Monitoring physical changes in stream channels with salt tracers

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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 2^{nd}-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 ($1-\beta$ 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 2^{nd}- and 3^{rd}-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.

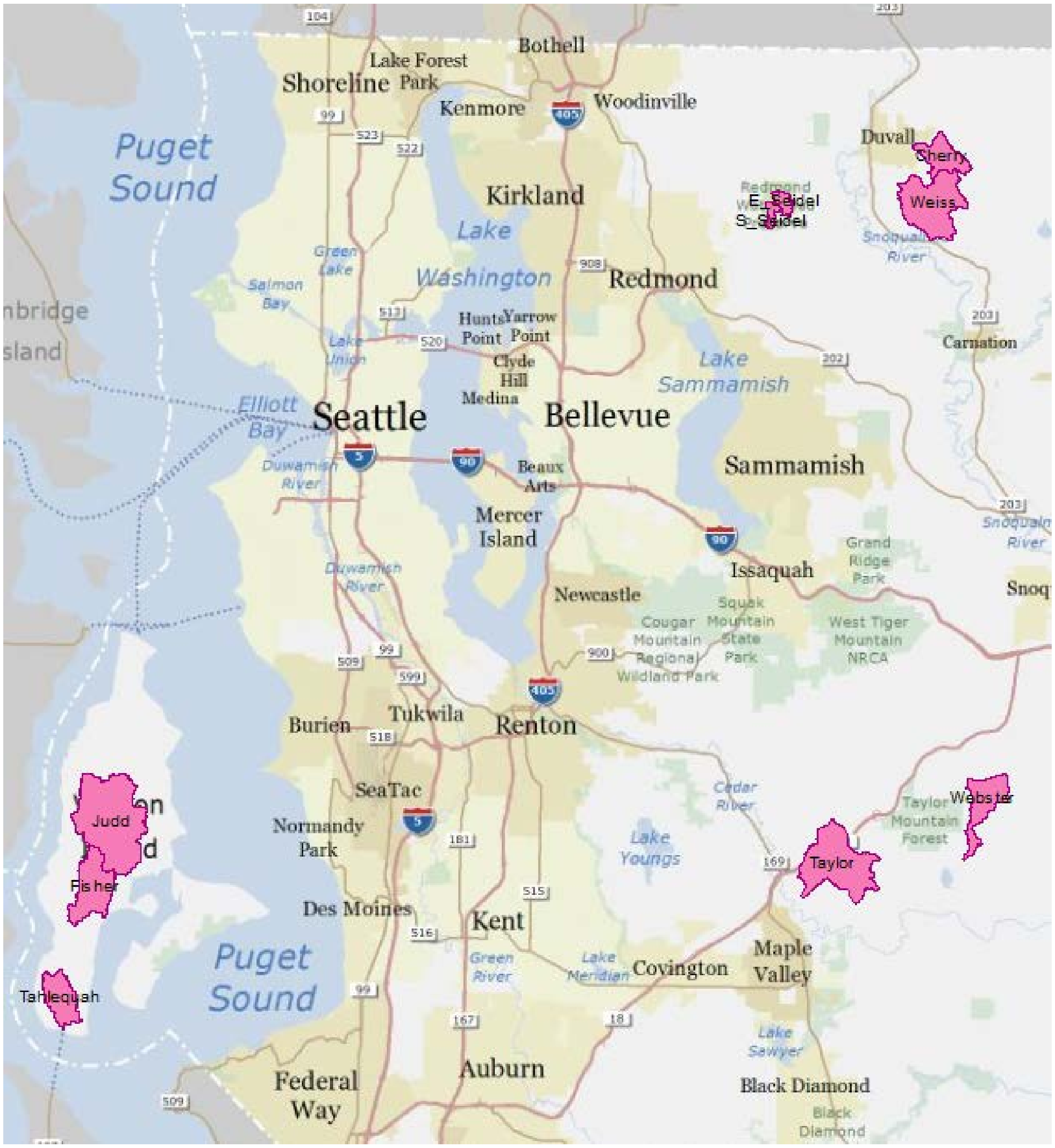


Table 1. Study streams.

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.

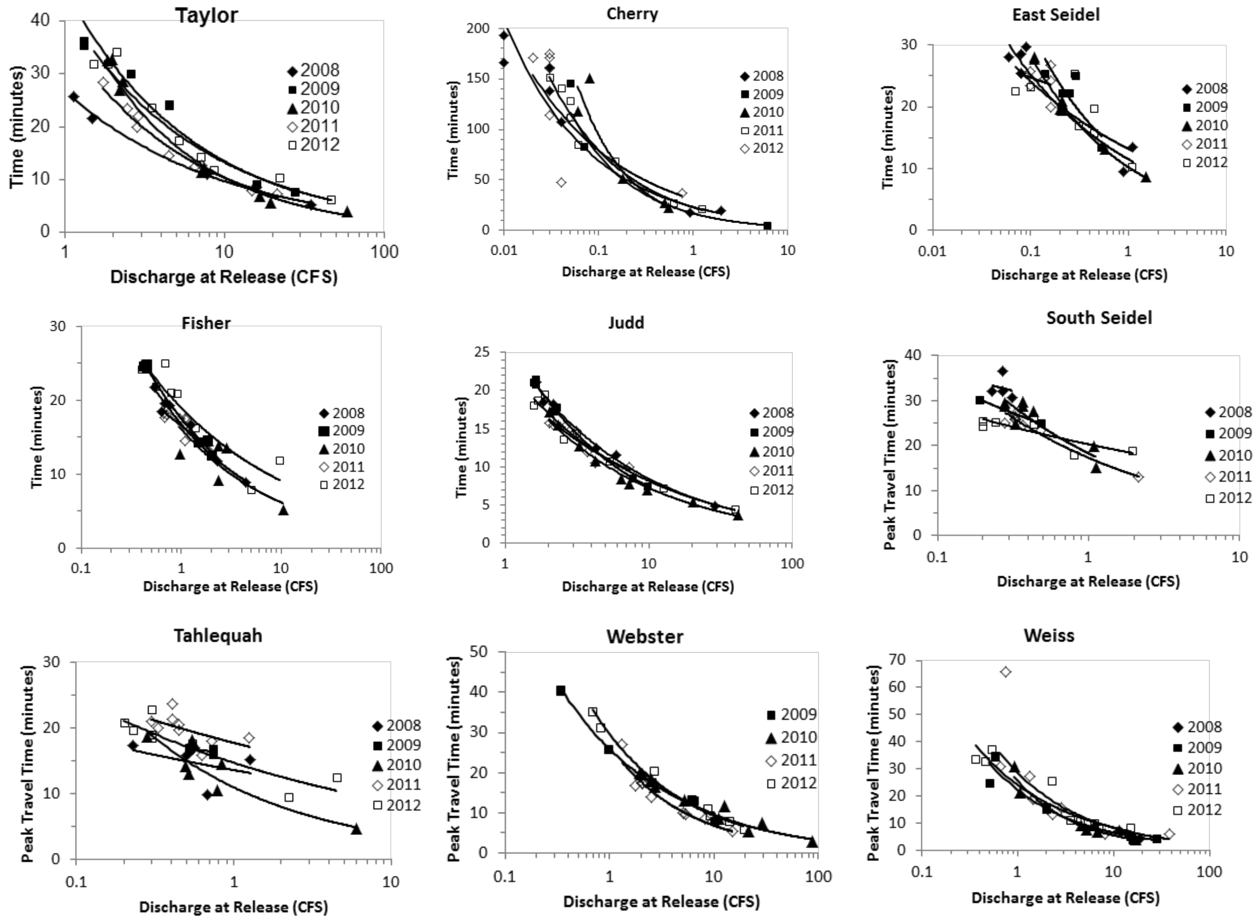
Table 4. Number of tracer runs in peak travel time T_n was recorded for at least the combined reach.

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.

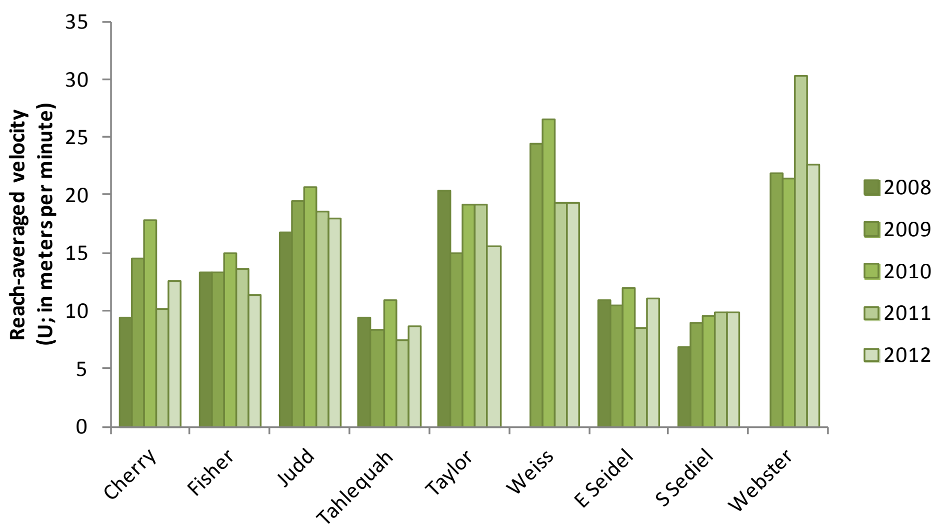
Site	Equation (y)					n					r^2				
	2008	2009	2010	2011	2012	2008	2009	2010	2011	2012	2008	2009	2010	2011	2012
Taylor	$26.76x^{-0.441}$	$46.032x^{-0.534}$	$46.36x^{-0.65}$	$36.928x^{-0.552}$	$42.093x^{-0.502}$	5	7	9	9	9	0.99	0.96	0.98	0.98	0.96
Cherry	$23.229x^{-0.474}$	$16.751x^{-0.647}$	$14.514x^{-0.813}$	$22.855x^{-0.54}$	$18.412x^{-0.557}$	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.



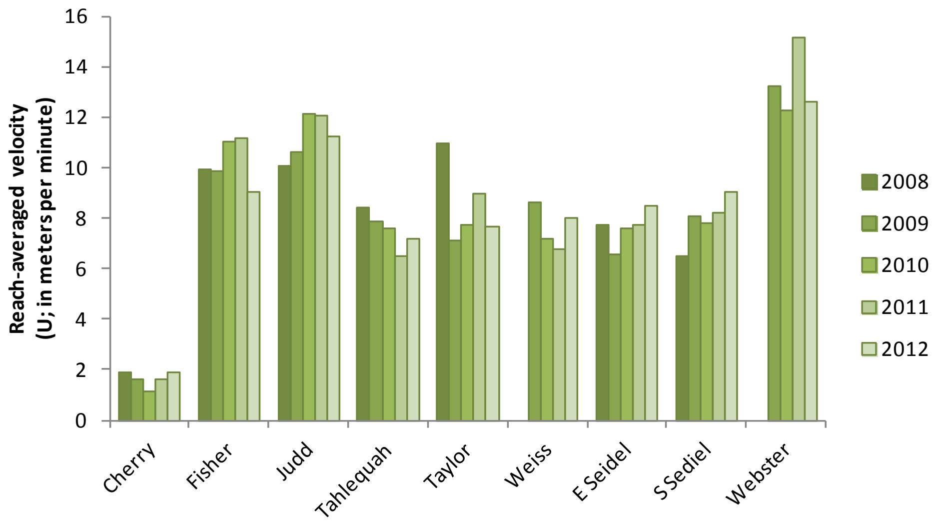


Table 7. U_{25} in meters per minute for each of three study reaches at each site and year.

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 75th and 25th 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.

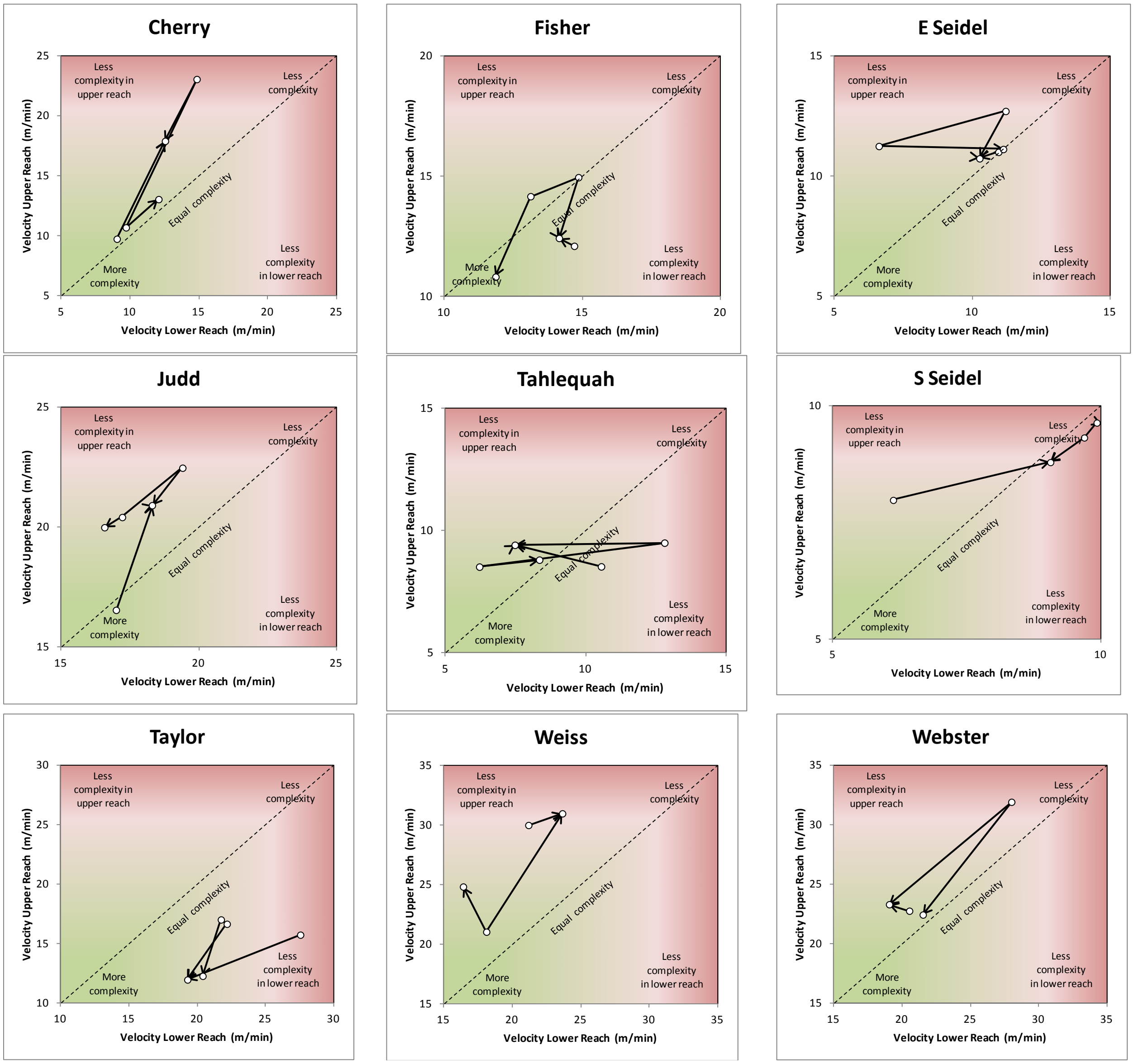


Table 9. Daily discharge statistics for study streams from the beginning of the record through WY 2012. Units are in cubic feet per second (cfs).

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.

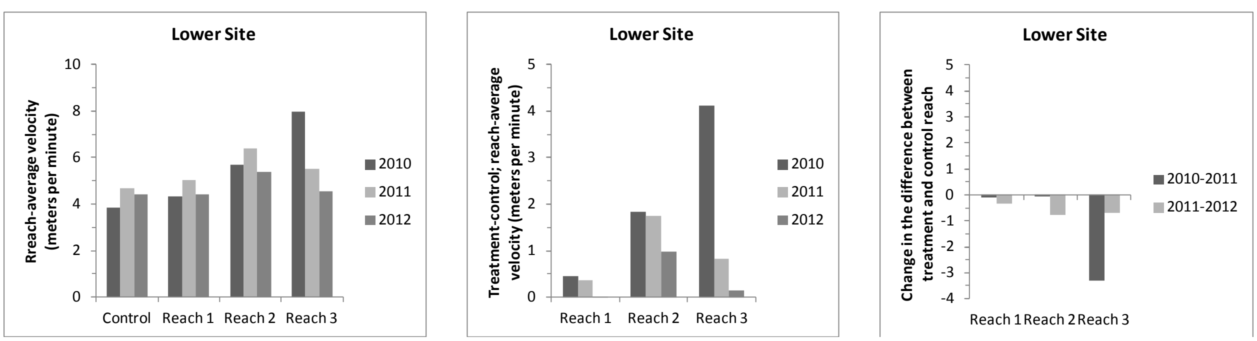
Type	Stream	Replicates	Measurement error range (%)	c_v range (%)
Treatment	Cherry	2	0.9-5.6	0.7-4.1
	Fisher	1	2.1	1.5
	Judd	2	1.2-2.9	0.8-2.0
	Tahlequah	2	1.5-4.7	1.0-3.2
	Taylor	2	1.0-2.1	0.7-1.5
	Weiss	3	2.2-3.2	1.5-2.3
Reference	E Seidel	1	0	0
	S Seidel	0	-	-
	Webster	2	0.6-1.9	0.4-1.3
All sites	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

Control Reach 1 Reach 2 Reach 3

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 $\mathrm{U_{MAD}}$ 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.

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