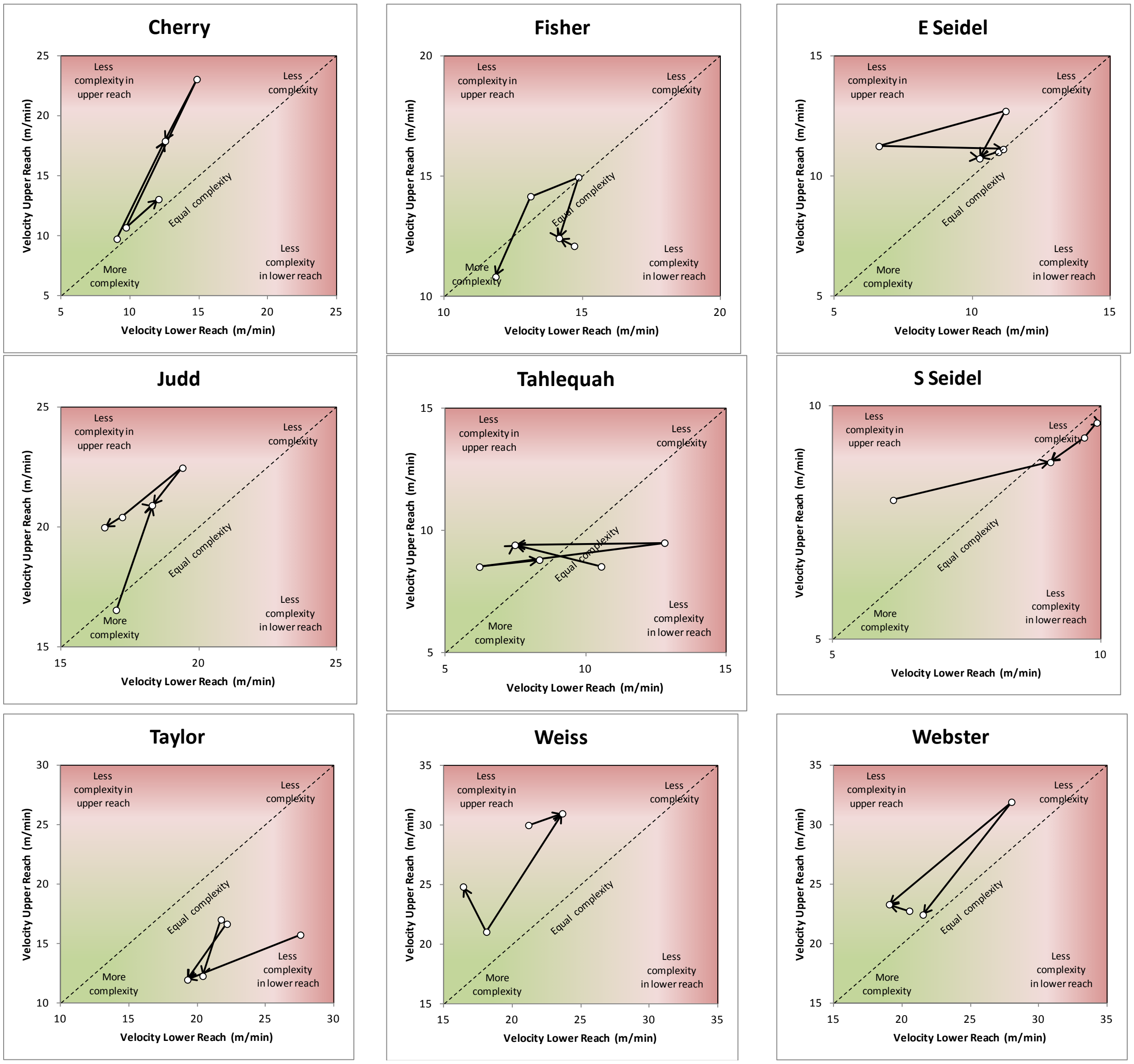


Table 9. Daily discharge statistics for study streams from the beginning of the record through WY 2012. Units are in cubic feet per second (cfs).

Type	Stream	25th percentile	Mean Annual Discharge	75th percentile
Treatment	Cherry	0.05	1.48	1.83
	Fisher	0.76	1.55	1.81
	Judd	2.07	5.87	6.08
	Tahlequah	0.34	0.75	0.8
	Taylor	2.35	9.41	12.03
	Weiss	0.91	6.23	7.76
Reference	E Seidel	0.14	0.39	0.49
	S Sediel	0.32	0.52	0.6
	Webster	2.63	8.45	10.55

PRECISION

Replicate tracer surveys demonstrated that peak travel time could be estimated precisely. The median measurement error, defined as the percent difference between T_n in the first and second replicates was 2% for the combined reaches, ranging from 0.0-5.6% among all sites and replicates (Table 10). We excluded one replicate from Weiss Creek, which was measured at flood stage; the measurement error (31%) was far higher than any other replicate. This sample was an outlier, in that peak travel time was only 5.25 minutes for the combined reach, which was unusually fast for the site and the study, in general.

The coefficient of variation (c_v) is a dimensionless statistic that indicates the precision of the measurements. Values ≤ 20 are preferred (Ramsey et al. 1992). In this study, c_v was calculated as the standard deviation of replicated measurements divided by the mean. The median coefficient of variation (c_v) for peak travel time in the combined reaches was 1.5%, ranging from 0.0 to 4.1% among all sites and replicates. These values indicate a high level of precision.

Table 10. Precision of peak travel time surveys based on replicate surveys of the combined reach.

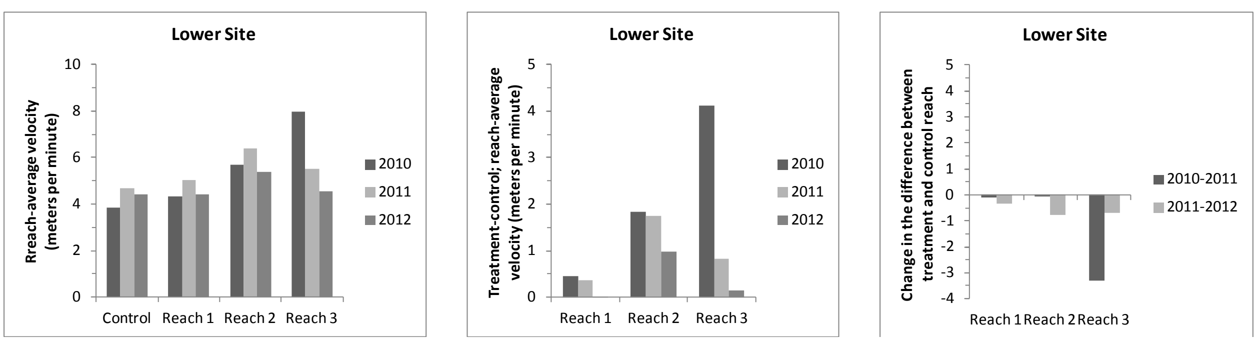
Type	Stream	Replicates	Measurement error range (%)	c_v range (%)
Treatment	Cherry	2	0.9-5.6	0.7-4.1
	Fisher	1	2.1	1.5
	Judd	2	1.2-2.9	0.8-2.0
	Tahlequah	2	1.5-4.7	1.0-3.2
	Taylor	2	1.0-2.1	0.7-1.5
	Weiss	3	2.2-3.2	1.5-2.3
Reference	E Seidel	1	0	0
	S Seidel	0	-	-
	Webster	2	0.6-1.9	0.4-1.3
All sites		15	0.0-5.6%	0.0-4.1%

Estimating U from T_n requires a measurements of thalweg length, which could potentially introduce measurement error and reduce precision. Three field crews independently measured the length of the thalweg in the combined reaches at Cherry and Taylor Creeks under the same flow conditions in 2011. Measurement error was calculated as the difference between the longest and shortest measurement among crews, divided by the average across crews. Calculated this way, the measurement error for thalweg length was 4.7% in Cherry Creek and 3.5% in Taylor Creek. The c_v of those measurements was 2.4% among teams, for Cherry Creek, and 1.9% for Taylor Creek – well within acceptable levels of precision.

Des Moines Creek Wood Placement Results

In Des Moines Creek, wood placement appears to have reduced U at the lower site, as evidenced by year-over-year declines (2010-2012) in the difference between U in the control and treatment reaches 1, 2, and 3 (Figure 5). The largest change, relative to the control, occurred in Reach 3, from 2010 to 2011 (Figure 6). The log cluster in this reach also appeared to have the most obvious effect on channel morphology, as evidenced by the formation of an upstream wedge of alluvial sediments. These interpretations are based on simple graphical interpretation; the statistical significance of changes cannot be tested without replicated surveys or multiple sites. The control reach is assumed to represent conditions that may have existed in the treatment reaches had no wood been added to them. Accordingly, if wood placement had no effect on stream velocities, then differences between the control and treatment reaches, where the wood was placed, would have remained similar to 2010 levels, or changed in a random or haphazard fashion.

Figure 5. Effects of wood placement on *U* in the Des Moines Creek ravine (lower) reach. Left panels show *U* in each study reach, by site. Middle panels show the difference in *U* between the treatment (wood added) and control (no wood added) reaches. Right panels show how the difference between the treatment and control has changed, relative to the control reach in the same year.



Surveys were performed over a narrow range of streamflows (1.00 to 1.40 cfs; King County gauging station 11d). Daily flows were estimated to be 1.16 cfs for 2010 (or 1.00 for 2010b), 1.40 cfs in 2011, and 1.10 in 2012. Nonetheless, some of the variability in velocities could be attributed to differences in flows at the time of each survey in different years. No flow-standardization (e.g., to mean annual discharge) is necessary in the Des Moines Creek study, however, because a control reach was surveyed for comparison.

Figure 6. Images of study reaches at Des Moines Creek from September 2012.

Table 11. *U* in meters per minute at the Des Moines Creek wood placement study by site, reach, and year. The upper site was surveyed twice in 2010; July 22 and July 27.

Reach	2010	2011	2012
Control	3.9	4.7	4.4
1	4.3	5.0	4.4
2	5.7	6.4	5.4
3	8.0	5.5	4.6

DISCUSSION

We conclude that using a salt solution to estimate changes in mean advective velocity (U) of stream channels is an advantageous way to characterize physical channel changes; this technique offers greater precision than other commonly-used field survey methods, provided that rating curves are derived for each stream and year of the study. This technique is advantageous in that it is unbiased and precise (i.e., median coefficient of variation (c_v) for peak travel time in the combined reaches was 1.5%, ranging from 0.0 to 4.1% among all sites and replicates.) compared to indicators evaluated in other studies (e.g., Kaufmann et al. 1999, Woodsmith et al. 2005). The added precision comes from the elimination of observer bias – incurred anytime 'professional judgment' is invoked – and from reducing measurement error (i.e., to the limits set by the conductivity logger). The method is easily taught to new personnel, provided that technicians regularly synchronize the clocks on all field equipment and they keep conductivity meters calibrated.

However, our results clearly demonstrate that estimates of U are strongly flow-dependent, which can reduce the cost-effectiveness of the technique. A common practice with other applications of salt tracers is to simply take a single measurement during summer low flow conditions. However, we found that flow-dependency is most problematic during summer low flow; small differences in flow, or small errors in flow measurement, can cause very large changes in velocity. This makes it hard to interpret year-over-year values based on estimates at a single discharge.

This problem is addressed by making rating curves based on multiple surveys at different flow levels, to derive a flow-standardized velocity (i.e., U_{MAD}). Technicians must visit streams multiple times in a sampling season, and must be able to precisely estimate flow at the time of the survey, either from a field measurement, or from a nearby stream gage. This requirement, in most cases, probably makes the cost of deriving U_{MAD} comparable to the cost of a traditional stream survey. Even so, U_{MAD} may be more cost-effective, if the highest priority is to detect change quickly. The higher precision of this method should allow earlier detection of impacts from upstream development, or from restoration activities, than if a real signal was masked or obscured by survey methods prone to higher observer bias and measurement error.

We recommend adding this metric to studies of land use effects or restoration projects on small streams, to complement existing techniques. This is justified, in part because it is very precise, but because the wood placement experiment in Des Moines Creek – and other examples from prior studies – indicates that U_{MAD} has the potential to detect relatively small changes in the physical attributes of streams. The addition of large wood clusters to the relatively high-gradient, wood-poor Lower Site in Des Moines Creek appears to have reduced U_{MAD}, most notably where the wood trapped sediment and created a sediment wedge. The wedge likely increased spill resistance (Curran and Wohl 2003), causing a reduction in U_{MAD}. We recommend using salt tracers in the context of a BACI study design in restoration applications. The use of a control reach avoids the need for a rating curve; U need not be standardized to a single flow level. Instead, the assessment of restoration effects can focus on the difference between the control and treatment or impact sites

We propose several recommendations to aid in future applications of the technique. First, we find that salt tracers are most useful in small streams and short distances. We suggest using study reaches that are generally <200 m in length. In longer reaches, the duration of peak tracer concentration may be prolonged, making it difficult to precisely measure the timing of the peak. We also suggest avoiding extreme low or high-flow conditions. At very low flows, it is difficult to precisely measure the timing of the peak, and the relative magnitude of the peak may be overly low, especially at the downstream end of the study reach. At very high flows, it is unlikely that the tracer is adequately mixed in the water column. This means the travel time is not very representative of the true hydraulic complexity of the channel. We also recommend using enough salt solution to raise background conductivity by at least 50-100 µS, to avoid producing a 'stair-stepped' conductivity curve. In the same fashion, we recommend logging measurements at 5-s intervals. In combination, these protocols will ensure the timing of the peak tracer concentration can be accurately and precisely measured. If the peak is smaller, or if the interval between measurements is longer, the logger is likely to record a single conductivity value for a prolonged period of time. In that case, it will be very difficult to determine the precise moment the peak was reached. For larger streams, longer reaches, and higher flows, rhodamine dye is preferable, though more expensive to employ and measure.

Although we found the salt tracer to be useful, it also has some practical and conceptual limitations. The most significant practical limitation is that the loggers require careful calibration, both with a standard solution, and with field timepieces or GMT. The timepieces in loggers tend to accumulate error over a relatively short period of time, requiring regular corrections. Another potential limitation is that repeatedly adding salt to streams could plausibly confound or influence biological responses that are also of interest, such as invertebrate drift (Wood and Dykes 2002). However, recent studies demonstrated that salt slugs (pulsed salt tracer injections) are benign from the standpoint of the biotic community (Muehlbauer et al. 2012). We have not seen any evidence for a decline in stream benthos diversity (e.g., BIBI) in our study sites, despite repeated delivery of salt solutions. Most likely, the short duration of the exposure to elevated concentrations of salt minimizes any undesirable effects on stream benthos. More biologically inert compounds, like rhodamine dye, have an advantage over salt, in this respect. However, by comparison with dye, table salt is food-grade, and is invisible to human river users. These attributes make salt less likely to alarm and concern downstream river users than rhodamine dye, so it is practical for daytime use in highly populated areas. Our own anecdotal observations indicate that juvenile salmonids show little or no behavioral response to salt slugs, even when they are in the immediate vicinity of the release point – where the salt solution is most concentrated.

Salt tracers also have conceptual limitations related to the fact that U_{MAD} integrates the effects of many stream features, but is diagnostic of none. For example, a change in U_{MAD} can be detected with confidence. That quality makes U_{MAD} a valuable indicator for detecting Urban Stream Syndrome (Walsh et al. 2005). But U_{MAD} cannot indicate whether the underlying cause of the change is the loss of pools or wood, fo example. Understanding the cause of that change is essential in forming an appropriate response. Unfortunately, detecting the change is not enough. Speculation and conjecture, preferably informed by additional physical habitat measurements, is also required inform a response. In this light, tracer methods supplement, but do not fully replace a first-hand knowledge of the site conditions, and how they have changed over time.

REFERENCES

Bernhardt, E. S., E. B. Sudduth, M. A. Palmer, J. D. Allan, J. L. Meyer, G. Alexander, J. Follastad-Shah, B. Hassett, R. Jenkinson, R. Lave, J. Rumps, and L. Pagano. 2007. Restoring rivers on reach at a time: results from a survey of U.S. river restoration practitioners. Restoration Ecology **15**:482-493.

Curran, J. H., and E. E. Wohl. 2003. Large woody debris and flow resistance in step-pool channels, Cascade Range, Washington. Geomorphology **51**:141-157.

DeGasperi, C. L., H. B. Berge, K. R. Whiting, J. J. Burkey, J. L. Cassin, and R. R. Fuerstenberg. 2009. Linking hydrologic alteration to biological impairment in urbanizing streams of the Puget Lowland, Washington, USA. Journal of the American Water Resources Association **45**:512-533.

Hart, D. D., and C. M. Finelli. 1999. Physical-biological coupling in streams: the pervasive effects of flow on benthic organisms. Annual review of ecology and systematics **30**:363-395.

Kaufmann, P. R., P. Levine, E. G. Robison, C. Seeliger, and D. V. Peck. 1999. Quantifying Physical Habitat in Wadeable Streams. EPA/620/R-99/003, Washington D.C.

Knighton, D. 1998. Fluvial forms and processes: A new perspective. Oxford University Press Inc., New York, NY.

Meyer, J. L., M. L. Paul, and W. K. Taulbee. 2005. Stream ecosystem function in urbanizing landscapes. Journal of the North American Benthological Society **24**:602-612.

Muehlbauer, J. D., J. M. Duncan, and M. W. Doyle. 2012. Benign use of salt slugs on aquatic macroinvertebrates: measuring discharge with salt during an aquatic ecology study. River Research and Applications **28**:2012.

O'Rollins, L. 1999. Analysis of habitat and geomorphological data collected at Des Moines Creek, King County, Washington. King County, Seattle, Washington.

Ramsey, S., M. Thompson, and M. Hale. 1992. Objective evaluation of precision requirements for geochemistry analysis using robust analysis of variance. Journal of Geochemical Exploration **44**.

Roni, P., M. Liermann, C. Jordan, and E. A. Steel. 2005. Steps for designing a monitoring and evaluation program for aquatic restoration. Pages 13-34 *in* P. Roni, editor. Monitoring stream and watershed restoration. American Fisheries Society, Bethesda, Maryland.

Roper, B. B., J. M. Buffington, S. Bennett, S. H. Lanigan, E. Archer, S. T. Downie, J. Faustini, T. W. Hillman, S. Hubler, K. Jones, C. Jordan, P. R. Kaufmann, G. Merritt, C. Moyer, and A. Pleus. 2010. A Comparison of the Performance and Compatibility of Protocols Used by Seven Monitoring Groups to Measure Stream Habitat in the Pacific Northwest. North American Journal of Fisheries Management **30**:565-587.

Roper, B. B., J. L. Kershner, E. Archer, R. A. Henderson, and N. Bouwes. 2002. An evaluation of physical stream attributes used to monitor streams. Journal of the American Water Resources Association **38**:1637-1646.

Roper, B. B., and D. L. Scarnecchia. 1995. Observer variability in classifying habitat types in stream surveys. North American Journal of Fisheries Management **15**:49-53.

Walsh, C. J., T. D. Fletcher, and A. R. Ladson. 2005. Stream restoration in urban catchments through redesigning stormwater systems: looking to the catchment to save the stream. Journal of the North American Benthological Society **24**:690-705.

Walsh, C. J., A. H. Roy, J. W. Feminella, P. D. Cottingham, P. M. Groffman, and R. P. Morgan II. 2009. The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society **24**:706-723.

Wang, L., T. D. Simonson, and J. Lyons. 1996. Accuracy and precision of selected stream habitat estimates. North American Journal of Fisheries Management **16**:340-347.

Whitacre, H. W., B. B. Roper, and J. L. Kershner. 2007. A comparison of protocols and observer precision for measuring physical stream attributes. Journal of the American Water Resources Association **43**:923-937.

Wood, P. J., and A. P. Dykes. 2002. The use of salt dilution gauging techniques: ecological considerations and insights. Water Research **36**:3054-3062.

Woodsmith, R. D., J. R. Noel, and M. L. Dilger. 2005. An approach to effectiveness monitoring of floodplain channel aquatic habitat: channel condition assessment. Landscape and Urban Planning **72**:177-204.

Appendix G.
BIBI scoring thresholds and biological condition categories

Scoring thresholds and expected response to human disturbance for the ten BIBI biological metrics. Criteria require identification of most insects to species level or lowest practical taxonomic resolution and chironomids (non-biting midges) to family level.

Measured Biological Metrics	Expected Response	Score		
		1	3	5
Taxa Richness & Composition				
Total taxa	Decrease	< 15	15-28	> 28
Ephemeroptera taxa (mayflies)	Decrease	< 4	4-8	> 8
Plecoptera taxa (stoneflies)	Decrease	< 3	3-7	> 7
Trichoptera taxa (caddisflies)	Decrease	< 5	5-10	≥ 10
Long-lived taxa	Decrease	< 2	2-4	> 4
Tolerant & Intolerant				
Intolerant taxa	Decrease	< 2	2-3	> 3
% Tolerant	Increase	≥ 50	19-50	≤ 19
Feeding & Habits				
% Predators	Decrease	< 10	10-20	≥ 20
Clinger taxa	Decrease	< 8	8-18	> 18
Population				
% Dominance	Increase	≥ 80	60-80	< 60

BIBI biological condition categories. Ten individual BIBI metric scores are combined to give a total BIBI score ranging from 10-50, which can be classified into five levels of biological condition. Modified from Karr et al. (1986) by Morley (2000).

Biological Condition	BIBI Range	Description
Excellent	46-50	Comparable to least disturbed reference condition; overall high taxa diversity, particularly of mayflies, stoneflies, caddis flies, long-lived, clinger, and intolerant taxa. Relative abundance of predators high.
Good	38-44	Slightly divergent from least disturbed condition; absence of some long-lived and intolerant taxa; slight decline in richness of mayflies, stoneflies, and caddis flies; proportion of tolerant taxa increases
Fair	28-36	Total taxa richness reduced – particularly intolerant, long-lived, stonefly, and clinger taxa; relative abundance of predators declines; proportion of tolerant taxa continues to increase
Poor	18-26	Overall taxa diversity depressed; proportion of predators greatly reduced as is long-lived taxa richness; few stoneflies or intolerant taxa present; dominance by three most abundant taxa often very high
Very Poor	10-16	Overall taxa diversity very low and dominated by a few highly tolerant taxa; mayfly, stonefly, caddis fly, clinger, long-lived, and intolerant taxa largely absent; relative abundance of predators very low.

Appendix H.

High pulse counts (HPCs) for ten modeled watersheds used to scale hydrologic influence of land cover and geology in a hydrologic condition index (HCI)

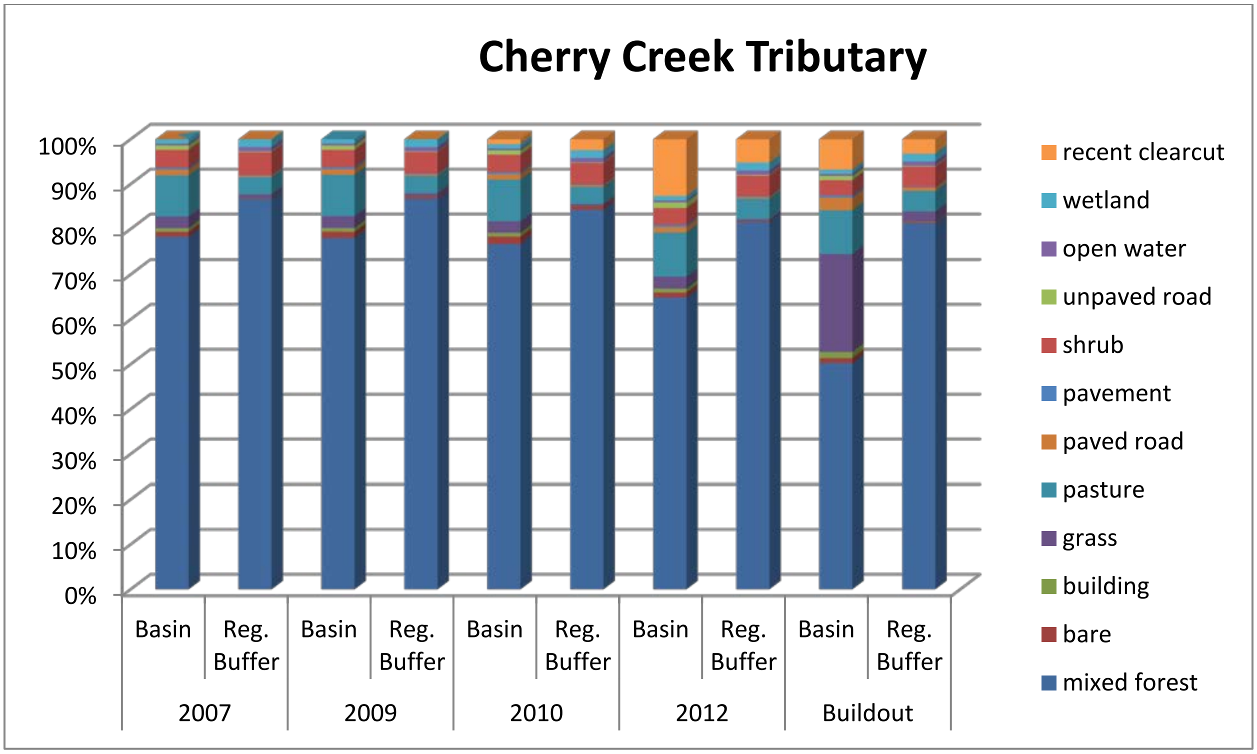
High pulse counts (HPCs) for ten modeled watersheds used to scale hydrologic influence of land cover and geology in a hydrologic condition index (HCI).

Geology	Land Cover	Average HPC (not incl. Walker Cr.)	Hamm Creek (set 1)	Big Soos	Miller Creek (set 2)	Black River	Des Moines Creek (Set 3)	Mill	Newaukum (set 4)	Olson	Duwamish LCL1 (set 5)	Walker Creek*
	mixed forest	3.976321	2.393443	2.590164	2.672131	2.786885	3.655738	4.508197	4.606557	5.524590	7.049180	10.508197
	shrub	4.422587	2.639344	2.967213	3.311475	3.688525	4.475410	4.311475	6.016393	5.311475	7.081967	14.016393
	pasture	4.653916	2.803279	2.803279	4.032787	4.049180	4.622951	4.360656	6.590164	5.016393	7.606557	15.639344
	wetland	5.508197	2.901639	4.049180	4.868852	7.098361	4.540984	4.836066	7.524590	5.508197	8.245902	8.229508
	clear cut	5.528233	3.819672	3.163934	5.032787	5.049180	5.360656	4.540984	8.606557	5.377049	8.803279	19.098361
	grass	6.415301	5.672131	3.229508	5.213115	5.393443	6.032787	5.754098	9.983607	7.983607	8.475410	16.459016
Till	**bare**	7.380692	5.114754	3.311475	8.524590	8.459016	7.901639	5.163934	10.508197	5.983607	11.459016	21.983607
	building	31.187614	30.508197	30.065574	34.803279	34.491803	33.491803	29.000000	29.622951	26.868852	31.836066	34.213115
	pavement	32.584699	26.540984	33.819672	36.885246	31.704918	36.508197	25.704918	34.032787	32.327869	35.737705	36.393443
	open water	34.295082	27.934426	34.803279	38.163934	32.819672	38.131148	27.491803	36.655738	34.868852	37.786885	37.754098
	unpaved road	35.233151	33.983607	34.016393	37.180328	37.163934	36.901639	33.360656	34.754098	33.065574	36.672131	36.721311
	paved road	35.657559	34.360656	34.245902	37.655738	37.327869	37.344262	33.934426	35.180328	33.655738	37.213115	37.081967
	mixed forest	3.777778	2.213115	2.065574	2.065574	5.442623	3.360656	4.196721	3.688525	5.540984	5.426230	0.803279
	shrub	3.870674	2.229508	2.196721	2.131148	5.311475	3.393443	4.213115	4.540984	5.245902	5.573770	0.885246
	pasture	4.071038	2.295082	2.573770	2.213115	5.295082	3.262295	4.163934	6.081967	5.229508	5.524590	0.967213
	clear cut	4.098361	2.295082	2.819672	2.213115	5.295082	3.262295	4.163934	6.081967	5.229508	5.524590	0.950820
	grass	4.300546	2.606557	3.131148	2.032787	5.524590	3.409836	4.459016	5.655738	6.180328	5.704918	1.721311
	bare	5.357013	3.245902	3.049180	3.311475	7.311475	4.557377	5.131148	7.639344	6.114754	7.852459	3.754098
Outwash	**wetland**	5.508197	2.901639	4.049180	4.868852	7.098361	4.540984	4.836066	7.524590	5.508197	8.245902	8.229508
	building	31.446266	31.409836	30.557377	35.459016	31.737705	33.245902	29.016393	31.737705	27.868852	31.983607	35.590164
	pavement	32.719490	26.573770	33.901639	37.114754	31.016393	36.475410	25.803279	35.049180	32.918033	35.622951	37.032787
	open water	34.202186	27.639344	34.803279	38.081967	32.819672	37.934426	27.278689	36.606557	34.836066	37.819672	37.819672
	unpaved road	35.406193	34.016393	34.081967	37.524590	36.639344	37.049180	33.360656	35.491803	33.672131	36.819672	37.295082
	paved road	35.668488	34.196721	34.278689	37.672131	37.049180	37.229508	33.704918	35.868852	33.918033	37.098361	37.491803

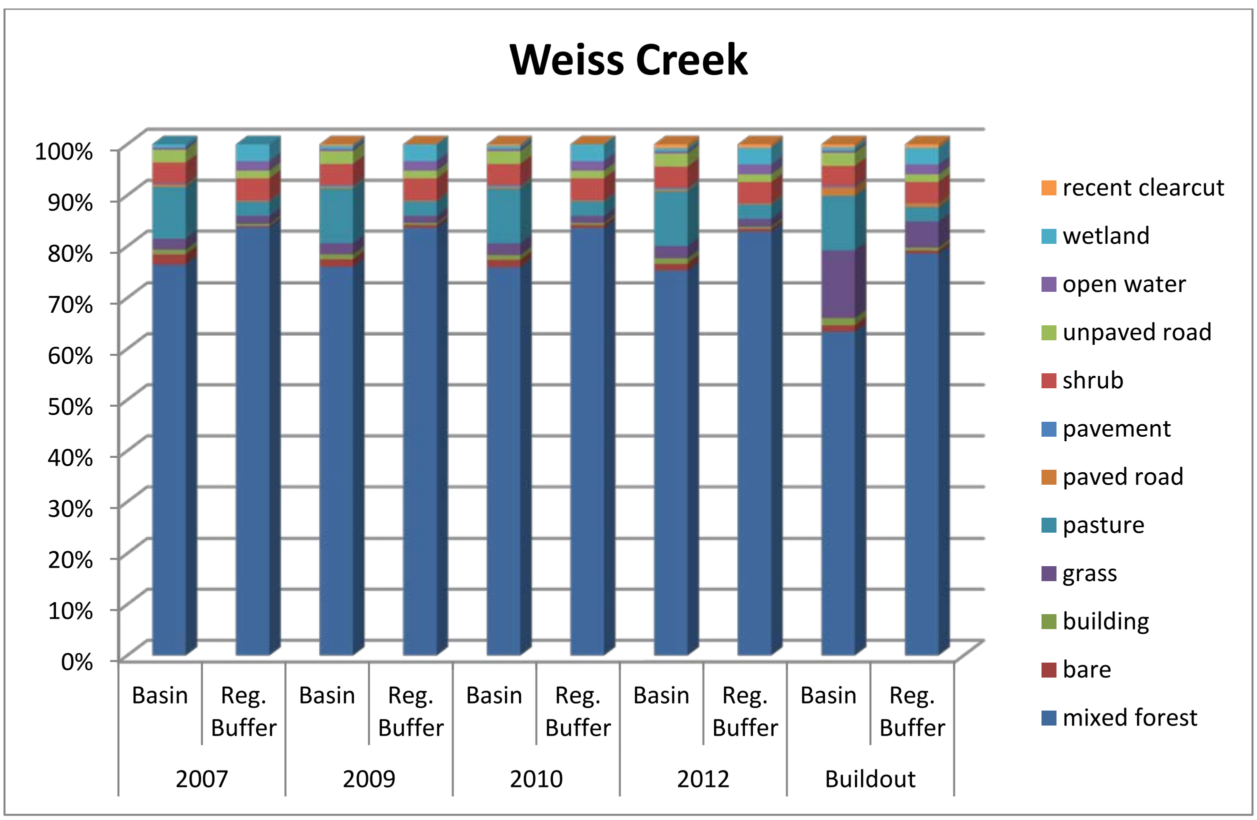
*For reference only - not used for scoring.

Appendix I.

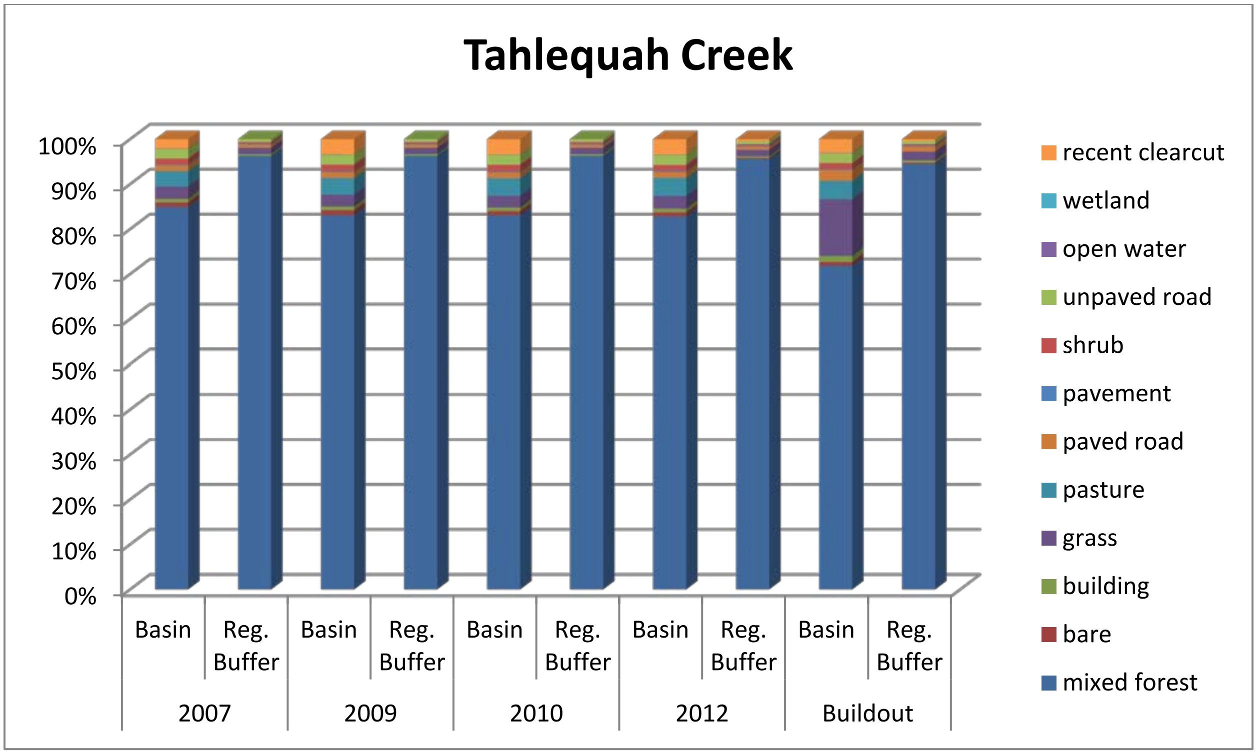
Unweighted Aggregate Tables by Watershed

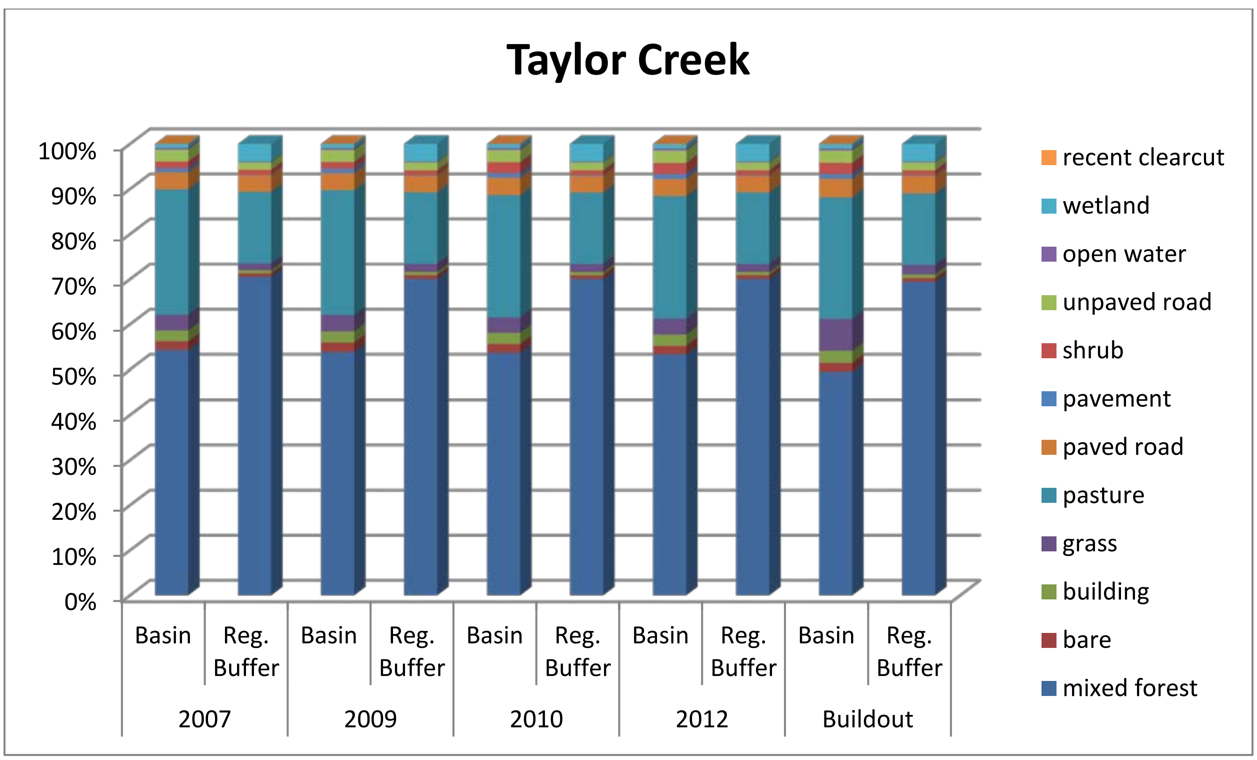


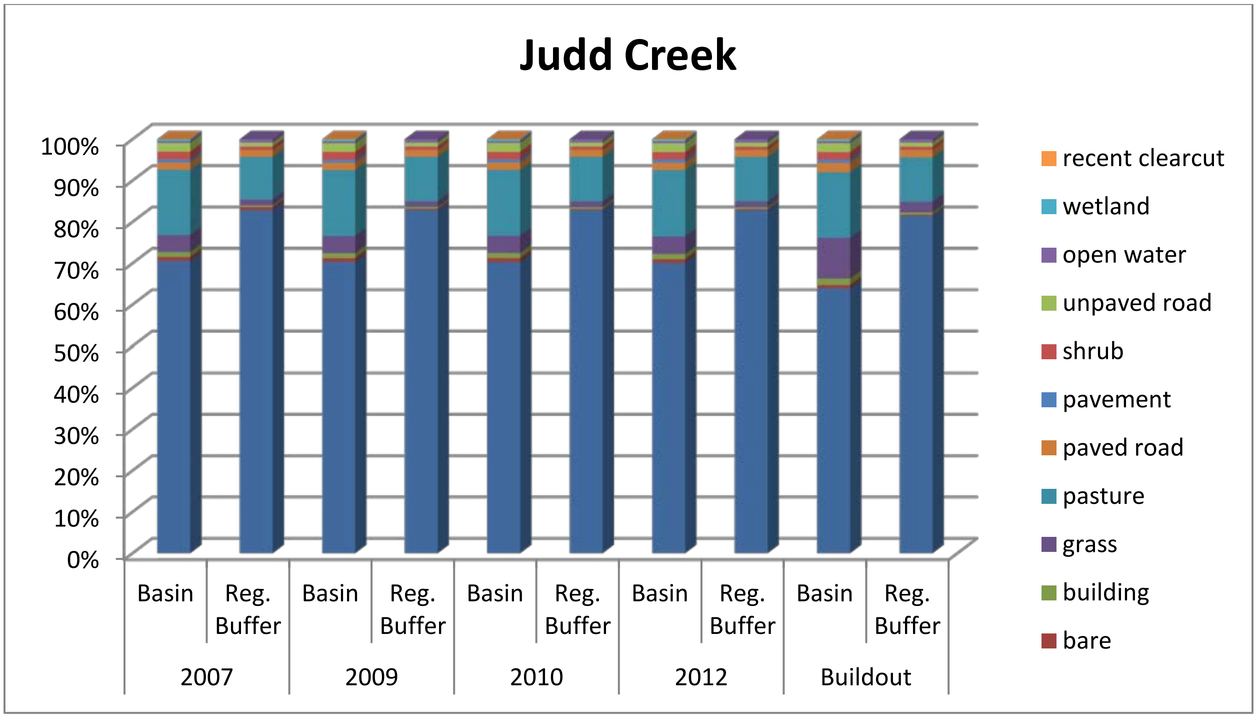
Cherry Creek Tributary
100%
90%
80%
70%
60%
50%
40%
30%
20%
10%
0%
Basin
Reg. Buffer
2007
2009
2010
2012
Buildout
recent clearcut
wetland
open water
unpaved road
shrub
pavement
paved road
pasture
grass
building
bare
mixed forest

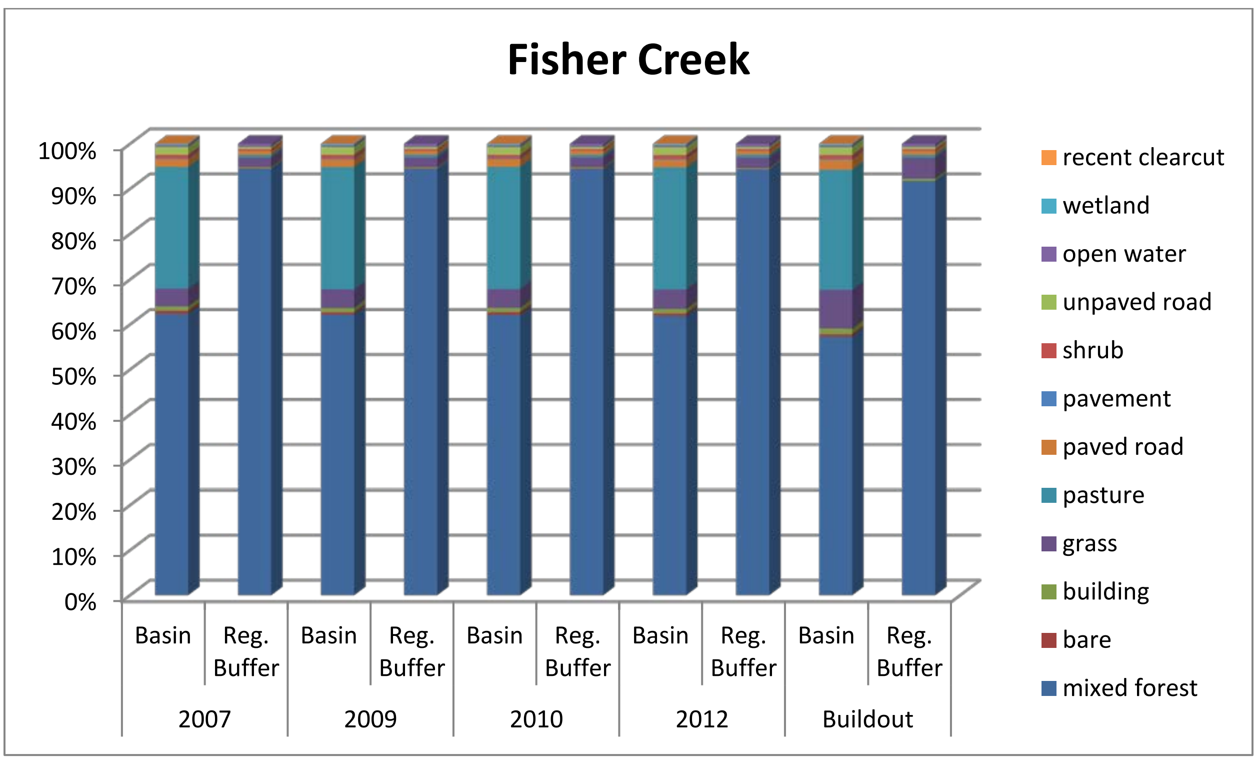


Weiss Creek
100%
90%
80%
70%
60%
50%
40%
30%
20%
10%
0%
Basin
Reg. Buffer
2007
2009
2010
2012
Buildout
recent clearcut
wetland
open water
unpaved road
shrub
pavement
paved road
pasture
grass
building
bare
mixed forest


Tahlequah Creek
100%
90%
80%
70%
60%
50%
40%
30%
20%
10%
0%
Basin
Reg. Buffer
Basin
Reg. Buffer
Basin
Reg. Buffer
Basin
Reg. Buffer
Basin
Reg. Buffer
2007
2009
2010
2012
Buildout
recent clearcut
wetland
open water
unpaved road
shrub
pavement
paved road
pasture
grass
building
bare
mixed forest


Taylor Creek
100%
90%
80%
70%
60%
50%
40%
30%
20%
10%
0%
Basin
Reg. Buffer
Basin
Reg. Buffer
Basin
Reg. Buffer
Basin
Reg. Buffer
Basin
Reg. Buffer
2007
2009
2010
2012
Buildout
recent clearcut
wetland
open water
unpaved road
shrub
pavement
paved road
pasture
grass
building
bare
mixed forest


Judd Creek
100%
90%
80%
70%
60%
50%
40%
30%
20%
10%
0%
Basin
Reg. Buffer
2007
2009
2010
2012
Buildout
recent clearcut
wetland
open water
unpaved road
shrub
pavement
paved road
pasture
grass
building
bare


Fisher Creek
100%
90%
80%
70%
60%
50%
40%
30%
20%
10%
0%
Basin
Reg. Buffer
2007
2009
2010
2012
Buildout
recent clearcut
wetland
open water
unpaved road
shrub
pavement
paved road
pasture
grass
building
bare
mixed forest